Sustainability Assessment of Land Use Systems

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Kurzfassung

Angesichts der wachsenden Beanspruchung natürlicher Ressourcen durch die Menschheit und der global ungleichen Verteilung der Vorteile ihrer Nutzung ist “Nachhaltige Entwicklung” zum einem weithin anerkannten Leitbild geworden. Das Ziel dieser Arbeit ist, einen Indikatorensatz zu konstruieren und mit ihm die Nachhaltigkeit eines Spinat-Produktionssystems im Kreis Borken (Nordrhein-Westfalen) zu bewerten.

In Kapitel 1 wird ein kurzer Überblick über die Initiative gegeben, die Rahmen für diese Arbeit bildet. In Kapitel 2 werden grundlegende Konzepte der Nachhaltigkeitesbewertung in der Landwirtschaft besprochen. Dabei werden drei Phasen der indikatoren-basierten Nachhaltigkeitsbewertung herausgearbeitet: (1) Konstruktion des Indikatorensatzes; (2) Auswertung des Indikatorensatzes, und (3) Strategieentwicklung. Diese Arbeit folgt mit ihren Hauptkapiteln diesen drei Phasen.

Kapitel 3 ist der ersten Phase gewidmet, der **Konstruktion des Indikatorensatzes**. Speziell befaßt es sich damit, den Begriff “Nachhaltige Landwirtschaft” zu konkretisieren, um ihn als Grundlage für wissenschaftliche Untersuchungen nutzen zu können. In der derzeitigen Literatur zum Thema wird diese Phase oft oberflächlich und nicht systematisch behandelt. Da es, unseres Wissens, keine Methode gibt, die die Konstruktion von Indikatorensätzen systematisch, nachvollziehbar und reproduzierbar gestaltet, wird hier eine solche Methode entwickelt. Sie erstellt zunächst ein Inventar potentieller Probleme, indem sie den laufenden Diskurs über nachhaltige Landwirtschaft in der Literatur nutzt. Und prüft dann, ob diese potentiellen Probleme im konkreten Fall tatsächlich von Bedeutung sind. Am Beispiel des Kreises Borken wird diese Methode demonstriert, um Umweltprobleme auszumachen, die für die Nachhaltigkeit der Landwirtschaft dort relevant sind.

Kapitel 4 unternimmt einen Exkurs zur Schätzung von Nährstoffverlusten von landwirtschaftlich genutzten Flächen. Nährstoffverluste aus der Landwirtschaft verursachen eine Reihe von Umweltproblemen. Sie möglichst genau und verläßlich abschätzen zu können ist eine wichtig Voraussetzung, um sachkundige Entscheidungen über unterschiedliche Bewirtschaftungsmaßnahmen treffen zu können. Einige häufig verwendete einfache Schätzmethoden für Nährstoffverluste werden besprochen. Es zeigt sich, daß sich mit ihnen, trotz ihrer Einfachheit, Nährstoffverluste von landwirtschaftlichen Flächen recht präzise und verläßlich schätzen lassen.

In Kapitel 5 wird eine Methode entwickelt, die sich mit der zweiten Phase indikatoren-basierter Nachhaltigkeitsbewertung befaßt, der **Auswertung des Indikatorensatzes**.

Die Methode erfüllt vier spezifische Anforderungen: (1) sie ist für Umwelt-, soziale und wirtschaftliche Indikatoren gleichermaßen geeignet; (2) sie erlaubt den Vergleich sehr unterschiedlicher Landnutzungssysteme; (3) sie integriert über die unterschiedlichen räumlichen Skalenniveaus, auf denen unterschiedliche Probleme auftreten; und (4) sie vermischt nicht deskriptive mit normativen Elementen der Indikatorenauswertung.

Die Methode umfaßt ein Standardisierungs- und ein Nachhaltigkeits-Bewertungsverfahren. Das *Standardisierungsverfahren* macht unterschiedliche Indikatoren vergleichbar und reflektiert, wie weit sich der tatsächliche Indikatorwert von einem Sollwert unterscheidet. Das *Nachhaltigkeits-Bewertungsverfahren* ordnet die standardisierten Indikatoren drei diskreten Nachhaltigkeitsklassen zu. Wir wenden diese Methode dann auf das Beispielproduktiossystem im Kreis Borken an. Unterschiede zwischen einzelnen standardisierten Indikatoren werden hauptsächlich vom Beitrag des Produktionssystems zum Gesamtproblem bestimmt, das der jeweilige Indikator beschreibt.

Durch den Standardisierungsprozeß "erben" und kombinieren die standardisierten Indikatoren die methodische Unsicherheit ihrer Eingangsdaten. Aufgrund von stochastischen Simulationen mit drei ausgewählten Indikatoren schätzen wir, daß der Standardisierungsprozeß die Unsicherheit um Faktor 2.0 bis 2.5 erhöht.

Kapitel 6 befaßt sich mit der dritten Phase Indikatoren-basierter Nachhaltigkeitsbewertung, der **Stategieentwicklung**. Minderungspotentiale und –möglichkeiten für die Umwelteffekte des Spinatproduktionssystems werden diskutiert. Eine Reihe von praktischen Empfehlungen wird gegeben. Kapitel 7 diskutiert die Umwelteffekte des Spinatproduktionssystems vor dem Hintergrund der finanziellen und sozialen Leistungen, die das Produktionssystem schafft.

Schließlich verortet und diskutiert Kapitel 8 die Methodik, die hier entwickelt wurde, im Kontext andere Bewertungssystemen. Es zeigt sich, daß die Methode Schwächen anderer verfügbare Methoden überwindet: Sie (1) ist umfassend, denn sie berücksichtigt ausdrücklich die Konstruktion des Indikatorensatzes; (2) erlaubt den Vergleich zwischen unterschiedlichen Landnutzungssystemen, denn sie kann auf jedes beliebige System angewendet werden, unabhängig von Art, Lage und Skalenniveau; (3) kann unterschiedliche Nachhaltigkeitsdimensionen integrieren (etwa die Umwelt-, soziale und wirtschaftliche Dimension); und sie (4) trennt eindeutig zwischen deskriptiven und normativen Elementen und erlaubt dadurch mit Normativität im wissenschaftlichen Prozeß umzugehen. Dies begründet drei große Vorteilen der Methode: Sie ist umfassend, flexible und transparent.

Schlagworte: Nachhaltige Landwirtschaft; Nachhaltigkeitsbewertung; Landnutzungssysteme

Abstract

In the face of increasing human pressure on the world's natural resources and global inequity in the benefits of their use, sustainable development has become a widely recognised guiding principle for policy and development. The objective of this study is to construct an indicator set and evaluate the sustainability of a spinach production system in the County of Borken, Northwest Germany.

In Chapter 1 a brief overview is given of the initiative that forms the context of this work. Chapter 2 reviews basic concepts of sustainable agriculture assessment. Three stages of indicator-based sustainability assessment are identified: (1) Indicator set construction; (2) Indicator set evaluation; and (3) Strategy development. These three stages also provide the structure for the main chapters of this study.

Chapter 3 addresses the first of these stages, **Indicator set construction**. Specifically, it deals with setting up workable conceptions of the term 'sustainable agriculture' as the basis of scientific inquiry. Reviewing the current literature on sustainable agriculture, we found that Stage 1 is often treated superficially in the literature on sustainable agriculture and not addressed in a systematic way. As there is, to our knowledge, no method that would address indicator set construction in a systematic, transparent and reproducible way, a method is developed to meet these requirements. It first makes an inventory of potential sustainability issues from the ongoing discourse on sustainable agriculture; and then tests whether or not a potential issue is actually relevant within a given context. The method is then applied to identify environmental issues of concern for agricultural sustainability in the case study area in the County of Borken.

Chapter 4 is an excursus on estimating nutrient losses from agricultural fields. Nutrient losses from agriculture are a significant driver of many environmental issues and estimating them accurately and reliably is a key requirement for making informed decisions about different agricultural management options. A number of often used simple estimation methods for such nutrient losses are reviewed. It is found that, in spite of their simplicity, they can well serve to estimate nutrient losses from agricultural fields relatively accurately and reliably.

In Chapter 5, a methodology is introduced that addresses the second stage of indicator based sustainability assessment, **Indicator set evaluation**. The method is designed to meet four specific requirements, namely (1) to be applicable to environmental, social

and economic indicators alike; (2) to allow for comparing very different land use systems; (3) to integrate over various spatial scales at which diverse issues emerge; and (4) to be clear about the descriptive and normative elements of the evaluation.

The method comprises a standardisation and a sustainability valuation procedure. The *standardisation procedure* makes different indicators comparable and accounts for the distance between the actual and the target value for a particular indicator. The *sustainability valuation procedure* assigns the standardised indicators to three discrete sustainability classes.

We then apply the method to the case study production system in the County of Borken. Differences between individual standardised indicators are predominantly determined by the relative share that the land use system contributes to a particular issue. Through the standardisation process, the standardised indicators 'inherit' and combine the methodological uncertainty from their input data. Based on stochastic simulation with three selected indicators, it is estimated that standardisation procedure increases the uncertainty by a factor of 2.0 to 2.5.

Chapter 6 addresses the third stage of indicator-based sustainability assessment, **Strategy development**. Mitigation potentials and options for the environmental impacts of the spinach production system are discussed. A number of practical recommendations regarding nutrient and soil related issues are made. Chapter 7 discusses the environmental impacts of the case study system within the context of financial and social benefits, which are also generated by the production system.

Chapter 8 finally discusses the methodology developed here within the context of comparable assessment schemes. It is found that the method overcomes shortcomings of other presently available methods. Namely it (1) is comprehensive in explicitly including indicator set construction; (2) allows for comparing diverse land use systems, because it can be applied to any land use system, regardless of type, location and scale level; (3) can integrate diverse sustainability dimensions, (e.g. environmental, social and economic dimension); and (4) clearly separates descriptive and normative elements and thereby allows for managing normativity within the scientific process. This results in three main benefits of the method, comprehensiveness, flexibility and transparency.

Keywords: sustainable agriculture; sustainability assessment; land use systems.

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Abbreviations

(–)	Unit symbol for dimensionless entities
CAN	Calcium ammonium nitrate
DM	Dry matter
eq	Equivalents
FM	Fresh matter
K	Potassium
LAP	Lead Agricultural Project within the Unilever Sustainable Agriculture Initiative
N	Nitrogen
NM VOC	Non-methane volatile organic compounds
ODP	Ozone depletion potential
OM	Organic matter
P	Phosphorus
TAN	Total ammoniacal nitrogen
SOM	Soil organic matter
UAN	Urea ammonium nitrate solution

Model parameters

A_i	issue specific area for issue *i*
cl	Cut-off point in sustainability valuation function
L_{act}	Actual impact level
L_{crit}	Critical impact level
NF	Normalisation factor
$sdt\ I_s$	Standardised indicator value
SF	Severity factor
$val\ I_s$	Indicator value after sustainability valuation

CHAPTER 1

Background and context: Iglo and Unilever's Sustainable Agriculture Initiative

Unilever's interest in sustainable agriculture

This study has been conducted in the course of the Iglo Lead Agricultural Project under the Unilever Sustainable Agriculture Initiative (Unilever, 2004). It aims at constructing an indicator set and evaluating Iglo's spinach operation in the County of Borken, Northwest Germany. We shall therefore briefly introduce the initiative and its motivation.

Agriculture provides more than two-thirds of the raw materials for Unilever's products. The company is among the world's largest users of agricultural raw materials (Unilever, 2002). These include vegetable oils (such as palm oil, sunflower, soy and rape seed), tomato-paste, tea and frozen vegetables (such as peas and spinach). These agricultural raw materials are grown on own plantations as well as produced by contract growers and bought on the open market.

Unilever's interest in sustainability is twofold: It is (1) to ensure continued access to the key agricultural raw materials – i.e. secure the supply chain; and (2) to anticipate growing consumer concerns about the origin of foods and allow customers and consumers to influence the sourcing of raw materials through their buying habits – i.e. secure the markets.

The company therefore launched three long-term sustainability initiatives on agriculture, fisheries and water. The sustainable agriculture programme came into being in the mid-1990s. The focus of the initiative is on improving the sustainability of current farming methods in particular locations. To this end Unilever started a number of Lead Agriculture Projects (LAPs) where there is directly influence on agricultural practices, i.e. its own plantations and in contract farming. LAPs were initiated in five key crops, where Unilever processes a significant share of the world volume. These are frozen spinach (28%, two LAPs), black tea (16%, three LAPs), frozen peas (13%, one LAP) tomatoes (7%, three LAPs) and palm oil (7%, two LAPs). One of the two spinach LAPs is located in the County of Borken, Northwest Germany, where Unilever produces frozen spinach for its Iglo brand.

Unilever's approach to sustainable agriculture

In 1995 Unilever commissioned a study, which captured the opinions of leading opinion formers among customers, farmers, the agribusiness, the food industry, retailers and non-government organisations with an interest in the environment and sustainable development. A subsequent workshop in 1998 drew participants worldwide from within the company and among agricultural experts from academia (Unilever, 2002). At this workshop, a mission statement was set up (see www.growingforthefuture.com) and four principles for sustainable agriculture were defined:

1. Producing crops with high yield and nutritional quality to meet existing and future needs, whilst keeping resource inputs as low as possible;
2. Ensuring that any adverse effects on soil fertility, water and air quality and biodiversity from agricultural activities are minimized and positive contribution will be made where possible;
3. Optimising the use of renewable resources whilst minimizing the use of non-renewable resources;
4. Sustainable agriculture should enable local communities to protect and improve their well-being and environments.

A third outcome of the workshop was the identification of ten broad areas of concern ('indicators' in the Unilever terminology), by which 'sustainable agriculture' should be concretised within the Lead Agricultural Projects:

1. Soil Fertility
2. Soil Loss
3. Nutrients
4. Pest Management
5. Biodiversity
6. Product Value
7. Energy
8. Water
9. Social & Human Capital
10. Local Economy.

Within each of these ten areas of concern, measurable parameters were to be defined, which reflect the local situation and the type of crop. Their purpose is to measure and monitor progress and improvements made.

To advise on this process, Unilever has set up a Sustainable Agriculture Advisory Board, which comprises eleven individuals from research institutes, the voluntary sector and academia. Members are selected for their individual quality, rather than to represent their organisations (details on the members and their backgrounds can be found on www.growingforthefuture.com).

The Unilever approach to sustainable agriculture is to involve a wide range of stakeholders (e.g. from non-government organisations, research institutes, agricultural experts and community organisations). Stakeholders are involved both at the international level and at the local level in the individual LAPs. As the company realises that its influence at the farm level is limited, it also sought the co-operation of others in the food industry. Together with Groupe Danone and Nestlé, Unilever founded the Sustainable Agriculture Initiative Platform (SAI Platform), a global food industry platform on sustainable agriculture (see www.saiplatform.org). It aims to gather knowledge about sustainable agriculture and develop common industry standards at the pre-competition stage.

Iglo's spinach operation in Borken, Germany

This study is concerned with the Lead Agricultural Project (LAP) in the County of Borken, Northwest Germany, where the Unilever's subsidiary company Iglo produces frozen spinach. More than 90% of the raw material is sourced from local contracted farmers. Iglo started contract growing in the County of Borken in 1963. Today, some one hundred contract farmers produce 30,000 to 40,000 t of spinach per year on roughly 1,000 ha. This is about 12% of the spinach volume processed for frozen foods globally.

Site

The growing area is located mainly in the southern part of the County of Borken (*Altkreis Borken*) in the Westmünsterland region. The region is a traditional livestock breeding area, where a shift from pasture to in-door keeping and large-scale land melioration and consolidation have allowed for intensivation and specialisation of agriculture during the 1960's to 1980's. Today nearly 60% of the arable land is cropped to fodder maize (LK WL, 2002). Nevertheless, there are still significant areas of woodland, hedges and small structural elements within the agricultural landscape (15 to 20% of the county area), which have coined the term 'park-landscape'.

Climate

The climate is temperate oceanic with an average temperature of 9°C, ranging from between 1.5°C in January to 17.5°C in July. The precipitation is 750 to 800 mm per year, which is relatively evenly distributed (67 ±16 mm per month, with highest rainfall in December and June/July and lowest in February and October). There are 10 to 15 days per month with more than 1 mm precipitation. The mean relative air humidity is 80% (von Kürten, 1977; DWD, 2001).

Soils

Soil types are predominantly sand (64%), loamy sand (24%) and sandy loam (8%) on diluvial boulder clay and wind borne sands (Lammers, 1999; Walter Markfort, 2001; personal communication). The average soil organic matter contents is 2.9% (1.2 to 5.8%), the water capacity is 20 to 30% and cation exchange capacity 12 to 16 cmol kg^{-1}. The pH values are maintained at 5.5 to 6.5. Soils are poorly buffered and prone to cation leaching and acidification (Walter Markfort, 2001; personal communication). Many fields have high levels of plant available potassium and phosphate due to long-term intensive manuring (Lammers, 1999).

According to the FAO classification (FAO, 1998) soils are humic and/or plithic Gleysols and (endo-) gleyic Cambisols. About 5% is plaggic Anthroposols. These are regionally typical *Plaggenesches* that have historically developed from transferring forest and heath soil sods to fields. They are characterized by a 40 to 60 cm deep dark humic layer and have usually better water storage and exchange properties then the original sandy soils.

Farm structure

The typical farm size is 40 to 80 ha, which often comprise significant areas of woodland and pasture. Farms are typically family businesses with a labour capacity of 1.5 to 3.0 full-time equivalents. The level of education among the farmers is generally high and the majority have completed three years of formal agricultural training and the pertinent exam (*landwirtschaftliche Ausbildung*). Farms are often specialised in livestock breeding and grow spinach for diversification of income sources. A small group of highly specialised growers also produce other vegetables and fine herbs for Iglo. Contract growers usually crop one fourth of their irrigated arable land to spinach. The average annual contract area per grower is seven to eleven hectares. The contribution of the spinach growing to the farm income is often significant.

Contract growing system

Iglo contracts fields from individual growers on a yearly basis. Uniform contract conditions and prices are negotiated a priory between Iglo and the growers' board. The growers' board are elected representatives of the roughly one hundred growers, which are organised in the *Verein zur Förderung des Gemüsebaus Westmünsterland, e.V.* Although contracts are made for one year only, there are long-standing relations with most growers and many farms produce spinach for Iglo in the second generation.

Iglo employs three full time fieldsmen for their spinach operation who work together closely with the growers and are in charge of production planning, fertilisation, pest management and harvest organisation. Iglo also selects varieties and buys the seeds.

Growers are responsible for soil preparation and sowing and irrigation management. They carry out fertilisation and pest management measures as mandated by the fieldsmen. A quality management system ensures that fertilisation and pest management measures are only carried out according to fieldsmen's advice. The quality management system also comprises a routine screening for residues of a broad range of pesticide and other contaminants before clearance for a field to be harvested is given.

Harvest and transportation of the spinach raw material are contracted to a specialised third party company who also own the harvesting and field transportation equipment. The relation to the contract harvester is also long-standing.

The standard spinach production scheme

Spinach is cropped in a one-in-four years rotation and grown twice on the same field in the contract year. The first crop is planted between April and June. A the second crop then follows between May and June. Occasionally, a third spinach crop is planted on some fields to compensate unforeseen crop losses during the first two crops. A winter cover crop is usually planted after spinach.

Fertilisation is carried out after soil testing, according to soil mineral nitrogen levels and crop demand. Pest control follows an integrated pest management (IPM) scheme. A nitrogen base dressing is applied as liquid sprayed fertiliser (urea ammonium-nitrate solution), top dressings are applied as calcium ammonium-nitrate. Usually, no phosphate is applied, because soil contents of phosphorus are sufficient as a consequence of intensive manuring. Potassium, magnesium and micronutrients are applied in relation to the measured soil contents and crop demand. Lime application rates depend on the measured soil pH and the soil type.

Herbicides are usually applied in repeated split applications to increase efficacy. Insecticides, if necessary, are selected to minimise environmental impacts. Fungicides are not

Box 1.1 Data assumed for the standard spinach production scheme.

Tillage
- 1st crop
  - Cultivator
  - Plough
  - Sow (drill + rototill)
  - Cultivator
- 2nd crop
  - Plough
  - Sow (drill + rototill)
  - Cultivator
  - Sow cover crop (drill + rototill *or* broadcast)

Fertilisation (kg ha^{-1} yr^{-1})

Liming (CaO) after soil testing	350 [a]
Micronutrients and Mg after soil testing	
1st crop	
N base dressing (UAN)*	72
N top dressing after soil testing (CAN)*	47 [a]
P	0
K	134 [a]
2nd crop	
N base dressing (UAN)*	36
N top dressing after soil testing (CAN)*	47 [a]
P	0
K	111 [a]

Pest management (% of annual allowed rate applied)

Herbicides	Phenmedipham (l AI ha^{-1} yr^{-1})*	0.703 (75%)
	Quizalofop-p-ethyl (l AI ha^{-1} yr^{-1})*	0.023 (37%)
Insecticides	Cypermetrine (kg AI ha^{-1} yr^{-1})*	0.029 (75%)
	Pirimicarb (kg AI ha^{-1} yr^{-1})*	0.015 (3%)

Yield

42 t ha^{-1} yr^{-1} (6% dry matter)

Model rotations

1. Spinach – Cereals [b] – Maize (grain) – Maize (silage)
2. Spinach – Fine herbs [d] – Sugar beets *or* potatoes [c] – Maize (silage)
3. Spinach – Fine herbs [d] – Maize (grain) – Maize (silage)

Standard rotation: weighted mean of 80% Rot 1, 10% Rot 2 and 10% Rot 3.

* UAN = Urea ammonim nitrate solution, CAN = calcium annomiun nitrate, AI = active ingredient.

[a] Applied after soil testing. Figure is 3-year-average rate from Iglo database.

[b] Ceraels are winter wheat or winter barley, 50% each.

[c] 50% each.

[d] Parsley.

applied. Downy mildew-resistant varieties are planted during periods of high disease pressure.

In this study, we assumed a standard growing scheme as shown in Box 1.1. Spinach is mainly grown in maize-maize-cereal rotations on farms with high livestock intensity. Some farms are specialised on fine herbs and other vegetables, a third farm type combined intensive livestock keeping with spinach and fine herbs production. To represent these three farm types, three different model rotations were assumed. The 'standard rotation' is a frequency weighted mean of these three rotations, as shown in Box 1.1.

Iglo's Lead Agricultural Programme

This study is part of the Sustainable Agriculture Project of Iglo, one of Unilever's ten Lead Agricultural Projects. The Iglo LAP follows the general structure of the Unilever Sustainable Agriculture Initiative in looking at (1) *Soil Fertility*, (2) *Soil Loss*, (3) *Nutrients*, (4) *Pest Management*, (5) *Biodiversity*, (6) *Product Value*, (7) *Energy*, (8) *Water*, (9) *Social and Human Capital* and (10) *Local Economy*. Note that in the Unilever terminology these broader areas of concern areas are referred to as 'Indicators', whereas in this study we use the term *indicator* to denote a measurable figure that quantifies a certain phenomenon, as most common in the wider sustainability discourse (cf. Chapter 2).

This study is concerned with quantifying and evaluating adverse environmental effects of agricultural production but a broad range of other activities and projects were launched under the Iglo LAP. Table 1.1 shows how this study fits into this overall programme and in order to draw a complete picture we shall briefly sketch its other activities and projects.

(1) Developing alternative pest management strategies with the University of Hannover (www.gartenbau.uni-hannover.de/ipp/ipp01.htm) and pheromone and pesticide producer Trifolio (www.trifolio.de). This PhD project (started 2001, ongoing) tests a range of alternative controls for insect pests, especially the silver-y-moth (*Autographa gamma*) and explores the potential for a monitoring and forecasting system.

(2) Developing and implementing a GIS based field database with CCgis (www.ccgis.de), combine harvester producer Claas (www.claas.com) and contract harvester Weddeling (www.weddeling.de). The system is presently used in production planning and to support harvesting logistics and also creates opportunities for a range of environmental applications.

Table 1.1 This study and other activities within the Iglo Lead Agricultural Programme (LAP) in Borken. Headings (shaded in grey) are the ten areas of concern ('Indicators') addressed by the Unilever Sustainable Agriculture Initiative. Issues in the left hand column were quantified through indicators and evaluated as described in Chapters 3 and 5 (footnotes for exceptions). In the right hand column, numbers in brackets refer to the description of activities in the text.

Issues addressed in this study	Other activities and projects under the Iglo LAP
Soil Fertility	
▪ Loss of soil organic matter ▪ Soil contamination (heavy metals) ▪ Damage to soil structure (compaction) ▪ Soil acidification ▪ Build-up of pathogen potential	▪ Qualitative soil structure assessment [3] ▪ Experimental programme on reduced tillage [9] ▪ New transportation system and tyres [8] ▪ Grower awareness seminars and training [13]
Soil Loss	
▪ Water erosion ▪ Wind erosion *	▪ Winter cover crops [4]
Nutrients	
▪ Consumption of mineral K resources ▪ Terrestrial eutrophication ▪ Marine eutrophication	▪ Experimental programme on reduced tillage [9] ▪ Winter cover crops [4] ▪ Grower seminars [13]
Pest Management	
▪ Ecotoxicity of pesticides ▪ Human toxicity of pesticides * ▪ Formation of pesticide resistances *	▪ Alternative pest management [1] ▪ Existing Iglo pesticide trial programme
Biodiversity	
▪ Land occupancy ▪ Eutrophication ▪ Acidification ▪ Habitat fragmentation *	▪ Development of biodiversity audit [5] ▪ Margin strip programme [6] ▪ Grower seminars on margin strips [13]
Product Value	
▪ Yield †	▪ Alternative pest management [1] ▪ Quality management system [12]
Energy	
▪ Consumption of fossil fuel ▪ Greenhouse gas emissions ▪ Summer smog/tropospheric ozone ▪ Stratospheric ozone depletion ▪ Generation of solid waste †	▪ Seminar on alternative energy sources [13]
Water	
▪ Water consumption ▪ Marine eutrophication ▪ Ecotoxicity of pesticides	▪ Irrigation management [10] ▪ Margin strip programme [6] ▪ Winter cover crops [4] ▪ Grower seminars [13]
Human & Social Capital	
	▪ Soft systems analysis [7] ▪ *Iglo-Land* programme [11] ▪ Workshops with local stakeholders & growers [13]
Local Economy	
▪ Farm-level production costs † ▪ Labour demand †	▪ Soft systems analysis [7]

* Not quantified and evaluated due to missing data or methodology.
† Quantified but not evaluated as described in Chapter 5.

(3) Qualitative soil structure analysis with Institute for Soil Conservation and Sustainable Agriculture (www.gesunde-erde.net). The aim of this programme was to (1) establish a baseline assessment of the soil structure of spinach fields, using Qualitative Soil Structure Analysis (Beste, 2002) and (2) to train growers in the assessment method. 100 fields were assessed in two years and five grower groups trained.

(4) Experiments on winter cover crops with the University of Hannover (www.gartenbau.uni-hannover.de/gem/) and the horticultural extension service of the Chamber of Agriculture Northrhine-Westphalia (www.gartenbauzentrum.de). The two-year experimental programme encompasses tests with different varieties and sowing dates. A statistical model for the N uptake by winter cover crops following spinach was developed.

(5) Developing a farm biodiversity audit with local nature conservation institution Biologische Station Zwillbrock (www.bszwillbrock.de/html/lowres/index.html) (started 2002, ongoing). Extensive inventories of flora and fauna species were conducted on two pilot farms. These were compared against a ‘basket’ of species that would typically be expected to occur in the local landscape. Based on this comparison, farm specific biodiversity action plans were developed, which contain a set of voluntary measures to conserve, develop or newly establish valuable landscape elements and habitats. Presently this approach is being modified to a simplified farm biodiversity audit.

(6) Field margin programme. The programme encompassed extensive field trials with a range of different vegetation strip types. Strips have in a dual purpose: (1) They are meant to enhance biodiversity by providing food and habitat to the wild flora and fauna. (2) They constitute a ‘technical buffer’ when located between field and watercourses. Since 2001, a mandatory buffer strip between fields and water bodies is stipulated in the Iglo contracts.

(7) Social and economic systems analysis with the horticultural extension service of the Chamber of Agriculture Northrhine-Westphalia (www.gartenbauzentrum.de). The software tool ‘SensitivitätsModell’ (Vester, 1999) is used to clarify the economic and social importance of spinach cropping for the region. The modelling activity involves local government, growers, Iglo employees, local suppliers and other local stakeholders (started 2003, ongoing).

(8) Developing a new transportation system to minimise soil compaction (in co-operation with third party contract harvester Weddeling). The new transportation system separates field and road transport of harvested spinach. This allows for using special trailers on the field. All harvesting equipment now uses tyres with inflation pressures of 100 kPa or below.

(9) Experimental programme on reduced tillage and whole rotation management. This co-operation with the University of Applied Sciences Süd-Westfalen (www.fh-soest.de/fb9/index.html) and five experimental farms explores the potential for introducing reduced tillage systems with the aim of enhancing soil structure, accumulating SOM and N and reducing leaching and run-off. The objective is to develop a whole rotation SOM and N management system to minimise adverse impacts on the soil and the environment (started 2003, ongoing).

(10) Experiments on irrigation management with the University of Hannover (www.gartenbau.uni-hannover.de/gem/) and the horticultural extension service of the Chamber of Agriculture Northrhine-Westphalia (www.gartenbauzentrum.de). This project was meant to investigate the need for a model based irrigation management programme. It involved field trials on four pilot farms and greenhouse experiments. Results showed that expert decisions on irrigation scheduling were equal or even superior to an evapotranspiration based model.

(11) 'Iglo-Land' project with the Agency for Promoting Sustainable Agriculture FNL (www.fnl.de), the rural women's association *Landfrauenverband* and the local tourism board *Münsterland Touristik Zentrale* (www.muensterland-tourismus.de/). This programme started in 2003 and offers factory and farm visits to consumers. It provides background information on spinach cropping and processing and is intended to allow consumers to reconnect with agriculture and the origin of their food. The programme has been fully booked since the beginning.

(12) Extension of an existing ISO Management System. This project (ongoing) with the Chamber of Agriculture Northrhine-Westphalia and a quality management consultant aims to integrate growers into an existing ISO 9001/2000 Quality Management System (ISO, 2003).

(13) Regular pilot grower group meetings, training seminars and workshops were conducted. Monthly meetings and field visits with the five pilot growers were held during the growing season in the years 2000 to 2002. Diverse topics and new practices were discussed. Local stakeholder and grower workshops are held annually. These are meant to inform stakeholders on the current state of the project, discuss proceedings gather feedback on the project. Also, training seminars and discussion groups, e.g. on soil structure, alternative energy source or margin strips, were held on various occasions.

Finally, this study was conducted in co-operation with the University of Hannover (www.gartenbau.uni-hannover.de/gem/) and the horticultural extension service of the Chamber of Agriculture Northrhine-Westphalia (www.gartenbauzentrum.de).

CHAPTER 2

Indicator-based assessment of sustainable agriculture

Assessing sustainable agriculture: a brief overview

Conceptual roots of sustainable agriculture

Perceptions and definitions of sustainable agriculture differ greatly. Today, sustainable agriculture is mostly understood as the agricultural element of sustainable development, which again is most frequently described in the words of the 'Brundtland Commission' (WCED, 1987): "Sustainable development is development that meets the needs of the present without compromising the ability of future generations meet their own needs" (pages 8 and 43).

This 'definition' leaves much room for interpretation – we shall discuss this in greater detail in Chapter 3. For a more profound understanding of the concept of sustainable agriculture, it is useful to recall its multi-rooted origin (for in-depth reviews see Becker, 1997; Christen, 1999). A number of very different conceptual roots have left their marks during the emergence of the sustainability concept:

Management of renewable biotic resources: Forestry has contributed terminology (both in English and in German language) as well as concepts (Becker, 1997). It shares with fisheries the concern for balancing harvest rates with stock-growth rates. Foresters have dealt with this problem systematically at least since the 18th century (Wiersum, 1995) and the quest for identifying 'maximum sustainable yields' (Ricker, 1975), especially in fisheries, attracts continued scientific attention (Walters, 1986; Ludwig et al., 1993; Rosenberg et al., 1993). Forestry and fisheries focus on the management of a single renewable biotic resource that is identical with the stock it is taken from. Agriculture deals with the simultaneous management of diverse resources, biotic and abiotic, and multiple goals. In spite of these quite profound differences, the idea of balanced harvest and re-growth has been readily adopted by agricultural policy makers and researchers as an idealised model for sustainable agriculture.

Concerns about food security: Thomas Malthus (1798) was probably the first to utter concerns about sufficient food supply in the face of an exponentially growing population and linearly growing agricultural productivity. Although there are indications that world population growth is now dampening (Lutz et al., 2001), hunger and malnourishment persist in the world. The concern about 'feeding the world' has coined the debate on sustainable agriculture from the very beginning (Douglass, 1984).

Emerging environmental sciences: Motivated by the demand placed on science to cope with the challenges of large-scale environmental issues, a range of novel scientific disciplines and sub-disciplines evolved since the 1980's. Among them are ecosystem theory (Müller, 1997), sustainability and ecological economics (Daly, 1996), environmental accounting (Friend, 1996) and others. They have contributed the ideas such as ecosystem health (Costanza et al., 1992) and extended the concept of capital to include natural, human and social capital (Fukuyuma, 1995; Daly, 1996).

Alternative agricultural movements: A range of alternative agriculture movements have left their ideological marks on what is today known as 'sustainable agriculture'. Certain system attributes, such as 'organic', 'conservation' or 'low-input', have often been equated with sustainable agriculture (Lockeretz, 1988). Alternative agricultural movements have provided some of the ideological and ethical concepts that are now deeply entrenched in the discourse on sustainable agriculture (Becker, 1997). Among them are the commitment to environmental stewardship and animal rights as an ethical obligation to the creation, the preference of low-input systems and solutions that mimic and link into natural systems.

Rural development work: Finally, learnings from agricultural and rural development projects as well as from the 'green revolution' have entered and shaped the debate on sustainable agriculture (De Kruijf and Van Vuuren, 1998; Pretty, 1998; Röling and Wagemakers, 1998; Bell and Morse, 1999). The emphasis of participatory structures, indigenous knowledge and traditional management systems, as well as the widespread scepticism to end-of-the-pipe solutions and technological fixes, reflect lessons learned from development work.

At first, these movements and agendas appear to have little in common. However, they are united by the concern that current patterns of production and consumption cannot be upheld or expanded without degrading the natural support systems of human life.

Today it is widely recognised that sustainable agriculture should be multidimensional and embrace agronomic, environmental and ecological, social and economic aspects of agriculture (Zinck and Farshad, 1995). In contrast, older perceptions of sustainable agriculture tend to be single dimensional (e.g. focussing on economy or on soil resources only). Since the late 1990's authors increasingly emphasise the importance of trans-disciplinary and participatory approaches (Röling and Wagemakers, 1998; Bell and Morse, 1999). For more detailed reviews of the emergence of and general concepts in sustainable agriculture we refer to Farshad and Zinck (1993), Christen (1996, 1999), Hansen (1996), Becker (1997) and Smith and McDonald (1998).

Types of sustainability assessment

Smith and McDonald (1998), amending Hansen (1996), distinguish five different types of sustainability assessment:

- *Adherence to prescribed approaches*, i.e. sustainability is judged depending on whether or not a certain set of rules or practices is followed;
- *Multiple qualitative and quantitative indicators*, i.e. sustainability is judged depending on a number of indicators and their realisations;
- *Time trends*; i.e. sustainability is judged depending on the temporal evolution of a single or multiple indicators;
- *Analysis of resilience and sensitivity*, i.e. sustainability is judged depending on how sensitive a system is to disturbances;
- *System simulation*, i.e. sustainability is judged depending on the behaviour of a model of the system.

Strictly speaking, only the first two are conceptually different, whereas the third to fifth are actually means of evaluating and generating indicators and data, respectively. Von Wirén-Lehr (2001) classified the first two types as 'means-orientated' and 'goals-orientated' approaches, i.e. approaches prescribing a certain set of rules to be followed (such as many organic agriculture schemes) as opposed to approaches prescribing certain environmental, social or economic goals to be met. Since it is difficult to prove whether goals are actually met in the individual case, strict goals-orientated approaches are rare. In practice, they often contain means-orientated elements, such as the Lower Austrian Eco-points (*Ökopunkte*) approach (Mayrhofer, 1997), which links agricultural subsidies to the amount of 'Eco-points' a farm receives for adopting certain practices and levels of production intensity.

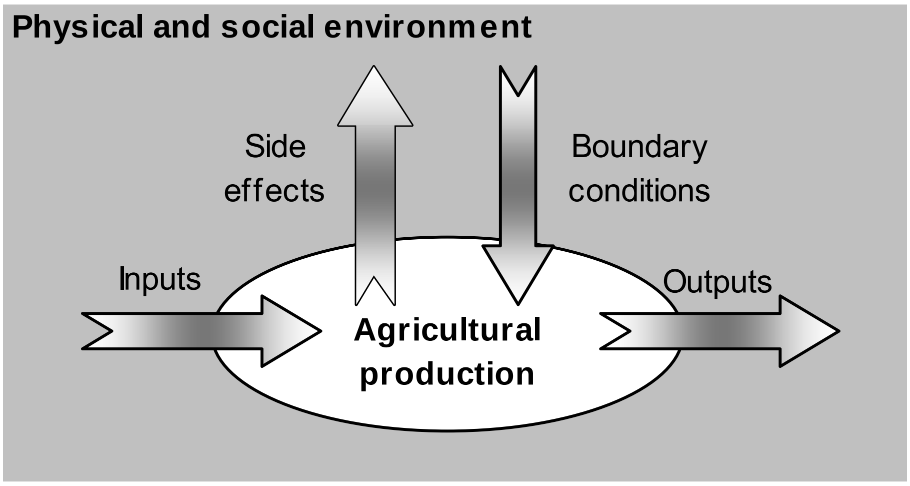

Figure 2.1 Relationship between agriculture and its physical and social environment: Agriculture is embedded in a physical and social environment, which provides *inputs*, consumes the agricultural *outputs*, absorbs undesired *side effects* and at the same time defines its *boundary conditions*. Threats to sustainability can affect all of the four relations.

The scope of sustainable agriculture assessment

Agricultural systems fulfil a number of primary and secondary functions, as provision of food or the maintenance of typical landscapes. At the same time, they are both a source of stress on their social and physical environment and threatened by outside pressures. Borrowing from Cornforth (1999), we may conceptualise these different aspects of agriculture as shown in Figure 2.1, where agriculture is embedded in a physical and social environment, which provides inputs, consumes the agricultural outputs, absorbs undesired side effects and at the same time defines the boundary conditions for agriculture.

Threats to sustainability can affect all of these interactions with the environment: The functioning of agricultural systems may depend on scarce or dwindling input factors such as phosphate fertilisers or well-trained farmers; agricultural outputs may not comply with expectations placed on them (e.g. not produced in a socially accepted way); agriculture may exert pressures on other systems (e.g. water consumption, emissions); and is itself subject to environmental pressures, such as pest invasions or international trade regulations. Note that many issues affect more than one of the four above interrelations, e.g. pesticide resistance or soil salinisation can both be an effect of and a threat to agriculture.

Most assessment schemes for sustainable agriculture focus on side-effects and outputs (arrows pointing *out* of the agricultural system in Figure 2.1). Typically, these are operationalised with impact and performance indicators. Environmental constraints and limitations to inputs (arrows pointing *into* the agricultural system) are less frequently addressed, presumably because they are less manageable and often difficult to operationalise. ‘Soft systems analysis’ approaches, e.g. as proposed by Bell and Morse (1999) or Bossel (1999), are especially suitable for assessment of these two types of interactions.

Analytical dimensions of sustainability assessment

In the literature it is common to distinguish three analytical dimensions of (or perspectives on) sustainability: an ecological one, an economic one and a social one. In the context of sustainable agriculture some authors narrow this down to ‘ecological soundness’, ‘economic viability’ and ‘social acceptability’ (e.g. Smith and McDonald, 1998).

Although common, Conrad (1999) argues that this trilogy is analytically rather inconsistent: There is a physical and a social reality of human behaviour and products (where ‘physical’ and ‘social’ are used in a wider sense, encompassing biological and geological phenomena or economic ones, respectively). Conrad thus proposes to distinguish only two dimensions: a physical and a social one. A similar classification has been used by Cornforth (1999) and is also consistent with the core concerns of sustainability: (1) ensure development (the social perspective) while (2) acknowledging the limited capacities of nature (the physical perspective).

Scales of sustainable agriculture assessment

Issues affecting and affected by agriculture span various spatial scales. Spatial scales range from the hectare scale (or even less) to the global scale, i.e. differ by as much as ten orders of magnitude. Temporal scales range from days and weeks to centuries or even longer. Institutional scales range from the individual to societies and economic scales from the business and household level to the world market.

Dumanski et al. (1998) note that there is a general gap between agronomic scales and policy-making scales. Causes and effects of the same problem often lie at different scale levels. E.g. nitrous oxide (N_2O) is emitted at field level but affects the climate system at the global scale (Mosier et al., 1996). Field emissions of N_2O negligible in agronomic terms (about 1% of the field applied nitrogen is lost as N_2O; Bouwman et al., 2002) but contribute roughly 40% of the annual N_2O global flux to the atmosphere (IFA/FAO, 2001; Mosier et al., 1998).

This example shows that the integration of different spatial scale levels is often crucial to fully appreciate sustainability issues. Although this fact is widely recognised in the literature on sustainable agriculture, only few authors propose explicitly multi-scale assessment schemes (e.g. Dumanski et al., 1998; Niu et al., 1993; and Smith and McDonald, 1998).

Indicators and indicator sets for sustainable agriculture

Indicators

In the literature on sustainable agriculture, conceptions of the term ‘indicator’ are often remarkably vague and heterogeneous. A widespread understanding of an indicator is that of “a measure of something in which one has an interest, but which is difficult to monitor directly” (Rigby et al, 2001; similar Mitchell et al., 1995; Eswaran et al., 1994). Many so-called definitions describe what indicators do, rather than what they are. The following three functions are named most frequently:

- *Diagnostic function*: measure change or inform about status (Eswaran et al., 1994; Smyth and Dumanski, 1995);
- *Facilitating function*: reduce complexity of complicated phenomena and processes and facilitate their communication (Zinck and Farshad, 1995; Merkle and Kaupenjohann, 2000; Bockstaller et al, 1997);
- *Policy performance function*: quantify the effects of policy or management measures (Münchhausen and Nieberg, 1997; Young, 1997; Stevenson and Lee, 2001).

We propose a more technical understanding of indicators: They are simply and first of all correlatives (and thus quite commonplace in science). Any variable that correlates with another variable can be used as an indicator thereof. This second variable is then called ‘indicandum’ (Rinne, 1994; Radermacher et al., 1998). A variable hence becomes an indicator as soon as it is used to describe another variable.

Different taxonomies for indicators have been used in the literature. Classification criteria fall into three groups:

<u>*Functional:*</u> What type of indicandum does the indicator describe? The OECD Driving Forces-Pressure-State-Impacts-Response model (1998) and its derivatives are well-known examples. Lewandowski et al. (1999) distinguish agricultural impact indicators (describing sources of agricultural impacts) from ecosystem effect indicators (describing their effects). This is similar to the source-receptor concept reported by Schäfer et al. (2002).

<u>*Qualitative:*</u> How closely related is the indicator to the indicandum? E.g. Eswaran et al. (1994) distinguish between direct, surrogate/proxy and crypto indicators, measuring the

area of concern (i.e. indicandum) directly (e.g. soil erosion rates), indirectly (number of persons considering a project a success) and very indirectly, respectively. As examples for crypto indicators the authors cite the sudden increase of attendance in temple ceremonies as an indication of woes of Indian villagers or the replacement of cattle by sheep as a consequence of grassland degradation. Similarly, Cornforth (1999) distinguished between direct and indirect indicators.

By normative contents: Does the indicator have prescriptive implications or not? ‘Critical loads exceedance’ is a typical example of a normative indicator, ‘acid deposition’ of a descriptive one.

The distinction between descriptive and normative indicators is important: Many authors argue that sustainability indicators should be normative, because they should allow for a decision whether a particular measurement value is sustainable or not (SRU, 1994; Stockle et al., 1994; Smyth and Dumanski, 1995; Syers et al., 1995; von Münchausen and Nieberg, 1997; Cornforth, 1999; von Wirén-Lehr, 2001).

The indicator – indicandum relationship

The relationship between indicator and indicandum is definitional. It does not exist for any logically or ontologically cogent reasons but is created through assertion (Radermacher, 1998). Thus indicators cannot simply be ‘derived’ or ‘deduced’ by applying some standard routine. Rather, finding indicators involves inventiveness, skill and expertise in the field of the indicandum.

We stated above that any correlative for any variable could be used as an indicator. We shall attempt to characterise this relationship in greater detail by the following features:

Statistical nature of the variables: Both indicandum and indicator can be discrete qualitative (nominal scale), discrete quantitative (ordinal or interval scale) or continuous quantitative (cardinal scale). There are thus nine possible combinations of variable types.

Closeness of correlation: The correlation between indicator and indicandum may be more or less close. Typical statistical measures are e.g. correlation coefficients (continuous variables/cardinal scale), rank correlations (discrete variable/ordinal scale) or contingency tables (discrete variables/nominal scale).

Empirical quality: The quality of data may range from sporadic measurements to statistically valid samples and from textbook figures to specific on-site measurements.

<u>*Theoretical foundation:*</u> The causal relation between indicandum and indicator may range from mere empirical observation to detailed theory and from indigenous knowledge to mechanistic models.

<u>*Credibility and acceptance:*</u> The credit of the hypothesised indicator-indicandum relationship may range from odd theory to established and accepted theory. Within science there may be competing schools which are each capable of explaining certain phenomena but fail to explain others. Also, the credit of a hypothesised indicator-indicandum relationship may depend on the larger context: The hypothesis that the Earth was a globe had, despite empirical evidence, little intellectual traction before the discovery of gravity. Today, homeopathy is increasingly recognised as a successful method in human and veterinary medicine, although up to now no one knows how it works. Finally, there may also be a religious or ideological component to credibility: The debate about the evolutionary theory of the creation of life in the light of creationism in the United States is a recent example.

Above we cited typical selection criteria for indicators named in the literature. The five features just discussed are usually subsumed under the criterion 'scientific soundness'. If 'scientific soundness' is to be quantified and describe in greater detail, measures of the above five features are needed. The *statistical nature of variables* and the *tightness of correlation* can be described in prose and by standard statistical measures, respectively. The *empirical quality*, *theoretical foundation* and *credibility and acceptance* are less straightforward to measure.

They do, however, correspond largely with the criteria for 'data pedigree' introduced by Funtowicz and Ravetz (1990) as detailed by Costanza et al., (1993). Funtowicz and Ravetz are concerned with a more comprehensive description of data uncertainty in re-

Table 2.1 Numerical estimate pedigree matrix for qualitative assessment of data uncertainty (adopted from Costanza et al., 1992).

Score	Theoretical, quality of model	Empirical, quality of data	Social, degree of acceptance
4	Established theory Many validation tests Causal mechanisms understood	Experimental data Statistically valid samples Controlled experiments	Total All but cranks
3	Theoretical model Few validation tests Causal mechanisms hypothesized	Historical/field data Some direct measurements Uncontrolled experiments	High All but rebels
2	Computational mode Engineering approximation Causal mechanisms approximated	Calculated data Indirect measurements Handbook estimates	Medium Competing schools
1	Statistical processing Simple correlations No causal mechanisms	Educated guesses Very indirect approximations "Rule of thumb" estimates	Low Embryonic field
0	Definitions/assertions	Pure guesses	None

lation to the theoretical, technical and social background on which data are gathered and generated. They argue that this information is highly relevant for decision makers but not communicated by standard statistical measures (such as variance or confidence limit). Costanza et al., (1992) propose a semi-quantitative assessment for the theoretical, empirical and social quality of data (Table 2.1). It is suitable for describing the above features of indicator-indicandum relationships as well: *empirical quality*, *theoretical foundation* and *credibility and acceptance*. As proposed by Costanza et al., a single score for an individual indicator-indicandum pair can be obtained by adding the scores for theoretical, empirical and social quality. Dividing the sum by twelve (the maximum total score) then normalises the result conveniently to an interval from zero to one, with one indicating the highest indicator quality.

Indicator sets

As an indicator set we define any one consistent collection of indicators that together describe all relevant aspects of an agricultural (or other) system. Although the literature about sustainable agriculture and indicators for sustainable agriculture is abundant, there are few specific publications on indicator sets. Mitchell et al. (1995) are concerned with having an adequate representation of indicators for sociological concerns, ecological issues and uncertainty in a set. Similarly, Stockle et al. (1994), Smyth and Dumanski (1995), Bossel (1999) Becker (1998) Bell and Morse (1999) and Cornforth (1999) list certain areas of concern that are to be covered by an indicator set, arguing from the perspective of systems analysis, practical project experience or the broader scientific or societal discourse, respectively. Radermacher at al. (1998) and Schäfer et al (2000) explicitly address indicator set construction from a sociological perspective as a socio-technical production process.

Review of literature on sustainable agriculture indicators

In contrast to literature on indicator sets, publications on indicators in the context of sustainable agriculture are abundant. Two major strands may be distinguished: (1) literature referring to technical and methodological aspects and (2) literature reporting on specific indicator sets and their practical application. Von Wirén-Lehr (2001) provides a systematic review of the latter category, which we extended as shown in Table 2.2.

Here we give a brief overview of the first strand of literature (technical and methodological aspects). Zinck and Farshad (1995) elaborate on definition, assessment and implementation of sustainability with regards to indicators. Smith and McDonald (1998) review definitions, concepts, and approaches to sustainable agriculture within the broader context of sustainable development. They propose to use ‘threats to sustain-

Table 2.2 Indicator sets for sustainable agriculture (after von Wirén-Lehr, 2001, amended).

Reference	Scope	Production system	Case study area	Target group	Spatial level	Definition of indicanda and indicators	Source of reference values
Biewinga and van der Bijl (1997)[a]	ecol	Energy crops	Europe	Policymakers	Region	Expert interviews	Not normalised
Bockstaller&Giraradin (2000a, b)	ecol	Crops	France	Farmers	Field/farm	Authors' appraisal	Authors' appraisal
de Koning et al. (1997)[a]	ecol	Crops	Ecuador	Policymakers	'Unit' (9 km x 9 km)	Authors' appraisal	Authors' appraisal
Eckert et al. (2000)	ecol	Crops	Germany	Farmers, policymakers	Field/farm	Expert interviews and authors' appraisal	Expert interviews
Halberg (1999)	ecol	Meat and milk	Denmark	Farmers	Field/farm	Author's appraisal	Author's appraisal
Isermann and Isermann (1997)[a]	ecol, econ, soc	Crops	Germany	Policymakers	Nation	Authors' appraisal	Existing legislative thresholds
Kessler (1994)[a]	ecol	Crops	Semi-arid regions	Policymakers	Region	Expert interviews and author's appraisal	Expert interviews and author's appraisal
Lefroy et al. (2000)	ecol, econ, soc	Crops	Vietnam, Nepal, Indonesia	Farmers, extension workers	Field/farm	Symth and Dumanski (1995), target group	Authors' appraisal
Plachter and Werner (1998)[a]	ecol, econ	Crops	Germany	Policymakers and farmers	Region	Expert interviews and stakeholders	Existing legislative thresholds
Reganold et al. (2000)	ecol, econ	Apples	Washington State, USA	Not specified	Field/farm	Authors' appraisal	Not normalised
Rossing et al. (1997)[a]	ecol, econ, soc	Crops	France, The Netherlands	Farmers	Field/farm	Expert interviews and stakeholders	Expert interviews and stakeholders
Sands and Podmore (2000)	ecol	Wheat, maize	Colorado, USA	Not specified	Field/farm	Authors' appraisal	Authors' appraisal
Taylor et al. (1993)	ecol	Cabbage	Malaysia	Farmers	Field/farm	Authors' appraisal	Authors' appraisal
Van Mansvelt and van der Lubbe (1999)	ecol, econ, soc	Crops	Europe	Farmers, policymakers	Field to landscape	Expert interviews, authors' appraisal, and target group	not normalised; suggestions based on authors' appraisal

[a] As cited by von Wirén-Lehr (2000).

ability' at various spatial scales as indicanda to evaluate sustainability of agricultural systems during the planning stage. Stockle et al. (1994) follow a similar strategy. They propose an assessment of factors that constrain the sustainability of agricultural production systems.

Becker (1998) uses a qualitative assessment matrix for quick appraisal of the sustainability of land systems. It is based on a conceptual model of sustainable development developed by the author. Lewandowski et al. (1999) present a framework for identifying indicanda and indicators on the basis of impacts pathways, linking emissions from agriculture to affected ecosystems. Related to this framework, Merkle and Kaupenjohann (2000) propose to identify 'ecosystem effect indicators' for agricultural emissions by intersecting a top-down approach (societal goals broken down to ecosystem or field scale) and a bottom-up approach (determine important factors by detailed analysis of processes on the field scale).

Syers et al. (1995) review the development and use of indicators and highlight some aspects of indicator assessment over time and the use of indirect indicators. Smyth and Dumanski (1995) report on the 'Framework for Evaluation of Sustainable Land Management' (FESLM) based on 'five pillars' of sustainable land management as proposed in the FAO *Guidelines for Land Evaluation* (as cited in Smyth and Dumanski, 1995). They suggest a 'logical pathway analysis' to identify site-specific indicators. Eswaran et al. (1994) provide some basic considerations about and concepts of indicators and indicator types. Cornforth (1999) focuses on the relationship between indicators, management and processes within the context of the FESLM.

Recent work emphasizes participatory elements within the research process. Stevenson and Lee (2001) present a conceptual framework to combine policy requirements and participatory elements with 'objective' science in indicator development. Bell and Morse (1999) highlight the integration of participatory processes and 'soft systems analysis' within a project setting for the definition of goals and indicators and propose a procedure for their assessment over time.

The three stages of indicator-based sustainability assessment

We propose that sustainability assessment on the basis of indicator sets may be conceptualised as consisting of three stages (also cf. von Wirèn-Lehr, 2001):

1. *Indicator set construction*, where indicanda are identified and appropriate indicators are assigned;
2. *Indicator set evaluation*, where indicators are interpreted with regards to sustainability; and

3. *Strategy development*, where strategies for improvement are developed and monitored.

We describe these three stages in more detail below. Figure 2.2 represents these stages graphically. Note that the temporal order of these stages is not always cogent, as the process may contain iterative elements (cf. Radermacher et al., 1998).

Stage 1: Indicator set construction

Indicator set construction comprises two steps: *Identification of indicanda* and *Assignment of indicators*.

Identification of indicanda

In the literature, indicanda are often set according to the authors' appraisal (see Table 2.2). Typical examples are the works of Eckert et al. (2000), Sands and Podmore (2000) and Reganold et al. (2001). The selection of indicanda there is either axiomatic (Sands and Podmore), claimed to be drawn from the general political discourse (Eckert et al.) or not justified at all (Reganold et al.).

Recently, there is a broadening consensus that indicanda should be identified in participatory processes (Röling and Wagemakers, 1998; Bell and Morse, 1999; Stevenson and Lee, 2001), either among experts on the national level (as in van Mansvelt and van der Lubbe, 1999 and Münchhausen and Nieberg, 1997); among local experts (Lefroy et al., 2000); or local stakeholders (Bosshard, 1997; Young, 1997).

A general formal problem with both approaches to identifying indicanda – author appraisal and participation – is a lack of transparency and reproducibility. It is difficult to understand, why certain issues are assumed to be relevant and other are not. Dunlap et al. (1992) found that the sustainability perception of diverse social groups involved with agriculture varies significantly. As every group of scientists and every project team have their own selection of themes, the exercise of identifying indicanda is somewhat arbitrary. In fact, Anderson and Lockeretz (1992) found that there is often a lack of consistency between the general sustainability perception proclaimed by the authors of a study and the criteria and indicators actually chosen.

Another problem with author appraisal and participation is that of undetected personal biases. Every expert or member of a project team also pursues individual or institutional agendas (Lélé and Norgaard, 1996; Fixdal, 1997) and the selection of indicanda will be more or less influenced by these agendas. The main problem here is not the fact that there *are* biases (because they are to some degree inevitable), but that it remains unclear *where* and to *what degree* they influence the results. Having a rigorous documentation

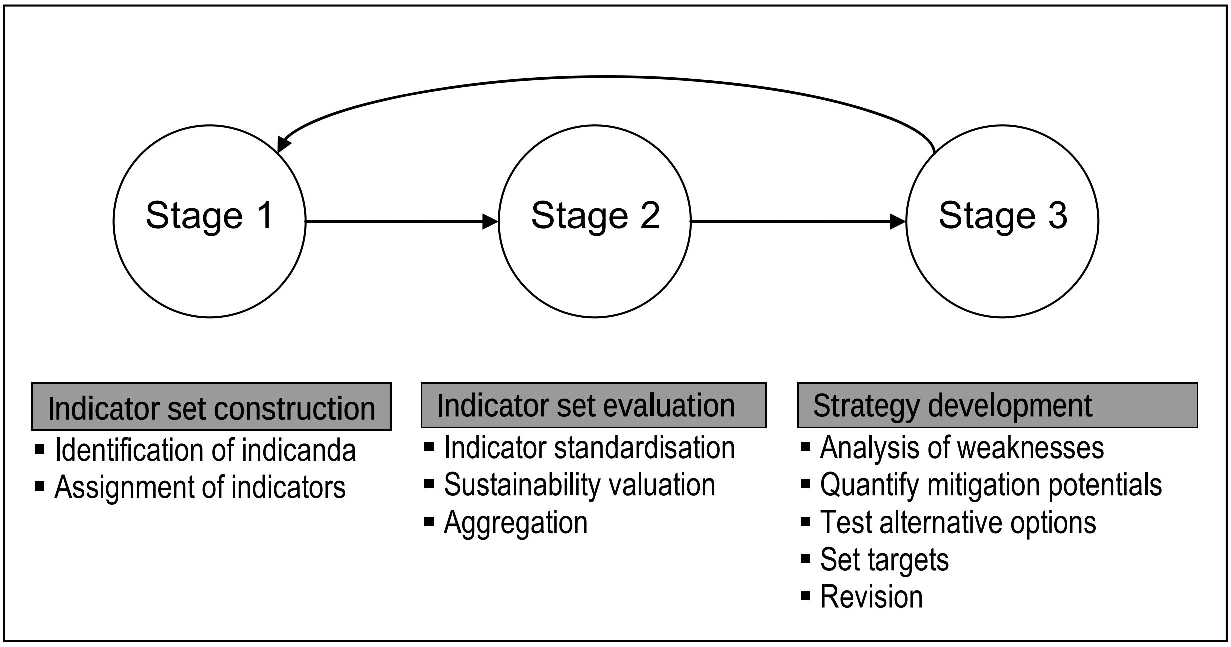

Figure 2.2 Stages of indictor-based sustainability assessment.

protocol in place (e.g. Tacconi, 1998) could in part alleviate these shortcomings, yet causes an immense amount of background paper work.

A specific problem with participatory identification of indicanda is the danger of settling for 'negotiated nonsense': A local group might 'negotiate' their very own conception of sustainability, which might be detached from the wider sustainability discourse or the understanding of the public.

Assignment of indicators

Assignment of indicators is often an informal process based on authors' or experts' backgrounds and opinions. In fact, indicator assignment involves a strong element of skill and creativity. The quality of an indicator depends on how well it describes the underlying indicandum as well as on its intended purposes (see above).

In the literature, different sets of criteria of indicator quality have been proposed (Eswaran et al., 1994; Mitchell et al., 1995; Smyth and Dumanski, 1995; Syers et al., 1995; Zinck and Farshad, 1995; Harger and Meyer, 1996; Cornforth, 1999; Stevenson and Lee, 2001). The most frequently named are 'scientific and analytical soundness', 'measurability against a threshold value', 'sensitivity to change' and 'cost efficiency'. These criteria can be used to select the best indicators once a pool of potential ones has been identified. In practice, these criteria seem to be hardly ever used.

The assignment of indicators is often, as the identification of indicanda, addressed in a participatory way (e.g. Bell and Morse, 1999) and prone to the same methodological difficulties, i.e. lack of transparency and reproducibility, lacking control of personal biases and the potential to produce ‘negotiated’ solutions.

Mitchell et al. (1995) propose a very detailed method for constructing sustainability indicators. It is mainly meant to ensure that indicators relate to measurements of quality of life and ecological integrity. They do, however, not detail who and by what legitimacy is supposed to identify indicanda and assign indicators.

In the literature, identification of indicanda and assignment of indicators are often treated as a single step. In fact, both steps are not entirely independent, because the perception of issues of concern is influenced by existing measures and concepts and vice versa (Radermacher, 1998).

Separating the two steps conceptually is still useful, because this helps being clear about the issues (i.e. the indicanda) and the required indicators to quantify them. Otherwise, indicators are sometimes perceived as an end of their own, which easily results in irresolvable debates about ‘the best’ indicators and favours the selection of ‘friendly indicators’ (i.e. measuring indicators with a positive trend and neglecting the actual issues).

In fact, the process of indicator assignment is difficult to formalise, because it involves creativity and skill and there are no logically or ontologically cogent criteria calling for a certain indicator to describe a particular indicandum. We here propose using the ‘pedigree’ assessment matrix by Costanza et al. (1993) cited above (Table 2.1) for a comprehensive description of indicator quality. If several indicators are available for the same indicandum, the pedigree assessment can be used as a decision criterion.

Stage 2: indicator set evaluation

Indicator set evaluation may comprise three (facultative) steps: *Indicator standardisation*, *Sustainability valuation* and further refinement, such as *Aggregation*.

Indicator standardisation

The objective of standardisation is to make different indicators comparable. Formally speaking, it puts the value of different indicators through a scaling function to map them onto a common interval or set of values (S):

$$f : i \mapsto s(i) \quad \text{with } s \in S = \{a, \ldots, b\} \text{ and } a < b \tag{2.1}$$

The scaling function can either be the same for every indicator, where only parameter values differ between individual indicators. Or it can differ for each indicator. Giegrich (1997) distinguished three fundamental patterns of ‘evaluation logic’, i.e. formal rules or procedures by which to standardise different indicators

- *The utility analytical approach*, where each indicator is assigned (a) a weight and (b) a target. The degree of target compliance for each indicator is determined and weighted by the indicator's weight. Normative charge enters through the choice of targets and the appraisal of target compliance. Indicator sets standardised by this approach are e.g. Eckert et al. (2000), Bockstaller et al. (1997) and Sands and Podmore (2000);
- *The damage/benefit approach*, where each indicator is converted and expressed in terms of a 'currency' unit, e.g. energy, toxicity equivalents or money. Normative charge enters through the assignment of a utility or damage value to the indicator values. Examples are Wackernagel and Yount (1998), Tellarini and Caporali (2000); and
- *The critical level approach*, where a critical level (or threshold) is defined for each indicator. Indicators are then normalised by division through this critical level. Normative charge enters through the choice of critical levels. Examples are Life Cycle Assessment methodologies (Brentrup, 2003) or the pesticide risk potentials developed by Burth et al. (2002).

The more parameters a scaling function has, the higher will normally be its normative contents, because parameter choice always implies value judgements or assumption. Schäfer et al. (2002) distinguish between different modes of scaling functions and parameter selection: They may be based on (1) negotiation among stakeholders (as e.g. proposed by Young, 1997); (2) expert elicitation (as e.g. the evaluation functions used by Eckert et al., 2000); (3) science, e.g. as acceptable daily intake of toxins (FAO/WHO, 1975; WHO, 1999) or critical loads for ecosystem eutrophication (Posch et al., 1995); and (4) data inherent, such as minimum and maximum of a data set or the mean plus/minus one standard deviation (as e.g. in Burth et al., 2002).

The set of values S onto which scaled indicator values are mapped can either be unconfined ($a = -\infty$ or $b = +\infty$ or both in Equation 2.1) or an interval ($-\infty < a < b < +\infty$). An example of the former is the normalisation step in Life Cycle Assessment where each indicator value is divided by a reference value (Brentrup, 2003). The latter is most prolific in the literature (e.g. Eckert et al., 2000) but has the disadvantage that it requires somewhat artificial cut-off points. All indicator values above (or below) the cut-off point are transformed to the same standardised value, which means losing information, e.g. on the distance from a target or the relation between two different values.

Sustainability valuation

Sustainability valuation interprets an indicator value in relation to sustainability and assigns it a normative value. Depending on the sustainability perception of the authors, this could be either a discrete classification (e.g. sustainable vs. unsustainable) or a continuous 'degree of sustainability' (e.g. Niu et al., 1993; Hansen, 1996). Smyth and Dumanski (1995) propose a discrete classification, based on the time that a land use system is believed to stay stable.

The above-cited types of parameter sources by Schäfer et al. (2002) also apply to the selection of threshold values and parameters of the sustainability valuation function.

These two steps under the evaluation stage – standardisation and sustainability valuation – are facultative. Many conceptual frameworks and indicator sets presented in the literature do not include any evaluation step at all (see also Table 2.3). Also, indicator standardisation and sustainability valuation are often merged into one single step. In fact, we have not found any one example in the literature where they are conceptually separated. This is interesting, because they are completely independent, both technically and logically: Indicators could be standardised without a sustainability valuation and valuated without being standardised.

Aggregation

Finally, sustainability evaluation may comprise rules for aggregation of single indicators to form higher-level indices, which would be a third (and also facultative) step. In order to provide a consistent basis for aggregation, indicators need to be made comparable, i.e. standardised.

Typical evaluation rules and procedures would simply add standardised indicators (Gierich, 1997). Practical examples are Young (1999) and Sands and Podmore (2000). If not yet included in the standardisation function, individual indicators could also be weighted with a weighting factor that describes their relative importance (Gierich, 1997; Andreoli and Tellarini, 2000):

$$\mathrm{Idx} = \sum_i \left(\mathrm{w}_i \times \mathrm{I}_i\right), \tag{2.2}$$

where *Idx* is the aggregated index and I_i are the individual indicators and w_i the pertinent weighting factors.

Rigby et al. (2001) and Taylor et al. (1993) use a simple scoring systems, which assigns values of e.g. zero, 0.5, 1 and 3 to a number of (supposedly) positive and negative agricultural practices. Scores are multiplied by the proportion of land/crops managed with a certain practice (e.g. green manuring) and then added to produce a total score (or index) for the farm. More complex or non-linear aggregation schemes are, although defendable, not found in the literature, probably because they require high normative input, which is often not available within the confines of a research project.

Stage 3: Strategy development

The third stage of indicator based sustainability assessment is the least easy to formalise. Strategy development will very much depend on the political and social context of the indicator set construction exercise, its purpose, environmental, economic and social boundary conditions and on the skills and intuition of those involved. Nevertheless, some elements may be considered generic:

Analysis of weaknesses: Identifying areas where the system under investigation performs well and those where there is room for improvement is the basis for all further development.

Quantify mitigation potentials: A subsequent step would be concerned with quantifying the room for improvement under different boundary conditions. Depending on the purpose of the indicator set construction exercise there could also be advice on alternative and improved management options.

Test alternative options: Often there will be an interest in developing and evaluating different scenarios (Sands and Podmore, 2000) or in testing different production systems (Reganold et al., 2001) against a standard or against one another.

Set targets: As the practical outcome there should be an action plan for improvement, including realistic targets and by what means to reach them.

Revision: Periodically, it should be monitored whether goals have been reached and what potential obstacles were. Also, the entire process should be iterated periodically to revise the indicator set itself.

Table 2.3 reviews the previously cited references against the three stages of indicator based sustainability assessment identified here. As it shows, none of the 22 publications addresses all three stages (or at least the first two) in a systematic manner. Most publications focus on evaluation techniques (Stage 2), which is also the most technical one of the three steps. Stage 1, indicator set construction, is only addressed in detail by Mitchell et al. (1995) and by Bell and Morse (1999), who take, as other authors, an participatory approach to indicator set construction.

Most references are also poorly documented when it comes to the epistemological and methodological foundations of the proposed methods. This leads us to a final issue, documentation.

Documentation

Having a documentation protocol in place will greatly enhance the transparency of any indicator based sustainability assessment. Specifically, it should inform on

Table 2.3 Stages of indicator based sustainability assessment reflected in publications on indicators and indicator set for sustainable agriculture. Stages are: (1) indicator set construction; (2) indicator set evaluation; (3) strategy development (Figure 2.2). 'x': stage addressed comprehensively, '(x)': only touched briefly.

Reference	Type [a]	Assessment stage 1	2	3
Andreoli and Tellarini (2000)	FW		x	
Becker (1998)	FW		(x)	
Bell and Morse (1999)	FW	x[b]	(x)[b]	(x)
Cornforth (1999)	TH			
Eckert et al. (2000)	FW		x	
Eswaran et al. (1994)	TH			
Halberg (1999)	APP			
Lefroy et al. (2000)	APP	(x)[b]	(x)[b]	(x)
Lewandowski et al. (1999)	FW			
Merkle and Kaupenjohann (2000)	TH		(x)	
Mitchell et al. (1995)	FW	x[b]		
Reganold et al. (2001)	APP			
Rigby et al. (2001)	APP		x	
Sands and Podmore (2000)	APP		x	
Smith and McDonald (1998)	FW			
Smyth and Dumanski (1995)	FW	(x)[b]	(x)	
Stevenson and Lee (2001)	FW			
Stockle et al. (1994)	FW			
Syers et al. (1995)	TH			
Taylor et al. (1993)	APP		x	
v. Mansvelt and v. d. Lubbe (1999)	FW	(x)[b]	(x)	
Zinck and Farshad (1995)	TH			

[a] Focus of work: App = applied or case study; FW = conceptual framework; TH = theoretical aspects of indicators and indicator sets.
[b] Addressed through participatory approach.

- *Products:* Specifying what exact methods and data have been used and what the underlying assumptions are;
- *Purpose:* Specifying why and to what exact purpose methods and data were used and what arguments were put forward for and against them;
- *Persons:* Specifying who was involved, by what legitimacy, what their personal backgrounds and agendas are;
- *Process:* Specifying formal structures, procedures and who made decisions.

Funtowicz and Ravetz (1990) call this the 'p-fourth' approach to quality assurance in issue-driven research.

CHAPTER 3

A systematic conception of sustainable agriculture as the basis for scientific inquiry*

Abstract

Definitions of sustainability abound. However, because they are not concrete, these definitions usually prove unsuitable as a basis for scientific inquiry. This chapter presents a way of arriving at workable conceptions of sustainability as the basis for scientific inquiry by systematically drawing on the existing sustainability discourse. We introduce three analytical constructs to guide the analysis of sustainability definitions: (1) issue-driven vs. goal-pursuing approaches, (2) normative vs. descriptive elements in the sustainability discourse and (3) positive vs. negative statements on sustainability.

We propose that firstly, issue-driven perceptions of sustainability are historically more defendable than goal-pursuing perceptions and that the 'driving issues' are often quite straightforward. Secondly, we find that science and scientists play both a normative part as well as a descriptive part in the sustainability debate. In order to ensure transparency and avoid personal biases it is important to be explicit about these two parts. Finally, we find a pronounced mismatch between the timescales we can overlook and those relevant for sustainability. As a consequence, we can make defendable statements on unsustainability but not on sustainability.

Based on these findings we then propose a structured and systematic approach to construct sustainability conceptions as the basis for scientific inquiry. The approach is issue-driven, assesses unsustainability (rather than sustainability) and explicitly acknowledges the limitations of science to the descriptive realm. It consists of two steps: (1) an

* A modified version of this chapter has been submitted to *Ecological Economics* as Walter and Stützel: "A new method for assessing the sustainability of land-use systems (I): Identifying the relevant issues".

inventory of potential issues; and (2) a contextualisation step to identify which of the potential issues are actually relevant by means of a defined decision criterion. We here propose using ‘distance-to-target’ ratios as a decision criterion. Distance-to-target ratios are the quotient of the actual impact level to a critical impact level for a given issues, e.g. critical loads for eutrophication.

Testing this approach by the example of an agricultural area in Northwest Germany shows that it is practicable. Unlike typical prose definitions of sustainability, it yields a set of concrete issues against which to test the impact of agricultural production systems.

Sustainable development and sustainable agriculture

Agriculture is humanity’s predominant interface with nature. It is our prevailing source of income and livelihood and shapes most of Earth’s surface directly or indirectly (FAO, 2003; Tilman et al., 2002). It markedly influences global energy and matter cycles (Matson et al., 1997; Vitousek et al., 1997) while it is itself highly dependent on nature, natural cycles and the functioning of ecosystems. Structuring agriculture in a way that is compatible with Earth’s life support systems is thus a core concern of sustainable development. (In this paper we use the terms ‘sustainable development’ and ‘sustainability’ synonymously and perceive sustainable agriculture as the agricultural aspect of sustainable development).

Efforts to implement sustainable agriculture are being undertaken on various political levels, from municipalities to intergovernmental organisations, and there is hardly any agricultural research programme that does not claim a link to sustainable agriculture. However, in spite of an intensive debate until the mid-1990s (Christen, 1996, 1999), sustainability remains a vague concept, which is open to diverging interpretations. For scientists this is a particularly awkward situation, because any assessment and evaluation of sustainability requires clear conceptions about what to assess (cf. Allen et al., 1991). At the same time achieving clarity remains a political task, because there are no scientific solutions to questions of values, risk and priorities (Funtowicz and Ravetz, 1993; Hoyningen-Huene, 1999).

Christen (1998) titled an essay “No more definition please”, where he takes the stand for concentrating research on the (known) core themes of sustainable agriculture instead of producing even more ad lib definitions. This paper attempts to arrive at a workable conception of sustainability as the basis for scientific inquiry, without adding another definition, but by rather drawing on the existing wealth of writing on sustainable agriculture in a systematic way.

Definitions of sustainable development

Sustainable development is probably the most commonly accepted guiding principle in development and environmental policy throughout the world. In the frequently cited wording of the UN-WCED report 'Our common future' (the so called Brundtland Report; WCED, 1987) "Sustainable development is development that meets the needs of the present without compromising the ability of future generations to meet their own needs" (pages 8 and 43). This is further detaailed on pages 43 – 46: The WCED understands *development* as an economic and social transformation process with the aim of satisfying human needs and aspirations. *Sustainable* development refers to development that explicitly accounts for the biophysical constrains imposed by nature and its limited ability to absorb the effects of human activities.

Beside the WCED's conception of sustainable development there are countless other definitions, some similar, some differing (for a brief review see Iyer-Raniga and Treloar, 2000). None has, however, gained such broad consensus as the WCED's definition of sustainable development. For a detailed review also refer to Sum and Hills (1998).

Two core elements are usually discussed in the context of sustainable development (Sum and Hills, 1998; Conrad, 1999): intra-generational equity and inter-generational equity. The former refers to balancing interests between people living in different places (the spatial aspect) and the latter to people living in different times (the temporal aspect).

Another salient feature of sustainable development is its explicit anthropocentrism: It makes human needs and the aspiration for an improved quality of life the principal goal of sustainable development. Principle 1 of the Rio Declaration (UNCED, 1992) reflects this as well: "Human beings are at the centre of concerns for sustainable development. They are entitled to a healthy and productive life in harmony with nature." Some groups vigorously opposed this explicit anthropocentrism in the course of the preparatory process of the Rio Summit (Becker, 1997; Jamieson, 1998), but it is now widely seen as a constituent feature of sustainable development.

Definitions of sustainable agriculture

Other than in sustainable development, there is no single common definition of sustainable agriculture. We cite three illustrative examples:

> *"A sustainable agriculture is one that, over the long term, enhances environmental quality and the resource base on which agriculture depends, provides for basic human food and fibre needs, is economically viable, and enhances the quality of life*

> *for farmers and society as a whole."* The American Society of Agronomy (ASA, 1989)
>
> *"Sustainable land management combines technologies, policies and activities aimed at integrating socio-economic principles with environmental concerns so as to simultaneously:*
>
> - *maintain or enhance production/services (Productivity);*
> - *reduce the level of production risk (Security);*
> - *protect the quality/potential of natural resources and prevent soil and water degradation (Protection);*
> - *be economically viable (Viability);*
> - *be socially acceptable (Acceptability)."*
>
> International Working Group for the development of a Framework for Evaluation of Sustainable Land Management as cited by Smyth and Dumanski (1995).
>
> *"Sustainable agriculture is productive, competitive and efficient while at the same time protecting and improving the natural environment and conditions of the local communities."* Unilever (2002).

Although individual definitions of sustainable agriculture found in the literature differ in scope, focus and degree of detail, they do draw from a set of characteristic and recurrent elements. Typically, definitions state that, to be sustainable, agriculture must

- supply humanity with food and fibre of sufficient quantity and quality;
- not endanger Earth's life support systems (such as the climate system and the functioning of ecosystems) or natural resources (including biotic and abiotic resources, soils and biodiversity);
- allow producers to make a secure livelihood;
- contribute to rural development and the enhancement of rural communities;
- ensure the health of workers, rural population and consumers;
- be equitable, just and produce in a socially accepted way.

This list is also congruent with core themes in sustainable agriculture definitions that have been identified by Stockle et al. (1994) and Christen (1996, 1999).

From a scientist's perspective, the difficulty with such definitions of sustainable agriculture is that they are empirically empty: They use terms and concepts that are open to differing interpretations (such as 'sufficient', 'secure' or 'healthy'). As opposed to that, concrete and empirically testable conceptions of sustainability are needed as a basis for scientific inquiry (Allen et al., 1991). Filling empirically empty terms with concrete meaning is, however, neither a matter of logical delineation, because value judgements and priorities are involved that are not subject to scientific determination; nor is it straightforward, because the terms and concepts under debate often affect multiple and potentially conflicting agendas.

From an applied science perspective, a concrete conception of sustainability should be clear about the following questions (Costanza and Patten, 1995; Jamieson, 1998):

1. *What* exactly is to be sustained (or what is to be avoided)?
2. *To what degree* is it to be sustained (or to what degree is it tolerable)?
3. *For how long* is it to be sustained (or for how long is it to be avoided)?

In the literature on sustainability it is usually proposed to answer these questions in a participatory way (Norgaard, 1988; Zinck and Farshad, 1995; Lélé and Noorgard, 1996; Bosshard, 1997; Röling and Wagemakers, 1998; Bell and Morse, 1999; Iyer-Raniga and Treloar, 2000). This is also in line with recent currents in the theory of science, which specifically deal with value-laden, potentially controversial and policy relevant research agendas (Funtowicz and Ravetz, 1993; O'Hara, 1996; Tacconi, 1998; Nowotny et al., 2001).

However, participatory approaches have a number of drawbacks: First, a stakeholder group (or other body with normative legitimacy) is often simply not available (Fixdal, 1997). They are costly to set up and their decision processes take time. Policy decisions in the context with sustainability are usually urgent (Costanza, 1993) and science, if asked for decision support, does not normally have the budget or the time to set up a stakeholder group.

Second, the results of participatory approaches lack transparency: As a matter of definition, stakeholders pursue (individual or institutional) agendas (cf. Lélé and Noorgard, 1996). In the outcome of a participatory approach it is usually not possible to tell whether decisions are a product of these agendas or of pertinent arguments. Also, results of participatory processes are, as any social process, not replicable. Except for plausibility, there is no way of making them understandable to outsiders or latecomers.

Finally, sustainability, both as a concept and as a political agenda, emerged (and still evolves) in an interactive and iterative process, in which groups across a wide range of political and social backgrounds and from the local to the global level are involved (Becker, 1997; Sum and Hills, 1998). If we assume that this has led to a historically grown collective 'connotative' conception of sustainability, we may ask to what degree it is appropriate to redefine sustainability within a participative process. At least, any local or project-specific conception of sustainability should link onto the large-scale discourses on the global, regional and national level. Otherwise, there is the danger of the participatory project group creating their own world-view that is not shared or compatible with the outside world (Wynne, 1992).

We therefore ask whether or not it is possible to arrive at a concrete conception of sustainability as the basis for scientific inquiry, which (a) is systematic and transparent and (b) draws on the existing and ongoing discourse on sustainability. Before exploring this question we shall introduce three constructs that are useful for guiding our further analysis.

Defining what to define: Three useful analytical constructs

This section introduces three analytical constructs that are helpful in clarifying what we talk about when discussing conceptions of sustainability: goal-pursuing vs. issue-driven perceptions of sustainability; the normative vs. the descriptive realm of sustainability; and positive vs. negative statements on sustainability.

Goal-pursuing or issue-driven?

The first construct introduced here refers to two different sustainability perceptions: The *goal-pursuing* perception conceptualises sustainability as a development towards (or state of having reached) distinct environmental, economic or social goals. It is proactive in nature and targets to describe an ideal (viz. sustainable) situation. The *issue-driven* perception aims at alleviating concrete present issues. It is reactive. Targets describe minimum standards and damage thresholds. While the goal-pursuing perception conceptualises sustainability directly, the issue-driven perception is rather concerned with unsustainability. It conceptualises sustainability – if at all –, in negative delimitation: as the absence of a set of concrete present or anticipated issues.

These two perceptions are extremes of a continuum rather than opposites. Making the distinction between goal-pursuing vs. issue-driven does however prove useful in the analysis of sustainability perceptions. The emergence of sustainability as a concept, has, historically, been issue-driven (although implicit goals will have been in mind): Dwindling reserves of fossil fuels (Meadows et al., 1972), over-exploitation of renewable resources such as forests and fisheries (Ludwig et al., 1993; Wiersum, 1995), hunger and malnutrition (Douglass, 1984), ozone depletion and climatic change, inequity in the distribution of benefits from the use of natural resources (WECD, 1987; De Kruijf and Van Vuuren, 1998) – these and other pressing issues have strongly contributed to the formation of the concept of sustainable development. It is important to keep the issue-drive origin of the sustainability concept in mind to appreciate the set of themes usually addressed under the umbrella of sustainability. Some authors have called for a systematic 'science-based' delineation of these themes. Bearing the issue-driven nature of sustainability in mind it becomes clear that this is inadequate because, real-world issues do not follow any formal structure.

The fundamental problem with the goal-pursuing perception is that societal goals are often not available or not agreed (cf. UBA, 2001b), whereas there will frequently be a quite consensual conception of what the issues are. Consequently, many authors have argued that it is much easier to assess unsustainability rather than sustainability (e.g. Stockle et al., 1994; Syers et al., 1995; Jamieson, 1998; Smith and McDonald, 1998).

Christen (2000) argues that the core themes are known and often concepts are readily available and there seems to be a broad consensus on what the issues are (cf. Matson et al., 1997). Interestingly, the issues discussed in the context of sustainable agriculture are neither novel nor specific to that context (we shall return to this later). Virtually all issues discussed in the context of sustainable agriculture are within the scope of a specific discipline (such as soil science, toxicology, climate science, agricultural economics or development planning). For many of these issues, concepts are established and often mitigation strategies are available (Tilman et al., 2002). We may thus conclude that the novel and characteristic feature of sustainability is not the set of issues themselves, but the explicit recognition of their interlinked nature and a commitment to resolve them in an integrated way.

Normative or descriptive?

The second construct refers to the fact that science and scientists play a dual role in the sustainability debate: On one hand they are a part of society and policy decisions do affect their individual environments. On the other hand they are to inform societal and political decision making processes. We may call the first role the *normative* role and the second one the *descriptive* role. There is a fine line between these two roles and they are not always easy to separate. In fact, they are extreme poles of a continuum rather than clear-cut opposites. Nevertheless it is crucial to separate these two roles conceptually, because science, as our most received and important way of generating knowledge and defending knowledge claims (Hoyningen-Huene, 1999), receives particular weight in societal value setting processes, while the individual agendas of science and scientists do not.

In a similar context, Stevenson and Lee (2001) use the terms 'objective/scientific' and 'objective/political'. We prefer the nomenclature of 'descriptive' vs. 'normative', because 'scientific facts' are not 'objective' or free from values: The perception of reality, as the object of scientific description, is largely shaped by the cultural, professional and individual backgrounds of those describing (Dunlap et al., 1992; Lélé and Norgaard, 1996). Conversely, the societal processes, which lead to the formation of values, are influenced by scientific conceptions of reality (The establishment of such abstract concepts as ‘natural capital’ in the wider sustainability discourse is a prominent example). Societal value formation (as external normative reference for scientific inquiry) and the generation scientific facts (as a determinant of societal value formation) are thus linked and to some extent recursive (Radermacher et al, 1998). In embryonic fields, especially those having to do with large-scale environmental or policy-driven issues (cf. Funtowicz and Ravetz, 1993; Tacconi, 1998), values are often still under debate and evolving, whereas in more mature fields, they have attained the status of accepted norms.

The distinction between normative and descriptive is, as a construct, useful to mark the limitations of scientific legitimacy in the sustainability debate. This is even more important, as in the context of sustainability societal actors (scientists and non-scientists alike!) often retreat to the supposedly safe grounds of ‘objective science’ or ‘logic’, while actually promoting a mixture of (descriptive) information with (normative) individual views (Lélé and Norgaard, 1996).

Positive or negative statements on sustainability?

The third construct we introduce refers to the type of statements (or knowledge claims) we can make about sustainability from a scientific perspective. Above we stated that many authors suggested assessing unsustainability rather than assessing sustainability. Ludwig et al. (1993) even warn policy makers not to trust any scientific claims about sustainability. We here argue that this position can be well supported by a simple epistemological argument: Sustainability is about the very long term. Taking into account the very complex and multidimensional nature of sustainability we must admit that our models are not capable of making trustworthy predictions over such time spans (Oreskes et al., 1994; Ruelle, 1997; Barkmann and Windhorst; 2000). Consequently, we cannot prove claims about sustainability. Conversely, we can prove claims about unsustainability, as soon as there is evidence within the limited period that we can make predictions for.

What is the temporal extent of sustainability? If we take sustainability seriously as an ethical premise, it is temporally unconfined: E.g. the above-cited sustainability definition of the Brundtland Commission names ‘future generations’ as the temporal extent for sustainability. This might be surprising (and rather awkward) from a practical point of view, but it is very much consequent in the light of ‘inter-generational’ equity (see above): On what premise could we justify making any future generation worse off than we are? Based on the average historic life-span of mammal species (Newman and

Palmer, 1999), we can assume that 'future generations' is equivalent to some ten thousand to ten million years – i.e. six to seven orders of magnitude longer than the time spans we normally operate and decide on.

The periods for which we can make predictions are limited by different types of uncertainty that interact and potentially amplify each other (Ruttan, 1994; Barkmann and Windhorst; 2000):

- *Social and technological uncertainty:* We know neither what human needs, priorities and values will be in the future nor what means future generations will have to meet them.

- *Epistemic uncertainty:* We can neither be confident that our models account for all relevant factors nor can we quantify the risk of relevant factors not being included (this is also known as 'completeness uncertainty'). The gaps in our models are likely to be fundamental, because so are the gaps in our knowledge of the mechanisms governing ecological, social and economic systems and their interactions.

- *Stochastic uncertainty:* True randomness and unforeseeable phenomena occur in both natural and social systems (Kauffman, 1993; Gell-Mann, 1994; Lange, 1999). Even in well-described and relatively simple deterministic systems the laws of chaos may construct a 'genuine wall of unpredictability' (Ruelle, 1997), beyond which we cannot make predictions.

The predictability of the combined social and biophysical systems will hardly exceed a century and will probably be much shorter. We thus cannot substantiate the claim that a system is sustainable because we cannot even nearly overlook its time evolution over the period that is relevant in the context of sustainability. We can, however, claim it is unsustainable if it proves to be so within the period we can overlook. This is a variant of Popper's asymmetry between verification and falsification (Popper, 1963).

Most perceptions of sustainability do acknowledge this fact by taking a precautionary stance (Dovers and Handmer, 1995; Kutsch et al., 2001): E.g. the commitment to conserving natural resources and Earth's life-support systems is a rational consequence of the fact that we simply do not know whether they will be important for future generations or not.

Towards more systematic conceptions of sustainability

In the previous section we found that (1) issue-driven perceptions of sustainability are historically more defendable than goal-pursuing perceptions. 'Driving issues' tend to be quite straightforward, whereas societal goals are often under debate and difficult to link to concrete problems; (2) science and scientists take a dual role in the sustainability debate: They are both societal actors who are affected by the debate and its outcome as well as producers of (supposedly neutral) scientific input to that debate. As this is crucial when we attempt to make sustainability concrete, it is important to be explicit about normative and descriptive elements and to keep them conceptually apart; and (3) there is a pronounced mismatch between the timescales we can overlook and those relevant for sustainability. As a consequence, we can make defendable statements on unsustainability but not on sustainability.

Based on these findings we propose a formal and structured way to arrive at a sustainability conception that can serve as a basis for scientific inquiry. It is issue-driven, addresses unsustainability (rather than sustainability) and explicitly acknowledges the limitations of science to the descriptive realm. It comprises of

1. An *issue inventory* step, aiming at identifying all potential issues;
2. A *contextualisation* step, aiming at selecting the issues that are actually relevant within a given context.

Inventory of issues: What are the potential issues?

We propose to inventorise issues that are discussed in the context of sustainability through a comprehensive literature review. This review should encompass the scientific literature as well as publications by other societal groups engaged in the sustainability discourse, as intergovernmental organisations, governments and government agencies, NGO's, unions, industry, farmers' associations and others.

The implicit assumption behind this approach is that the literature provides a sufficiently accurate picture of the sustainability discourse. Indeed, much of the discourse takes place in literal form or is at least well documented (e.g. meetings of the WCED and UN-CSD; UN, 2003). However, we may also argue that not all opinions have equitable access to publication. Especially smaller groups at the local level may be precluded from publishing. It is therefore important to check results from the literature with local experts and stakeholders.

Contextualisation: How to decide whether an issue is relevant or not?

Within a concrete local context not all of the potential issues will actually be relevant. The decision of which ones are relevant will be straightforward in some instances, but debatable and subject to diverse perceptions in many others. It is thus desirable to have a standardised and transparent criterion for deciding whether or not a (potential) issue is actually relevant in a given context. Borrowing methodology from Life Cycle Assessment (LCA), we propose that ‘distance-to-target’ ratios (Müller-Wenk, 1996) could provide such a criterion. (The term ‘distance’ to target is highly misleading – see below – but will be used here in accordance with the original LCA terminology.)

In LCA, the distance-to-target ratio is the ratio between an *actual* level of an environmental effect and a *critical* level for that effect that must not be exceeded (Brentrup et al., 2004). This concept could be used as a criterion to decide whether or not a sustainability issue is relevant: A distance-to-target ratio greater than one would indicate the issue is relevant (i.e. the actual value is greater than the critical). A ratio less than one would indicate it is not. (Obviously there is not an issue if the actual level of an impact is below its critical value).

Using distance-to-target ratios as a decision criterion allows for integrating issues at different spatial scales. This makes it very attractive for our purpose, because sustainability issues emerge at a number of different scales (Niu et al., 1993; Dumanski et al., 1998; Smith and McDonald, 1998). Distance-to-target ratios, being a quotient of two values with the same dimension, yield dimensionless numbers that are comparable among each other.

Obviously it is important how the critical value is set. It can either be policy-based (e.g. international conventions) or science-based (e.g. critical load concepts). Policy-based critical values are adequate if targets are, at least to some degree, negotiable. This is normally the case in the social and economic dimension of sustainability. Policy-based critical values are, however, not adequate where bio-physical realities are involved, which are not negotiable. This is normally the case in the bio-physical dimension of sustainability: Global warming is (given one accepts that it actually happens) unaffected by the political debate about reducing emissions and no natural ecosystem will become less sensitive to nutrient inputs, even if agriculturalists nutrient emissions are at an unavoidable level.

Based on the above deliberations we thus opt for using science-based critical values wherever available. If unavailable, policy-based critical values may be taken as a substitute. This should however, be marked explicitly in order to ensure transparency.

Case study: Application to agriculture in Borken

In this section we report results from an application of the approach outlined above. The task was to profile sustainability issues for an agricultural production system situated in the County of Borken, North Western Germany, close to the Dutch border. The area is characterised by intensive agriculture with high livestock densities. The climate is temperate oceanic with a mean temperature of 9.7°C and mild winters, the annual precipitation is 700 to 800 mm. Soils are sands and loamy sands with 1.7% soil organic carbon, the soil pH ranges from 5.5 to 6.5. Farms are typically family businesses providing between one and two fulltime jobs. Farmers are usually well trained and the level of mechanisation is high. Typical farms encompass 40 to 80 ha of arable land, pasture and woodland. Maize is the dominant crop in rotations (40 – 50%). Farmers also crop (irrigated) vegetables (mainly spinach) and fine herbs for a local frozen foods factory, which allows them to diversify rotations and income sources. The rotation assumed here was the average of three typical four-year rotations. Further details on management and local conditions are given in Chapter 1.

Method and data

Inventory of potential issues: An inventory of potential issues was made on the basis of a literature review. The review included 44 references on sustainable agriculture published before 2002 (refer to footnote of Table 3.1 a). It was selected from the evaluation of a much larger pool of publications found by searching major publication databases for 'sustainable agriculture'. Criteria for selecting the references to be reviewed were originality and representativeness. The review included 30 *scientific* publications (papers published in reviewed journals, proceedings and books), seven publications from *international stakeholders* of sustainable agriculture and seven from *national organisations* (e.g. NGOs, unions and farmers associations, industry associations, government and intergovernmental organisations, churches and others).

These 44 references were analysed in four steps (modified Qualitative Contents Analysis; Mayring, 1993): (1) All items that the authors of a reference named and held relevant for threatening or constraining sustainability were recorded in the original wording. (2) Similar and identical items were grouped together. (3) Groups of items were paraphrased to match, where possible, existing and common scientific concepts (e.g. 'eutrophication' instead of 'nitrate losses to water courses'). In the following we refer to these paraphrased groups of items as 'issues'. (4) Finally, issues were assigned to broader thematic categories (e.g. 'resource related issues').

This four-step procedure was carried out separately for the physical and the social dimension (where 'physical' and 'social' are used in a wider sense and include geo-bio-chemical and economic issues, respectively; c.f. Chapter 2, Conrad, 1999). Based on roughly 240 single items named in the original literature, nearly 30 issues were identified each in the physical and social dimension (Tables 3.1 a and 3.1 b).

The following assumptions were made:

- The loss of biodiversity (named in 48% of the references) was regarded a secondary issue entirely determined by these primary issues (Sala et al., 2000): land use, habitat destruction, eutrophication, acidification and pesticide ecotoxicity. All of these determining primary issues are already listed.
- Likewise, the decline of soil biological activity (named in 11% of the references) was not listed separately because it was considered to be determined by the following primary issues: soil nutrient status, pH, organic matter status, salinity, compaction, tillage intensity and pesticide use (Syers and Springett, 1984; Buckerfield et al., 1997; Bandick and Dick, 1999; Haynes and Tregurtha, 1999).
- The decline in soil cation exchange capacity was assumed to be a function of the primary issues: soil pH, organic matter content and soil texture (Helling et al., 1964; Yuan et al., 1966).

Note that issues may appear in both the physical and the social dimension, but differ in perspective: E.g. 'water consumption' in the physical dimension is concerned with altering the hydrology of natural ecosystems while in the social dimension it is concerned with competition for alternative human uses.

We then asked if the sample of references taken from the literature was large enough to discover the majority of issues discussed in the literature sufficiently. This was assessed by plotting the number of issues found against the number of references analysed. One would expect the relation between the number of references analysed and the number of issues found to follow either a logarithmic or a saturation curve: The likelihood of discovering a 'new' issue by analysing more references should be lower, the more references have already been analysed. In fact, logarithmic functions described this relation for both the physical and social dimension well (r^2>0.96), as shown in Figure 3.1. The first 11 references contained 80% of all the issues identified. The regression functions suggested that, between 45 to 100 references, one would have to analyse over ten additional references to find one new issue. Analysing further references was thus unlikely to substantially add to the set of issues already found and was therefore carried out.

Contextualisation: The contextualisation was limited to the physical dimension due to the focus of the research project, but the methodology could be applied analogously to issues in the social dimension.

Concepts that describe the issues and appropriate spatial scales for their assessment were identified through literature research and expert interviews. It was assumed that the appropriate spatial scales are those at which an issue emerges and that

Table 3.1 a Inventory of potential issues in the *physical dimension* (encompassing ecological and bio-geo-chemical issues) of sustainable agriculture in Germany. Refer to text for details.

Code	Issues	Listed in reference nos. Arabic numbers: scientific literature lowerscript roman numbers : internat. stakeholders upperscript roman numbers: national stakeholders	% of total refs. (n=44)
A	*Soil fertility related issues*	*1-12, 14, 19-21, 24-27, 29, i-vii, I-VII*	*80*
A. 1	Degradation of biophysical properties	1-3, 7-13, 15, 19, 24, 27, 28, iii-vii, I, III, IV, VI	55
A. 1. 1	Soil loss	1-3, 7, 10-13, 15, 19, 24, 27, 28, iii-vii, I, III, IV, VI	50
A. 1. 2	Damage to soil structure	1, 3, 7-10, 12, 13, 15, 24, v-vii, I, VI, III	34
A. 2	Degradation of biochemical soil properties	1-3, 7-13, 15, 24, 26, 27, i, iii-vii, I, IV-VII	57
A. 2. 1	Salinisation	1, 2, 8, 13, 24, v	14
A. 2. 2	Acidification/alkalinisation	1, 3, 8, 10, 13, iii, iv, v, vii, VII	23
A. 2. 3	Contamination	9, 13, 15, vi, vii, IV, V	16
A. 2. 4	Depletion of soil organic matter	3, 7, 10, 12, 13, 15, 24, 27, I, VI	23
A. 2. 5	Nutrient depletion	8, 10, 24, 27, iii, iv, v,	16
A. 3	Soil hygienic degradation	11, 12, 26, 27, i	11
A. 3. 1	Build-up of pathogen potential	11, 12, 26, 27, i	11
B	*Resource related issues*	*1, 2, 4, 7-10, 12-16, 19, 21, 23-25, 28, 29, i-vii, I, III, IV, VI, VII*	*70*
B. 1	Consumption of non-renewable resources	2, 4, 8-10, 12, 13, 15, 16, 19, 21, 23, 25, 28, 29, iv, vii, IV, VII	43
B. 1. 1	Fossil fuel	2, 8-10, 12, 13, 15, 19, 23, 25, 29, III, IV, VI	32
B. 1. 2	Minerals (phosphate, potassium and limestone)	2, 9, 15, 19	9
B. 2	Occupancy of limited renewable resources	4, 7, 12-15, 19, 21, 24, 29, ii-v, vii,	32
B. 2. 1	Land	1, 4, 13	7
B. 2. 2	Water	1, 7, 12-15, 19, 24, 29, ii-v, vii,	30
C	*Emission related issues*	*1-10, 12-16, 19, 20-23, 25-30, i-vii, I-VII*	*91*
C. 1	Emission of climate relevant gases	2, 3, 9, 10, 12, 14, 15, 19, 20, 22, 26, i, vii, II, IV-VI	39
C. 1. 1	Greenhouse gases	3, 10, 14, i, vii, II, IV-VI	20
C. 1. 2	Stratospheric ozone depleters	20	2
C. 1. 3	Summer smog/ground level ozone	9, 14	5
C. 2	Emissions that affect other ecosystems negatively	1-10, 12-16, 20-23, 25-30, ii-vii, I-VII	89
C. 2. 1	Acidifying substances	3-5, 7, 9, 10, 12, 14, 21, 22, 26, 27, ii-vii, IV, VI, VII	48
C. 2. 2	Eutrophying substances	2-5, 7, 9, 10, 12-15, 21-23, 26, 27, ii-vii, I-VII	68
C. 2. 3	Pesticides	2-4, 6, 7, 9, 10, 12, 13, 15, 21, 27, ii-vii, I-VII	59
C. 3	Other emissions	7, 13, 26, ii, v, vi, I	16
C. 3. 1	Odours & noise	13	2
C. 3. 2	Waste	26, ii, I	7
C. 3. 3	Sediments	7, 13, v, vi	9
D	*Complex ecological issues*	*1-4, 8, 10-15, 20, 24-26, 28-30, i-vii, I-VII*	*73*
D. 1	Human health risks	1, 4, 8, 12, 13, 28-30, iii, v-vii, I, V, VII	34
D. 1. 1	Consumer health	4, 8, 12, 13, 30, iii, vi, vii, I, V, VII	25
D. 1. 2	Producer health	4, 8, 12, 13, 29, 30, I, V, VII	20
D. 1. 3	Local people/neighbours	12	2
D. 2	Impacts on species communities in ecosystems	1-2, 3, 10, 12-15, 20, 24-26, 28, i-vii, I-VII	61
D. 2. 1	Habitat destruction	1-3, 10, 12, 13, 15, 20, 25, 26, 28, ii-v, vii, I-VII	50
D. 2. 2	Formation of pesticide resistances	2, 11, 24, iii	9
D. 2. 3	Undesired ecological effects of GM crops	ii, iii	5

[1] Eswaran et al, 1994; [2] Ruttan, 1994; [3] v. Münchhausen & Nieberg, 1997; [4] Hansen, 1996; [5] Smyth & Dumansky, 1995; [6] Smith & McDonald, 1998; [7] Sands & Podmore, 2000; [8] Reganold et al., 2001; [9] Halberg, 1999; [10] Eckert et al., 2000; [11] Lefroy et al., 2000; [12] van Mansveldt & van der Lubbe, 1999; [13] Stockle et al., 1994; [14] Lewandowski et al., 1999; [15] Bockstaller et al., 1997; [16] Christen, 1996; [17] Roberts, 1995 (as cited by Christen, 1996); [18] Allen et al., 1991; [19] Miller & Wali, 1995; [20] Zinck & Farshad, 1995; [21] Farshad & Zinck, 1993; [22] Addiscott, 1995; [23] Steinborn & Svirezhev, 2000, [24] Wenz, 1999; [25] Dunlap et al., 1992; [26] Cornforth, 1999; [27] Taylor et al., 1993; [28] Becker, 1998; [29] Abelson, 1995; [30] Weil, 1990.

[i] Becker, 1997; [ii] IFAP/Via Campesina, 2000; [iii] NGOs at CSD8, 2000; [iv] International Agri-Food Network, 2000; [v] OECD, 2000; [vi] OECD, 1995; [vii] EU, 1999.
[I] DBV, 2000; [II] Hagedorn, 1997; [III] UBA, 1997; [IV] Loske, 1996; [V] RNE, 2002; [VI] Enquete-Kommission, 1994; [VII] SRU, 1994.

Table 3.1 b Inventory of potential issues in the *social dimension* (also encompassing political and economic issues) of sustainable agriculture in Germany. Refer to text for details, footnotes as in Table 3.1 a.

Code	Issues	Listed in reference nos. Arabic numbers: scientific literature lowerscript roman numbers: internat. stakeholders upperscript roman numbers: national stakeholders	% of total refs. (n=44)
D Society related issues		*1-16, 18-21, 24-30, i-vii, I-VII*	*93*
D. 1	Non-compliance with societal expectations	1, 3, 4, 6-10, 12-16, 18-21, 24-26, 28-30, i, ii, iv-vii, I-VII	82
D. 1. 1	Use of socially not accepted production techniques (e.g. agrochemicals, GM crops)	1, 6-10, 12, 14, 25, 26, i, ii, iv, v, I-IV, VI, VII	45
D. 1. 2	Socially not accepted production intensity (e.g. 'industrialised agriculture')	12, 18, 19, ii, iv, vii	14
D. 1. 3	Alteration of traditional landscapes and land use systems	3, 9, 10, 12, 13, 15, ii, vi, I, V-VII	27
D. 1. 4	Alteration of valued rural structures (social and economic)	4, 16, 25, 29, 30, ii, I, V, VI, VII	23
D. 1. 5	Lacking transparency of food production	25, i, V	7
D. 1. 6	Dependency on inequitable social structures (exploitative working relations, land tenure)	1, 4, 6, 16, 18, 19-21, 24, 28, i, ii	27
D. 1. 7	Disregard of animal welfare	12, 29, I, III, V	11
D. 2	Conflicting resources uses (external effects)	1-16, 19, 21, 24-30, i-vii, I-VII	89
D. 2. 1	Degrading soil use, overexploitation of marginal land	1-15, 19, 21, 24-27, 29, i-vii, I-VII	86
D. 2. 2	Land/soil surface occupancy	1, 12, 13, 25, 28-30, ii, iv, v	23
D. 2. 3	Deforestation/land clearing for agriculture	1, 2, 20	7
D. 2. 4	Water consumption	1, 7, 12, 14, 15, 19, 20, 24, ii, iv, v	25
D. 2. 5	Water pollution	1-3, 5-7, 9, 12, 14, 15, 20, 21, 28, 29, ii-vii, I-VII	64
D. 2. 6	Adverse impacts on hydrology (e.g. water table, salinisation, sedimentation)	7, 13, 24, iv-vii	16
D. 2. 7	Air pollution	3, 7, 9, 10, 12, 13, 15, 20, i, v-vii, II, IV-VI	39
D. 2. 8	Consumption of non-renewable or scarce resources	2, 4, 5, 9, 12, 15, 20, 21, 25, 29, i, iii, vii	30
D. 2. 9	Negative impact on biodiversity	1, 3, 9, 10, 12, 13-16, 25, 29, 30, ii, iii, v-vii, I-III, V, VI	50
D. 2. 10	Negative impact on quality of life in rural areas	4, 13, 25, ii	9
E Business related issues		*1, 2, 4-6, 8-19, 21, 25, 26, 28-30, i-vii, I-VII*	*84*
E. 1	Insufficient proceeds	1, 2, 4-6, 8-19, 21, 25, 26, 28-30, i-vii, I-VII	84
E. 1. 1	Low productivity	5, 6, 8, 10-14, 16, 17, 19, 21, 25, 26, 28-30, i, ii, iv, I, III, IV, VI	55
E. 1. 2	Poor product quality	4, 5, 8, 12, 13, 16, 26, ii-iv, vi, V	27
E. 1. 3	Lacking profitability, high cost and labour demand	1, 5, 6, 8, 11-13, 16-18, 21, 25, 30, V	32
E. 1. 4	High production risk	1, 5, 11, 13, 25, iv	14
E. 1. 5	Insufficient satisfaction of producers' spiritual goals	21, 30	5
E. 2	Lacking farm autonomy	4, 26	5
E. 2. 1	Dependency on dwindling resources	9, 12, 15, 16, 19, 26, ii, v, vi, III, IV, VI, VII	30
E. 2. 2	Dependency on single/few income sources	11, 28, iv	7
E. 2. 3	Dependency on subsidies or other external support	2, 12, 19, 28, ii, III, V-VII	20
E. 2. 4	Dependency on purchased inputs	21, 25, III, IV, VI, VII	14
F Market related issues		*8-10, 12, 13, 17, 24, 28, iii-v, I, III-VII*	*39*
F. 1	Inefficient resource use, poor allocation	8-10, 12, 13, 17, 24, 28, iii-v, I, III-VII	39
F. 2	Overproduction/Contribution to supply in over-saturated commodity markets	iii	2

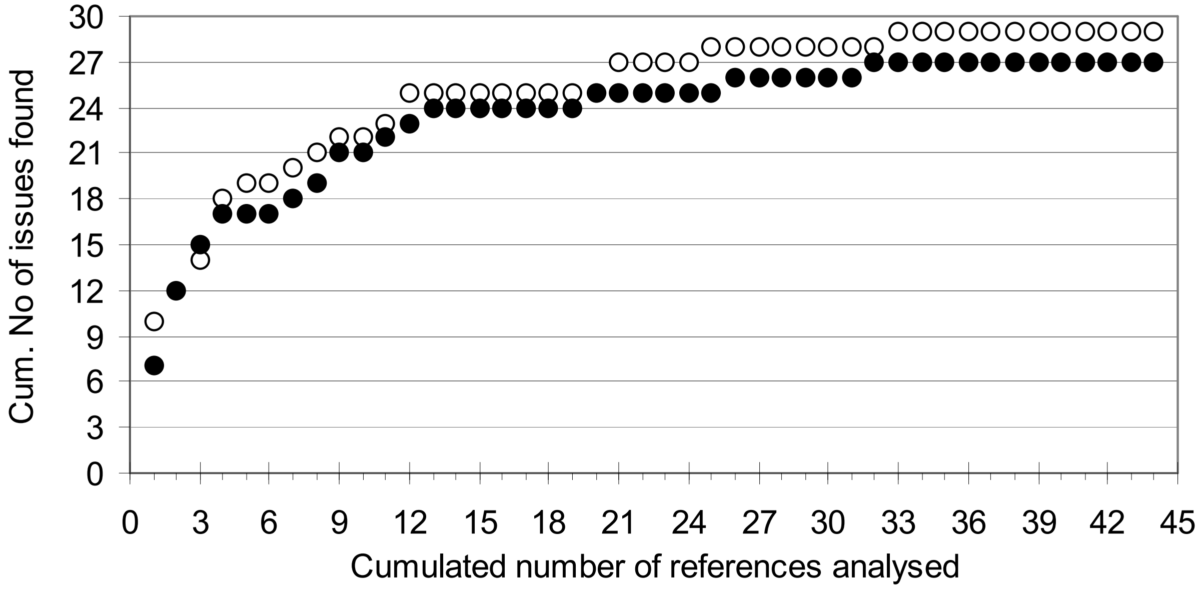

Figure 3.1 The cumulated number of potential sustainability issues found plotted against the cumulated number of references analysed (see text for details). Solid symbols (●): physical dimension, open symbols (○): social dimension. Curves can be fitted with a logarithmic function (r^2 = 0.97 for the physical and r^2 = 0.98 for the social dimension). References 1 to 30 are from the scientific literature (refs. 1 – 30 in Table 3.1); references 31 to 37 by international organisations and stakeholders of sustainable agriculture (refs. i – vii in Table 3.1); references 38 to 44 by national organisations and stakeholders of sustainable agriculture (refs. I – VII in Table 3.1).

these levels are defined within the pertinent disciplines. All data were, wherever possible, taken from the public domain or official statistics. Science-based *critical values* were prioritised. If unavailable, policy-based values were used as a proxy. Outcome and source data are documented in Table 3.2.

For some potential issues, it was not possible to compute distance-to-target ratios:

- For some potential issues, concepts and methods were lacking. These were: *Odours and noise* (C.3.1), *Waste production* (C.3.2), *Emission of sediments* (C.3.3) and *Formation of pesticide resistances* (D.2.2).
- No information on actual and/or critical values was available for *Toxicity of pesticides to humans* (D.1.2 and D.1.3) and *Habitat destruction* (D.2.1). Although concepts are available for Human toxicity of pesticides (e.g. ADI-values; FAO/WHO, 1975), daily intake and exposure data were not. Models for human exposure (WHO, 1997; WHO, 1999) could be used to generate such data, but this was beyond the scope of this study.

Table 3.2 Potential sustainability issues and distance-to-target ratios for agriculture in the County of Borken, Germany (physical dimension of sustainable agriculture). Issues with a distance-to-target ratio greater or equal to one are classified 'relevant'. Uncertainty was assessed according to the NUSAP notation scheme for data quality (Costanza et al. 1992). The three entries under 'pedigree' are scores for (1) the theoretical quality, (2) the empirical quality and (3) the social quality, as described in Table 2.1. If not indicated otherwise, all data are for 2000.

Code	Issue	Spatial scale level	Underlying goal (target level of goal achievement)	Indicator	Distance to target: actual / target	(unit)	Ratio	Notes	Relevant	Uncertainty: Spread‡	Pedigree	Grade
A Soil fertility related themes												
A. 1. 1	Soil loss	field	Protect soil fertility (No net loss)	Input/output ratio (mass balance)	1.0 / 1.0	dimensionless	1.0	1 a	yes	VH	{ 4; 2; 3 }	0.7
A. 1. 2	Damage to soil structure	field	Protect soil fertility (Secure physical conditions for crop growth)	Soil pressure/resistance ratio	1.7 / 1.0	dimensionless	1.7	2 a	yes	H	{ 2; 2; 2 }	0.5
A. 2. 2	Acidification/alkalinisation	field	Protect soil fertility (No net input)	Input/output ratio (mass balance)	1.8 / 1.0	dimensionless	1.8	3 a	yes	H	{ 4; 2; 3 }	0.7
A. 2. 3	Contamination	field	Protect soil fertility (No net input)	Input/output ratio (mass balance)	3.5 / 1.0	dimensionless	3.5	4 a	yes	H	{ 3; 1; 1 }	0.4
A. 2. 4	Depletion of SOM	field	Protect soil fertility (No net loss)	Input/output ratio (mass balance)	1.1 / 1.0	dimensionless	1.1	5 a	yes	VH	{ 2; 1; 2 }	0.4
A. 2. 5	Nutrient depletion	field	Protect soil fertility (No net loss)	Input/output ratio (mass balance)	1.0 / 1.0	dimensionless	1.0	6 a	yes	M	{ 3; 3; 3 }	0.7
A. 3. 1	Build-up of pathogen potential	field	Protect soil fertility (Safe cropping intervals)	Cropping frequency	0.7 / 1.0	dimensionless	0.7	7 a	no	S	{ 1; 2; 3 }	0.5
B Resource related themes												
B. 1. 1	Fossil fuel consumption	global	Prevent rapid and complete resource depletion (Phase out consumption)	Consumption rates	36 / 32	(Million TJ primary energy)	1.1	8	yes	S	{ 2; 3; 2 }	0.5
B. 1. 2	Consumption of minerals (phosphate, potash, limestone)	global	Prevent rapid and complete resource depletion (Phase out consumption)	Consumption rates	27 / 24	(Mt K_2O-eq.)	1.1	8	yes	S	{ 2; 3; 2 }	0.5
B. 2. 1	Land occupancy	regional	Conserve land for wildlife/ecological services (Extensively farmed land with 10% nat. area)	Naturalness Degradation Potential	0.64/0.55	dimensionless	1.2	9 a	yes	M	{ 1; 2; 1 }	0.3
B. 2. 2	Water consumption	catchment	Protect integrity of aquatic ecosystems (Sufficient water flow for freshwater ecosyst.)	Ecological Water Scarcity Index	0.75/0.30	dimensionless	2.5	10 b	yes	VH	{ 0; 1; 3 }	0.3
C Emission related themes												
C. 1. 1	Greenhouse gases	global	Stop global warming (stabilise atmosph. CO_2-level at 450 ppmv)	Global Warming Potential	24 / 11	(Gt CO_2-eq.)	2.2	11	yes	M	{ 3; 3; 3 }	0.7
C. 1. 2	Stratospheric ozone depleting substances	global	Protect atmospheric ozone layer (Stabilise atmoph. ozone at pre-1980s level)	Ozone Depletion Potential	0.6 / 1.0	(Gt ODP)	0.6	12 b	no	M	{ 4; 3; 3 }	0.8
C. 1. 3	Summer smog/ground level ozone precursors	continental	Protect human health; prevent damage to vegetation (Stay below critical exposure levels)	Modelled ground-level ozone concentrations	16.8 / 5.8	(Mt NM VOC)	2.9	13	yes	M	{ 2; 3; 3 }	0.6
C. 2. 1	Acidifying substances	continental	Conserve sensitive (semi-) natural ecosystems (Stay below critical loads for >95% of ecosyst.)	Acidification Potential	36 / 44	(Gt SO_2-eq.)	0.8	14	no	M	{ 3; 3; 3 }	0.7
C. 2. 2	Eutrophying substances (terrestrial)	continental	Conserve sensitive (semi-) natural ecosystems (Stay below critical loads for >95% of ecosyst.)	Terrestrial Eutrophication Potential	45 / 22	(Gt NO_x-eq.)	2.0	15	yes	M	{ 3; 3; 2 }	0.6
C. 2. 2	Eutrophying substances (marine)	catchment	Conserve sensitive (semi-) natural ecosystems (Stay below critical loads for >95% of ecosyst.)	Aquatic Eutrophication Potential	0.65/0.58	(Gt PO_4-eq.)	1.1	16	yes	VH	{ 2; 2; 2 }	0.5
C. 2. 3	Pesticides (eco-tox)	regional	Conserve sensitive (semi-) natural ecosystems (Potential ambient concentrations below NOELs*)	Normalised Treatment Index	2.4 / 4.6	dimensionless	0.5	17 a	no	VH	{ 1; 3; 2 }	0.5
C. 3. 1	Odours & noise	field/regional	Protect human health and well-being	NA†					?	-		
C. 3. 2	Production of waste	regional-global	Protect human health & ecological functions	NA†					?	-		
C. 3. 3	Emission of sediments	regional	Conserve water bodies for wildlife/ hydrological functions	NA†			<1	b	no	-	{ 0; 1; 3 }	0.3
D Complex ecological themes												
D. 1. 1	Consumer health	national	Protect human health (Exposure below NOELs*)	Pot. exposure (risk assessment)	96 / 100	(%exceedances)	1.0	18	yes	M	{ 1; 3; 1 }	0.4
D. 1. 2	Producer health	regional	Protect human health (Exposure below NOELs*)	Pot. exposure (risk assessment)	NA†				?	-		
D. 1. 3	Local people/neighbours	regional	Protect human health (Exposure below NOELs*)	Pot. exposure (risk assessment)	NA†				?	-		
D. 2. 1	Habitat destruction	regional	Conserve biodiversity and species communities	Level of fragmentation	NA†				?	-	{ 1; 2; 1 }	0.3
D. 2. 2	Pesticide resistances	regional-global	Food security/ availability of control mechanisms	NA†					?	-		

[a] See Annex for details on rotation and management.
[b] Appraised not to be relevant, based on expert opinion (see text for details).

* NOEL = No-Observed-Effect-Level.
† NA = not available

‡ Uncertainty estimate for distance-to-targe ratio: L= low (10%); M = medium (50%); H = high (100%); VH = very high (>100%).

Notes to Table 3.2

[1] *Actual value*: Soil Input/Output Ratio, where input = soil formation rate (average given by Troeh et al., 1998) and output = erosion rate. Erosion rate after Universal Soil Loss Equation (Renard et al., 1997) as adapted for Germany by Schwertmann et al. (1987) and Hennings (2000). Also see Annex I.
Critical value: Input/Output Ratio = 1 (i.e. outputs do not exceed inputs). Refer to Table A.1 (Annex II) for data used.

[2] *Actual value*: Soil Compaction Index (potential compaction). Soil compaction index = $\Sigma(P_{ex})/\Sigma(R_{ex})$, where $\Sigma(P_{ex})$ is the sum of tyre pressures of compacting wheel passes and $\Sigma(R_{ex})$ the sum of soil mechanical resistances of compacting wheel passes. 'Compacting wheel passes' are passes where contact area pressures in 15 cm depth exceed the soil's mechanical resistance in 15 cm depth. The soil's mechanical resistance is defined equivalent to the pressure above which, at a given soil moisture, the soil pore volume declines below 8% (Horn et al., 1996; Paul, 1999). This is equated with insufficient gas and water exchange for crop growth (Drew, 1992). Also see Annex I.
Critical value: Soil Compaction Index = 1 (i.e. pressures do not exceed resistance). Refer to Table A.2 (Annex II) for data used.

[3] *Actual value*: Proton Input/Output Ratio. Proton equivalents based on stoichiometry. Inputs: N fertilisation, atmospheric N and S deposition. Outputs: crop uptake, NH_3 volatilisation, denitrification, liming. Full nitrification of NH_X assumed (after subtraction of volatilisation), all N uptake assumed to be NO_3, of which are 50% assumed to lead to proton consumption. Also see Annex I.
Critical value: Input/Output Ratio =1 (i.e. inputs do not exceed outputs). Refer to Table A.3 (Annex II) for data used.

[4] *Actual value*: Heavy Metal Accumulation Index. Metals taken into account: Cd, Cr, Cu, Hg, Ni, Pb and Zn. Accumulation index = $\Sigma(\text{Inputs}_{ex})/\Sigma(\text{Outputs}_{ex})$, where Σ (Inputs_{ex}) is the sum of normalised inputs of metals where the input-output balance is positive and $\Sigma(\text{Outputs}_{ex})$ is the sum of outputs of these metals. Inputs = metal loads in fertilisers + atmospheric deposition. Outputs = plant offtake + leaching. Normalisation was achieved by dividing all values through the '*Vorsorgewert*' of each metal (ecotoxicological precautionary level, as specified in BBodSchV, 1999). Also see Annex I.
Critical value: Heavy Metal Accumulation Index =1 (i.e. inputs do not exceed outputs). Refer to Table A.4 (Annex II) for data used.

[5] *Actual value*: Soil Organic Matter (SOM) Output/Input Ratio. Input: organic fertilisers, harvest residues. Output: SOM decomposition due to cropping. SOM loss and reproduction values are defaults from Leithold et al. (1997). Also see Annex I.
Critical value: SOM Output/Input Ratio 1 (i.e. outputs do not exceed inputs). Refer to Table A.5 (Annex II) for data used.

[6] *Actual value*: Output/Input Ratios for P and K (only figures for P shown; the ratio for K is less constraining: 129 / 157 = 0.8, leaching not accounted for). Other soil nutrient levels were not assessed but are regarded sufficient in 90% of the soils in the region (classes C to E in German LUFA classification scheme; Lammers, 1999). Also see Annex I.
Critical value: Nutrient Output/Input Ratios =1 (i.e. outputs do not exceed inputs). Refer to Table A.6 (Annex II) for data used (all data 1999 values).

[7] *Actual value*: Crop Rotation Index in standard rotation (Box 1.1). Crop rotation index = $\Sigma(FA_{ex})/\Sigma(FR_{ex})$, where $\Sigma(FA_{ex})$ is the sum of actual cropping frequencies of crops exceeding the recommended frequency and $\Sigma(FR_{ex})$ the sum of recommended frequencies for these crops. Cropping frequencies were calculated as years in which a crop is grown divided by the rotation length in years. The recommended frequency is the phytosanitary safe frequency (local experts and agricultural extension services).
Critical value: Crop Rotation Index = 1 (i.e. no copping frequency higher than recommended). Refer to Table A.5 (Annex II) for data used.

[8] *Actual value*: Global consumption of individual fossil and mineral resources.
Critical value: Given by power law $C_i = 0{,}9{*}C_{i-1}$, where C_i is the annual consumption in decade i and C_{i-1} is the annual consumption in the previous decade. Based on the premises that (a) a sustainable use would not allow for complete depletion and (b) extraction should be phased out subsequently to allow for substitution or other adjustment (Dresselhaus and Thomas, 2001). Base-line year is 1999, i.e. in the decade 2000 to 2009 the critical value for the annual consumption is 90 % of the consumption in 1999. The factor 0.9 is arbitrary. Note, that the absolute depletion differs between resources. Fossil energy reserves and consumption data from EIA (2001), mineral resources data (here shown for potash) from USGS (2001). Global reserves of limestone are vast (no quantitative estimates available). The consumption of limestone was therefore not regarded to constrain sustainability.

[9] *Actual value*: Naturalness Degradation Potential (Brentrup et al., 2002) of actual land cover in the County of Borken. Also see Annex I.
Critical value: Naturalness Degradation Potential of desired land cover. Desired rural land cover: Diverse cultural landscape sustaining agricultural production and conserving biodiversity as defined by SRU (1985, 1994): 10 % (semi-) natural habitat within a diverse landscape and largely extensive agriculture. This was here assumed to be equivalent to ETC/LC code 2.4.3, 'land principally occupied by agriculture, with significant areas of natural vegetation (25-75% agricultural area)'. Urban settlement area was accounted for with 12% of the total land area and equated to an average of ETC/LC codes 1.1 and 1.2. Land use data from Kreis Borken (2002).

[10] *Actual value*: Ecological Water Scarcity Index for Rhine-Mass basin (Smakhtin et al., 2004, data for Rhine-Mass basin taken from WRI, 2003b).
Critical value: Ecological Water Scarcity Index = 0.3 Ecological Water Scarcity Index = (Human water abstraction) / [(river basin run-off) – (ecological flow requirement)]. Ecological flow requirement: flow requirement and seasonal amplitude of flow (prior to human abstraction) assumed to form the ecological niche, to which biocoenoses are adapted (Smakhtin et al., 2004; Petra Döll, 2004, personal communication).

[11] *Actual value*: Global Warming Potential of global greenhouse gas emissions: CO_2 from fossil fuel combustion and cement manufacturing (WRI, 2003a), and N_2O and CH_4 emissions in CO_2-equivalents (N_2O and CH_4 are EDGAR estimates; Olivier et al., 2002. All EDGAR estimates 1995 data; changes between 1995 and 2000 are considered negligible.). Also see Annex I.
Critical value: Long-term (2100) energy related emission level at which atmospheric CO_2-concentrations would stabilise at 450 ppmv (IPCC, 2001). Based on the IPCC post-SRES scenario evaluation (IPCC, 2001) this emission level was assumed to be 3 Gt C per year. Methodological differences between the emission estimates (IPCC post-SRES scenarios vs. WRI/EDAGAR) were less than 5% in the comparable reference year 1990 and thus assumed to be tolerable.

[12] *Actual value*: Ozone Depletion Potential (WMO, 2002) of global CFC emissions.
Critical value: Emissions of ozone depleting substances at pre-1980's rate, which is assumed to stabilise global stratospheric ozone concentration at a pre-1980's level in the long term (WMO, 2002).

[13] *Actual value*: European NMVOC emissions as a proxy indicator for ground level ozone concentrations. The correlation between ozone concentrations and NMVOC emissions was assumed to be linear for Europe (Heyes et al., 1996). Ground level ozone concentrations calculated with RAINS from NO_x and NMVOC emissions (Alcamo et al., 1990; IIASA, 2003). Actual NO_x and NMVOC emission levels from EMEP (Vestreng, 2003).
Critical value: NMVOC emissions that lead to less than 0.1 ppm*hours exceedance of AOT60 (8-hour average), which was defined as a critical level for human health protection by WHO-EH (Amann & Lutz, 2000). This is reached at emission levels of the RAINS scenario 'MFR_{ult}' (Amann et al., 1999). The critical level for damage to vegetation is less constraining (UN/ECE; Amman & Lutz, 2000) and always met for the above scenario.

Notes to Table 3.2 (cont.)

[14] *Actual value*: Acidification Potential of European NO_x, SO_4 and NH_3 emissions. Acidification Potentials after Huijbregts et al. (2000) as modified by Brentrup et al. (2003). Also see Annex I.
Critical value: Acidification Potential of emissions leading to critical load exceedance in less than 5% of ecosystems (Posh et al., 1995; UBA, 1996). Exceedance of critical loads calculated with the online version of RAINS (as of July 2003, Alcamo et al., 1990; IIASA, 2003). The target is met at emission levels that were determined by linear interpolation between (1) the RAINS 1990 base line and (2) the 'REF' scenario (Amann et al., 1999). Emission data from EMEP (2002a, b).

[15] *Actual value*: Terrestrial Eutrophication Potential of European NH_3 and NO_X emissions. Terrestrial Eutrophication Potentials after Huijbregts et al. (2000) as modified by Brentrup et al. (2003). Also see Annex I.
Critical value: Eutrophication Potential of emissions leading to critical load exceedance in less than 5% of ecosystems (Posh et al., 1995; UBA, 1996). Exceedance of critical loads calculated with the online version of RAINS (as of July 2003, Alcamo et al., 1990; IIASA, 2003). The target is met at emission levels slightly below that of the 'MFRult' scenario (Amann et al., 1999). Emission data from EMEP (2002a, b).

[16] *Actual value*: Aquatic Eutrophication Potential of dissolved N and P emissions and atmospheric N deposition into the North Sea catchment. Aquatic Eutrophication Potentials after Redfield (1958) with fate factors for gaseous emissions from Huijbregts & Seppälä (2001) as described by Brentrup et al. (2003). Also see Annex I.
Critical value (*policy-based*): Reduction of nutrient input to 50% of the 1985 level (CONSSO, 2000). Emission data for North Sea catchment from CONSSO (2000) and OSPAR (2000). UK data not included because of methodological differences. Atmospheric N deposition (1998 values) from EMEP (1999). Critical values for freshwater ecosystems are normally less constraining than those for marine ecosystems and were therefore assumed not to be exceeded when target levels for marine ecosystems were met.
Analogue data for the Baltic Sea: *Critical value* (*policy-based*): 50% of the 1980 nutrient input level. Distance-to-target ratio: 0.40 / 0.37 = 1.1. Data from Finnish Environment Institute (2002); HELCOM (2003); EMEP (1999).

[17] *Actual value*: Pesticide Use Intensity Index (*Behandlungsindex*) for the County of Borken after Gutsche and Enzian (2002). Also see Annex I.
Critical value (*policy-based*): The critical value was derived in two steps: (1) On the municipality (*Gemeinde*) level, critical values for the Pesticide Use Intensity Index were derived by Gutsche and Enzian (2002), based on a municipality's endowment with small (semi-) natural landscape elements (BBA, 2002). This is in line with expert recommendation of the SRU (1985, 1994). (2) The critical values on municipality level are based on the assumption that the overall national use intensity is at an acceptable level. This is, however is under debate. Toxicology-based national reduction targets are not yet available and policy-based ones range between 50% (PAN/EEB, 2002) and 15% reduction. Based on expert opinion (Volkmar Gutsche, 2003, personal communication), we assumed a future consensual target of 20%.
The total *critical value* was therefore calculated by multiplying the critical value for the Pesticide Use Intensity Index of 5.8 with a factor of 0.8 to account for the (assumed) national reduction target (0.8*5.8 = 4.6). Data from BBA (2002), Roßberg et al. (2002) and own data (Box 1.1). 2004 version of Relation between endowment with landscape elements and critical Pesticide Use Intensity Index from Enzian (2003, personal communication).

[18] *Actual value*: % exceedance of Maximum Residue Levels (MLRs) in food sold in Germany (EU, 2003).
Critical value: 100% compliance with EU MLRs. Ratio inverted to obtain distance-to-target ratio.

- *Sedimentation* (C.3.3) was judged irrelevant based on local expert appraisal (local water supplier, local water conservation board and ecological research station). For the remaining six cases a decision about relevance could not be made.
- Science-based critical values for *Marine eutrophication* (C.2.2 b) and *Emission of pesticides* (C.2.3) were unavailable and substituted with policy-based ones.
- *Soil salinisation* (A.2.1) and *Soil alkalinisation* (A.2.2) were appraised irrelevant due to climatic and soil conditions and not evaluated, likewise *Risks due to genetically modified crops* (D.2.3), as no GM varieties are cropped in the region.

For further documentation refer to the notes to Table 3.2.

The methodological and epistemological uncertainty of the distance-to-target ratios was described using the ‘assessment grade’ of the NUSAP notation scheme for data quality (Funtowicz and Ravetz, 1990, as modified by Costanza et al., 1992). The assessment grade is a semi-quantitative measure of a number’s quality, assessed by three criteria: (1) theoretical quality (how well does the underlying model explain a phenomenon?); (2) empirical quality (how good are the data? how were they gathered?); and (3) degree of acceptance (how well established is the theory in the scientific community?). Each of these criteria is scored on an ordinal scale from 0 to 4, as shown in Table 2.1. The ‘assessment grade’ is the average of these scores. For easier reference the score was divided by three to yield a number between zero and one, where 1 denotes the lowest and 0 the highest uncertainty.

Results

Results are shown in Figure 3.2. A number of potential issues were identified as irrelevant in the study area: Nutrient depletion (A.2.5), Build-up of soil pathogen potential (A.3.1), Contribution to ozone depletion (C.1.2), Emission of acidifying substances (C.2.1), Pesticide use (C.2.3) and Consumer health risks (D.1.1). The five issues with the highest distance-to-target ratios (i.e. the largest exceedance of critical values) are: Heavy metal contamination of soils (A.2.3) > Ground level ozone (C.1.3) > Water consumption (B.2.2) > Emission of greenhouse gases (C.1.1) > Terrestrial eutrophication (C.2.2 a). In total, thirteen of the potential issues were identified as being relevant. No decision could be made for seven potential issues due to lacking methods or data and two potential issues were classified as irrelevant on the basis of expert appraisal.

The physical dimension of sustainable agriculture in the County of Borken can thus be defined as consisting of these thirteen issues. Any sustainability assessment for an agricultural production system then requires testing whether or not the system contributes to these issues thirteen issues. This step will be addressed in Chapter 5.

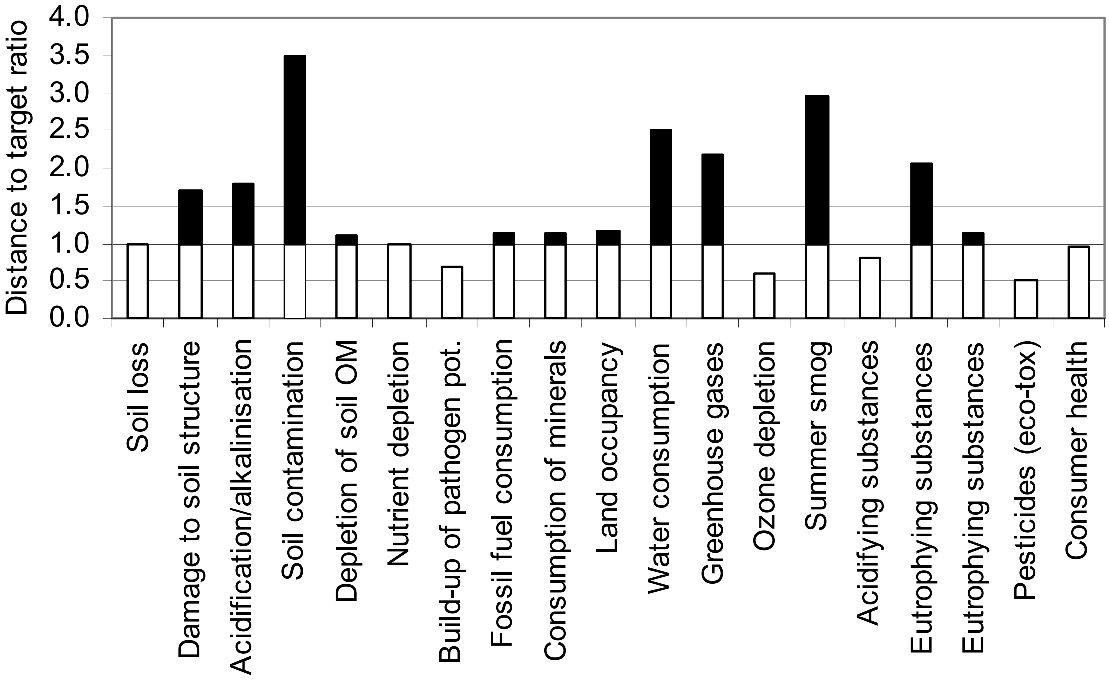

Figure 3.2 Distance-to-target ratios for potential issues in sustainable agriculture calculated for the County of Borken, Northwest Germany (physical dimension; some potential issues not assessed due to lack of methodology or data; see text). Distance-to-target ratio greater than one (black section of bars) indicate that an issue is relevant within the study area, ratios less than one (white bars) indicates it is not relevant).

Chapter summary and discussion

Most definitions of sustainability are unsuitable as a basis for scientific inquiry because they lack concreteness. We therefore proposed a structured and systematic approach to constructing context-specific sustainability conceptions. We also discussed three requirements the approach should meet, namely (1) to taken an issue-driven perspective on sustainability (rather than a goal-pursuing one), because this is historically more adequate and much simpler to manage; (2) to be explicit about and keep apart the normative and descriptive elements of any sustainability conception, because in the context of sustainability science and scientists play both a normative and a descriptive role; and (3) to restrict itself to making statements about unsustainability (rather than sustainability), because sustainability implies a long-term perspective that we cannot overlook, whereas symptoms or threats of unsustainability are often already visible.

Our approach to constructing sustainability conceptions breaks the task of ‘defining’ sustainability down to two questions that can be answered in a systematic way: (1) What are the potential issues? and (2) which potential issues are actually relevant in a specific context? Answering these questions yields what ecologists call the ‘constraint envelope' of sustainable agriculture (O'Neill et al., 1989), i.e. the set of factors that threaten (or constrain) the sustainability of the particular system under investigation. The actual sustainability assessment then consists of investigating if and how much the system contributes to these constraining factors.

We outlined a systematic way to answer these two questions. We proposed it would separate normative and descriptive elements conceptually. However, normativity inevitably enters the process through inevitable choices. E.g. in the *Inventory of potential issues*, the first of the two steps, these are:

- *'Sampling' of references:* The set of potential issues found fundamentally depends on the selection of references. The necessary 'sampling size' can, as was done here, be approximated by plotting the number of issues fund against the number of references analysed (Figure 3.1) in order to indicate whether further references are likely to add new issues to the list.
- *Choice of scale:* Emergence of certain phenomena and patterns depends largely on the choice of the scale at which we assess them. The most 'appropriate' scale is that at which the phenomenon or pattern under investigation emerges most pronouncedly (Wiens, 1989). We assumed that the appropriate spatial scale on which to evaluate an issue is an external function set by the pertinent discipline. Scales are usually quite straightforward, e.g. the dynamics of N_2O are global and affect the global climate system while eco-toxicological effects of pesticides are normally confined to a local scale.
- *Choice of methods:* The choice of methods and concepts implies certain fundamental beliefs and assumptions. We aimed at using methods and concepts that are common and accepted within relevant disciplines, such as Greenhouse Warming Potentials (IPCC, 2001) or the Universal Soil Loss Equation (Renard et al., 1997). However, often there are competing schools of thought or the field is new and emerging. A useful means of controlling biases due to methodological choices (or conceptual 'pedigree') is the NUSAP notation scheme.
- *Data collection and handling:* Collection and processing of data requires individual choices, e.g. data source, treatment of missing data and of methodological inconsistencies. We aimed at maximum transparency by using public domain databases and thorough documentation.

For the second step of the process, the *Contextualisation*, the manner of determining critical values is obviously important. Above we argued for using science-based critical values, but science-based critical values are, of course, neither 'objective' nor free from values. However, they are rooted in a broader disciplinary discourse and are subject to standard mechanisms of scientific quality control (such as peer reviewing). They have therefore been aligned with the paradigm of the pertinent discipline(s). This makes (inevitable) choices and value judgements more transparent and defendable. At the same time it prevents those involved in sustainability projects from constructing their 'own' epistemology that is not shared outside their group.

As the case study showed, distance-to-target ratios can be computed from publicly available data for most potential sustainability issues. A major advantage is their potential to integrate issues at different spatial scales, because the division operation eliminates different spatial extents.

One could argue that distance-to-target ratios are too one-dimensional a criterion for deciding whether a potential issue is actually relevant or not. They do not account for a range of other politically relevant factors, such as the number of people affected by an issue or its reversibility. More complex criteria could arguably take into consideration more than one factor. But they would most likely lose transparency and certainly carry a much higher normative charge than the simple distance-to-target ratios proposed here.

CHAPTER 4

Estimating N and P emissions from agricultural fields

Abstract

Emissions of nitrogen and phosphorus species from agricultural fields are significant drivers of large-scale environmental issues, such as the decline of biodiversity and climatic change. Quick, reliable and cost efficient quantification of these emissions is therefore key to many research agendas and policy purposes. In this paper we introduce a number of frequently used simple methods to estimate ammonia, nitrous and nitric oxide, nitrate and phosphorus emissions from agricultural fields. The study focuses on Germany, but may be taken as exemplary for temperate climate regions.

We evaluate the introduced methods for (1) their sensitivity to environmental and management factors, (2) their uncertainty and (3) their robustness against aggregation biases. While methods for estimating gaseous emissions are available that meet these criteria well, methods for estimating losses to water are subject to high uncertainties. A case study shows that using different estimation methods for the same emission type may yield results that differ significantly.

Introduction

Agricultural fields are a substantial source of environmentally relevant N and P-species (Matson et al., 1997). Table 4.1 shows emissions from agricultural field in relation to the total anthropogenic emissions of ammonia (NH_3), nitrous oxide (N_2O), nitric oxide (NO_x), nitrate (NO_3) and phosphorus (P). These nutrient losses contribute to a number of relevant environmental issues: N_2O (as a highly potent greenhouse gas) and NH_3 (as an aerosol) are involved in global warming (Enquete Kommission, 1994); N_2O might also play a role in stratospheric ozone depletion (Mosier, 2001). NO_x, regulates the formation of tropospheric ozone and is a major constituent of 'summer smog' (Enquete

Kommission, 1994; Lerdau et al., 2000). NH_3, NO_x, NO_3 and P cause eutrophication of terrestrial and aquatic ecosystems (Isermann and Isermann, 1998; Reynolds and Davies, 2001; Tilman et al., 2002). Climatic change and eutrophication are also major drivers of the global decline of biodiversity and of ecosystem change (Sala et al., 2000; Walther et al., 2002). In addition, NO_3 losses are relevant with regard to drinking water quality. Although NO_3 is viewed less critically with respect to the human health, now (Jenkinson, 2001), low levels in drinking water are stipulated by the EU legislation (EU, 1998). Finally, NH_3 and the acid rain precursor NO_x contribute to acidification (Lerdau et al., 2000; Mosier, 2001) with negative impact on soils, natural ecosystems and buildings.

Quantification of these emissions is needed for a number of purposes, e.g. greenhouse gas and other emission inventories (IPCC, 1996; Lewis et al., 1999), Life Cycle Assessment (Brentrup, 2003), evaluations of sustainability (Addiscott, 1995; Lewandowski et al., 1999; Smith and McDonald, 1998) or scientific information of policy decisions (ECETOC, 1994; Werner et al., 1991). For these purposes, often relatively simple estimation methods are needed, because data availability, time and costs constrain the detail of the investigation. However, the need for simple methods may obviously conflict with the quality of the estimates.

This paper briefly summarizes the factors governing ammonia, nitrous and nitric oxide, nitrate and phosphorus emissions from agricultural fields. Accuracy, suitability and practicability of frequently used estimation methods are then analysed. Finally, we use a field study to test and compare the introduced methods. The paper takes a German perspective, but results may be taken as exemplary for other developed countries in temperate climate.

Qualifying criteria

Simplicity is often a key requirement for methods used in environmental impact studies. However, this should not impair the accuracy, reliability and agronomic soundness of the results. In this section, three qualifying criteria for estimation methods for nutrient

Table 4.1 Share of emissions from agricultural fields to the total anthropogenic emission of ammonia, nitrous and nitric oxide, nitrate and phosphorus.

	World	W. Europe	Germany
NH_3	25[a]	50[b]	45[c]
N_2O	40[a]	55[d]	60[d]
NO_x	<4[a]	n.a.	1[e]
NO_3	n.a.	60[f]	45[g, h]
P	n.a.	35[f]	30[g]

[a] IFA/FAO (2001); Mosier (1998)
[b] Ferm (1998); Bouwman et al. (1997); (Bouwman et al., 2002a)
[c] Döhler et al. (2002); UBA (2002)
[d] Calculated from Freibauer (2002)
[e] Jörß and Handke (2003); [f] Isermann (1990)
[g] Werner (1997); [h] Stanners and Bourdeau (1995)

emissions from agricultural fields are introduced: (1) sensitivity to the environmental and management factors governing the emissions, (2) low method uncertainty and (3) robustness against aggregation biases.

Methods' sensitivity to environmental and management factors

The purpose of environmental impact studies in agriculture is usually to support policy decisions of some kind. To policy makers, it may be attractive to encourage the use of rather simplistic methods in order to obtain quick and cheap results. This may suffice for a coarse first impression of the status quo. For optimising production systems or evaluating trade-offs, however, it is key that the estimation methods used do reflect dif-

Table 4.2 Environmental and management factors regulating nutrient emissions from agricultural fields (adopted from Gäth and Wohlrab, 1994; Bouwman 1995; Bouwman et al., 1997; Werner, 1997; Brentrup et al., 2000; Reynold and Davies, 2001).

Emission	Regulating factors				
	Site properties	Soil properties	Climate/weather	Land use system	Fertilisation
NH_3		- pH and buffer capacity - Infiltration rate - Cation exchange capacity (fixation of NH_4^+)	- Soil and air temperature - Wind speed - Precipitation after application	- Land use (arable, permanent crops, flooding) - Crop type (species)	- N application rate - Kind and form N-fertiliser - Application mode (e.g. broad spreading, trail hose, injection) - Timing of incorporation
N_2O, NO		- Physical factors determining frequency/duration of O_2-deficiency (texture, bulk density, drainage conditions) - Total N and C, C:N ratio - pH	- Temperature - Amount/distribution of precipitation - Freeze-thaw and drying-rewetting cycles	- Land use (arable, grassland/permanent crops, flooding) - Crop type (legumes, cereals, vegetables)	- N application rate - Kind and form N-fertiliser - Timing of input in relation to canopy uptake - (Green-) manuring
NO_3	- Hydrological conditions (distance to water table, bedrock material) - Drainage conditions - Atmospheric N-deposition	- N mineralisation potential - Field capacity in the rooting zone - Infiltration	- Climatic factor governing microbial N mineralisation and N immobilisation - Climatic water balance - Events causing flushes of mineralisation, e.g. drying-rewetting cycles or frost	- Use and management of irrigation - Crop type (N-fixation, N requirements and uptake)	- N application rate in relation to canopy uptake - Timing of input in relation to canopy uptake - Kind and form N-fertiliser
P	- Hydrological conditions (e.g. topsoil thickness, bedrock material) - Topography - Drainage conditions	- Total P contents, fraction of bioavailable P - Water infiltration rate - Soil stability (determined by texture, organic carbon, pH, compaction and other factors)	- Frequency and temporal distribution of precipitation events causing erosion/run-off	- Land use (tillage regime, irrigation management) - Crop specific soil cover period	- Long-term P application rate in relation to removal with harvested produce - Timing of input in relation to erosive precipitation events - Kind and form P-fertiliser used

ferences in different scenarios, production systems or management choices as realistically as possible (Lewis et al., 1999). I.e. estimation methods need to take these factors into account as input variables. Table 4.2 summarises the key environmental and management factors governing the nutrient emissions discussed here.

In the course of this paper, we evaluate a method's sensitivity to environmental and management factors qualitatively by the proportion of relevant factors (as given in Table 4.2) that are accounted for.

Methods' uncertainty

Frequently, data, time and budget for environmental impact studies does not allow for thorough context specific validation of the estimation methods. To still be able to control and communicate the accuracy of the results, stating uncertainty ranges is common in climate and environmental sciences. The uncertainty range is conceptualised as a confidence interval of the estimate, usually the 95% confidence interval, with lower and upper limits at 2.5 and 97.5%, respectively. I.e. it denotes the range within which the true emission will lie with a 95% chance. For normally distributed variables, the upper and lower boundaries of the confidence interval are approximately the (arithmetic) mean plus/minus two times the standard deviation (Köhler et al., 1996; IPCC, 2000).

It is important to note, that such uncertainty estimates are compound entities, encompassing various uncertainty components (Costanza et al., 1992; Funtowicz and Ravetz, 1993): Technical uncertainty stems from the measurement errors of the data on which the estimation method has been validated. This component of uncertainty will normally be comparatively small. Methodological uncertainty enters as either different approaches or inconsistent (but sound) data produce diverging results. This kind of uncertainty tends to be larger, especially when it interacts with the third kind, epistemological uncertainty, stemming from incomplete knowledge of the phenomena under investigation. Ignorance can be substantial in environmental sciences because many problems are novel or highly complex or both (Dovers and Handmer, 1995). Obviously, quantifying this epistemological uncertainty is difficult. In this paper the term 'uncertainty' denotes the combined technical and methodological uncertainty, but does not encompass epistemological uncertainty.

Uncertainties – i.e. the 95% confidence interval – of the methods discussed here have either been taken from the original references, or, if not given there, were estimated according to the IPCC *Good Practice Guidance and Uncertainty Management in National Greenhouse Gas Inventories* (IPCC, 2000). The guideline distinguishes between two tiers of uncertainty estimates: Tier 1 uncertainties are derived by arithmetic error propagation, namely by two rules for addition and multiplication of uncertain quantities:

Addition

$$U_{total} = \frac{\sqrt{(U_1 \cdot x_1)^2 + (U_2 \cdot x_2)^2 + ... + (U_n \cdot x_n)^2}}{x_1 + x_2 + ... + x_n} \quad (4.1)$$

Multiplication

$$U_{total} = \sqrt{U_1{}^2 + U_2{}^2 + ... + U_n{}^2} \quad (4.2)$$

where U_{total} is the percentage uncertainty of the combined quantity and U_i are the percentage uncertainties associated with the quantities x_i. These two error propagation rules are confined to (approximately) normally distributed variables with relatively small uncertainties (standard deviations divided by the mean less than 0.3) and no significant covariance between variables (IPCC, 2000). Tier 2 uncertainties are derived by using stochastic simulation (e.g. Monte Carlo technique) to generate a density distribution for the uncertain quantity.

Note that the uncertainties given here do not yet account for uncertainty of the data used as input. Input data may be highly uncertain because of poor quality of the data itself (sparse measurements, non-representative, literature or estimated data) or simply because of their inherent variability.

Methods' robustness against aggregation biases

Using methods with non-linear input-output relations can produce systematically biased results. This is the case if (1) the input data are arithmetic means (or otherwise aggregated figures) that integrate over a range of individual cases (e.g. data from official statistics); or if (2) the estimates produced themselves are to be further aggregated. This applies in most cases, since the goal of environmental impact studies is usually to determine average long-term effects rather then impacts from a single field or year (Brentrup, 2003; Stockle et al., 1994). I.e. the spatial and temporal resolution needed for communicating results is often coarser than that of the results themselves.

If either of the two above cases applies, only linear input-output relations are unbiased. Non-linear relations cause systematic over or underestimation of the real emissions. To illustrate this, consider the following example: Assume that the emission E of a certain nutrient species is linked to the nutrient input I by the exponential relation $E = 0.1 * EXP(0.05 * I)$. Also assume that I is normally distributed with a mean of 100 and a standard deviation of 30. Using this mean as input to the equation will yield $E \approx 15$.

However, disaggregating the given information we can approximate the frequency distribution of I by assuming that 68% of values of I will lie within the interval of the mean plus/minus one standard deviation, 95% within mean plus/minus two standard deviations and 99% within mean plus/minus three standard deviations (Köhler et al., 1996). Neglecting the remaining one percent of the values, this gives us six classes for *I*, with class means of 5, 55, 85, 115, 145 and 175, respectively. We can now calculate *E* for each of the classes separately: It is 0.3, 1.6, 7.0, 31.4, 141 and 631, respectively. Weighting these results by the class frequencies and summing them up afterwards (frequency weighted mean) yields the total emission, which is 45 – three times the value obtained by using the simple mean. Obviously, it is not adequate to use the simple mean here, as it results in a systematic underestimation of the emission.

Consequently, if the available input data is already aggregated, the choice of unbiased estimation methods is confined to linear relations. The same is true if results are to be further aggregated, although in this case, data stratification (as in the above example) may be used to approximate the real emission. Stochastic simulation is another approach to dealing with such situations. We checked methods introduced here for linearity and, in case non-linear relations are involved, for the goodness of linear approximations (quasi-linearity) and the range in which such approximations hold.

Estimation methods for nutrient losses from agricultural fields

In this section we introduce a number of estimation methods for field losses of NH_3, N_2O, NO NO_3 and P. Detailed method descriptions are given in Boxes 4.1 to 4.5. For easier reference, short names are used as given in Table 4.3. The methods are briefly discussed with regards to the three qualifying criteria given above. Results are summarised in Table 4.4.

Table 4.3 Estimation methods discussed in this study.

Emission type	Method	Short name	Scope	Source	Description
NH_3	EMEP/CORINAIR	EMEP-NH_3	Eur./Germ.	EEA, 2001; Döhler et al., 2002	Box 4.1
	Bouwman et al.	Bouwman-NH_3	Global	IFA/FAO, 2001; Bouwman et al., 2002a	Box 4.1
N_2O (direct)	IPCC	IPCC-N_2O	Global	IPCC, 1996; Mosier et al., 1998	Box 4.2
	Freibauer	Freibauer-N_2O	Europe	Freibauer, 2002	Box 4.2
	Bouwman et al.	Bouwman-N_2O	Global	IFA/FAO, 2001; Bouwman et al., 2002b, c	Box 4.2
N_2O (indirect)	IPCC	–	Global	IPCC1996	Box 4.2
NO	EMEP/CORINAIR	EMEP-NO	Europe	EEA, 2001	Box 4.3
	Bouwman et al.	Bouwman-NO	Global	IFA/FAO, 2001; Bouwman et al., 2002b, c	Box 4.3
NO_3	DBG	DGB-NO_3	Germany	Gäth and Wohlrab, 1994; Brentrup et al., 2000	Box 4.4
P (particulate)	Auerswald et al.	Particulate-P	Germany	Auerswald, 1989; Auerswald & Weigand, 1999	Box 4.5
P (dissolved)	Werner et al.	Dissolved-P	Germany	Werner et al., 1991; Auerswald & Weigand, 1999	Box 4.5

Ammonia (NH_3)

NH_3 escapes when ammonium (NH_4^+) solutions come into contact with ambient air, where NH_3 concentrations are much lower. Volatilisation rates from field applied fertilisers are highest during the first few days after application (Döhler et al, 2002; Sommer and Hutchings, 2001). For detailed reviews on HN_3 volatilisation refer to ECETOC (1994), IFA/FAO (2001) and Sommer and Hutchings (2001).

Estimation methods

Both *EMEP-NH$_3$* and *Bouwman-NH$_3$* are emission factor approaches (refer to Box 4.1). I.e. the NH_3 emission is given as a fixed proportion of the total N applied or the total ammoniacal N (TAN = NH_4-N + NH_3-N) applied, respectively. Yet the methods differ conceptually: *EMEP-NH$_3$* emission factors (Table A.4.1) base on European emission measurements (synthetic fertilisers: Asman, 1992; ECETOC, 1994; Sutton et al., 1995; manures: see Döhler et al., 2002) and on expert judgement to synthesise these data into plausible emission factors. Emission factor sets for organic fertilisers are elaborated in each country individually. The factor set for Germany (Döhler et al., 2002; Tables A.4.2 and A.4.3) is somewhat ambiguous because the emission factor tables were not detailed for all input factor combinations. E.g. the temperature dependence of emissions is only detailed for cattle and pig slurry and only for the application modes broadcast and trailing hose. Other factor combinations have to be extrapolated by the user, which leaves space for individual interpretation.

As opposed to the expert based approach of *EMEP-NH$_3$*, *Bouwman-NH$_3$* follows a strictly statistical approach. The authors first selected relevant factors influencing NH_3

Box 4.1 Estimation methods for NH_3 emissions

EMEP-NH_3 (EEA, 2001; Döhler et al, 2002). Applicable to all N sources listed in Tables 4.A.1–4, Scope: Europe/Germany.

$E_{NH3} = EF_{NH3}$ * N rate * $factor_{N\_FRAC}$

where:

E_{NH3}	: NH_3 emission (kg NH_3-N ha^{-1} yr^{-1}).
EF_{NH3}	: Emission factor for NH_3 (–). Values: Tables 4.A.1 (synthetic fertilisers); 4.A.2 and 4.A.3 (manures).
N rate	: N application rate (kg N ha^{-1} yr^{-1}).
$factor_{N\_FRAC}$	: Factor for N fraction subject to volatilisation (–). Values: 1 for synthetic fertilisers, TAN fraction (Table 4.A.4) for manures.

Bouwman-NH_3 (IFA/FAO, 2001; Bouwman et al., 2002a). Applicable to N fertilisers listed in Table 4.A.5, Scope: World.

$E_{NH3} = EF_{NH3}$ * N rate

where:

E_{NH3}	: NH_3 emission (kg NH_3-N ha^{-1} yr^{-1}).
N rate	: N application rate (kg N ha^{-1} yr^{-1}).
EF_{NH3}	: Emission factor for NH_3 (–). EF_{NH3} = exp [Σ factors], where
Σ factors	: Sum of factor values for crop type, fertiliser type, application mode, soil pH, CEC and climate (–). See Table 4.A.5.

emissions by Residual Maximum Likelihood (REML) analysis from a large dataset of global emission measurements (available at http//:arch.rivm.nl/ieweb/ieweb/databases/). Emission factors were then determined by (REML) regression on the log-transformed values of the relevant variables (IFA/FAO, 2001).

Method sensitivity to regulating factors

The two approaches emphasise different factors to govern NH_3 emissions: *EMEP-NH*$_3$ for synthetic fertilisers reflects only fertiliser type and country group, where the grouping of the countries reflects average soil and climatic conditions within regions. It is most sensitive to fertiliser type. Emission factors for manures are more differentiated and are most sensitive to timing of incorporation, and temperature at application. *Bouwman-NH*$_3$ does not reflect the timing of incorporation and the temperature (although climate is accounted for). It is most sensitive to fertiliser type, mode of application and soil pH. There is no differentiation between different types of manure. Both methods use discrete input factors only and transform continuous variables (such as temperature or soil pH) into classified variables. Neither method take soil infiltration into account, e.g. as influenced by soil texture or surface preparation. This factor has pronounces effects on the NH_3 volatilisation rate from liquid fertiliser (Horlacher and Marschner, 1990; Sommer and Hutchings, 2001).

Note that *EMEP-NH*$_3$ emission factors do not include NH_3 emission from the crop canopy. Holtan-Hartwig and Bøckman (1994, as cited by Ferm, 1998) estimate that canopy emissions in temperate climate are roughly 1.5 kg N ha^{-1} yr^{-1}. They may be higher for fertiliser rates exceeding plant uptake and under stress conditions (Bouwman et al., 1997). *Bouwman-NH*$_3$ is not consistent in terms of canopy emissions: Depending on the measurement techniques used in the original reference, some data points in the underlying dataset did, others did not include canopy emissions (IFA/FAO, 2001).

Method uncertainty

EMEP-NH$_3$ has an uncertainty of ±50% for synthetic fertilisers (EEA, 2001). For organic fertilisers Döhler et al. (2002) do not state uncertainty ranges. We here assume that the total uncertainty is similar to that of emission factors for synthetic fertilisers (±50%), which is also consistent with results Sutton et al. (1995) obtained for emission factors in the UK. The average uncertainties of the *Bouwman-NH*$_3$ are: ±35% for synthetic fertilisers (with some individual fertilisers having much wider range; see original reference for details) and –17% and +26% for manures.

Robustness against aggregation biases

Both methods are emission factor methods and thus respond linearly to the N application rate. *EMEP-NH$_3$* for organic fertilisers takes two continuous (but classified) variables as input: temperature at application and time of incorporation. For both variables the response of the emission factor is non-linear. The response to temperature is approximately linear within a range of three classes. Data on temperature as an input factor are relatively easy to obtain. In contrast to that the emission factor is highly sensitive to changes in the time of incorporation (e.g. the emission factor after 4 hours is three times higher than after one hour). Although taking timing of incorporation into account is sensible from an agronomic point of view, it impairs the practicability of the method severely, because this information will seldom be retrievable with sufficient accuracy.

The *Bouwman-NH$_3$* emission factor is an exponential function of the sum of factor class values for crop type, fertiliser type, application mode, soil pH, soil cation exchange capacity and climate. Since the factor classes for continuous variables are broad, they are relatively robust against variability within the input data. If input data spans more than one class (e.g. CECs ranging from 12 to 20 cmol kg^{-1}) or different factor values for discrete factors apply (e.g. two different types of fertilisers), emission factors should be computed individually for each factor class combination.

Nitrous oxide (N_2O)

N_2O from fertilised agriculture occur on two occasions: directly on field application of N-fertilisers (direct emissions) and later, on re-deposition and downstream cycling of reactive N compounds that have previously been lost from the field system (indirect emissions). The total N_2O emission is the sum of direct and indirect emission, the uncertainty of the total emission can be computed by tier 1 (error propagation) as give above.

Direct N_2O emissions

Direct N_2O emissions occur when N_2O is formed as a free gaseous intermediate product during both nitrification and denitrification. A third process forming N_2O is the assimilatory reduction of NO_3 to NH_4 under high levels of available C and consequent N limitation (Rochette et al, 2000). These processes can take place simultaneously, although at different micro-sites (von Rheinbaben, 1990; Jenkinson, 2001). In fertilised arable soils, denitrification is generally presumed to be the prevalent source of N_2O emissions from soils. For reviews see Brentrup et al. (2000), IFA/FAO (2001) and Freibauer and Kaltschmitt (2002).

Estimation methods

Methods for N_2O emissions are described in Box 4.2. *IPCC-N_2O* has been developed as a global method for national greenhouse gas inventories. It is based on a linear regression model originally developed by Bouwman (1995; not to be mistaken for the Bouwman et al. approach presented here) and later amended (Mosier et al., 1998). *Freibauer-N_2O* has been developed to differentiate the global IPCC approach for European countries. Both *IPCC-N_2O* and *Freibauer-N_2O* are based on linear regression analysis. (The underlying dataset for the IPCC approach is published in Bouwman, 1995. The Freibauer dataset is unpublished.) The *Bouwman-N_2O* approach follows the same structure as *Bouwman-NH_3*, using REML analysis (IFA/FAO, 2001) of a large dataset of published measurements (available at http//:arch.rivm.nl/ieweb/ieweb/databases/). While *IPCC-N_2O* and *Freibauer-N_2O* are applicable to all types of N input, *Bouwman-N_2O* covers synthetic fertilisers and manures only and is not valid for returned harvest residues, legume N or green manure.

Method sensitivity to regulating factors

IPCC-N_2O is mainly sensitive to fertiliser type, where organic soils are cultivated this is reflected as well. Indeed, the method is most sensitive to the differentiation of soils (mineral vs. organic) for N application rates less 300 kg ha^{-1} yr^{-1}. The same is true for *Freibauer-N_2O*, although it accounts for some site properties. This suggests that the primary way of managing N_2O emissions is via the application rate.

The *Bouwman-N_2O* is more detailed. It is most sensitive to crop type, soil organic carbon and climate. The N application rate and fertiliser type show a pronounced influence for application rates > 200 kg N ha^{-1} yr^{-1}. Criticism of *Bouwman-N_2O* could relate to the fact the factor values for some very similar fertilisers diverge largely, although N_2O emissions should be similar as well. E.g. the value for ammonium nitrate (AN) is nearly twice as high as that for calcium ammonium nitrate (CAN). Bouwman et al. suggest that this could be due to differences in the pH chemistry, but unaccounted differences in experimental conditions seem to be another likely explanation: AN is predominantly used in developing, CAN in developed countries and hardly any study does compare both fertilisers within the same experiment. The difference between the factor value may thus largely be an expression of different climatic, soil or management conditions rather than of differences in the chemistry of the fertilisers.

Freibauer-N_2O is the only of the three methods that explicitly accounts for the freeze-thaw cycles in some geographic regions. Freeze-thaw cycles are an important driver of winter emissions (Wagner-Riddle and Thurtell, 1998), which may contribute 50% or more of the annual N_2O emissions (Kaiser and Ruser, 2000).

Box 4.2 Estimation methods for N_2O emissions

Total N_2O emission = direct + indirect N_2O emission

A. Direct N_2O emission

IPCC-N_2O (IPCC, 1996). Applicable to all N inputs. Scope: World.

$E_{N2O\_DIR} = 1 + [0.0125 * \text{N rate} * (1 – \text{factor}_{NH3\_VOLAT})] + \text{emission}_{HIST}$

where:

E_{N2O_DIR}	: Direct N_2O emission (kg N_2O-N ha^{-1} yr^{-1}).
N rate	: N application rate (kg N ha^{-1} yr^{-1}).
$\text{factor}_{NH3\_VOLAT}$	: Subtraction for NH_3 volatilisation (–). Default values: 0.1 for synthetic, 0.2 for organic fertilisers.
emission_{HIST}	: Additional emission for the cultivation of organic soils (histosols) (kg N_2O-N ha^{-1} yr^{-1}). Default values: 0 for mineral soils, 5 for organic soils (10 in tropical climate).

Freibauer-N_2O (Freibauer, 2002). Applicable to all N inputs. Scope: Europe.

A. Arable crops in temperate oceanic and Mediterranean climate (Eastern/Southern Austria, Belgium, Denmark, France, Germany North of 49°N, Greece, Luxembourg, Ireland, Italy, The Netherlands, Portugal, Spain, South of Sweden, UK):

$E_{N2O\_DIR} = 0.6 + 0.002 * \text{N rate} + 12.7 * \text{soil C} – 0.24 * \text{sand} + \text{emission}_{HIST}$

B. Arable crops in (pre-) alpine and sub-boreal climate (Northern/Western Austria, Finland, Germany South of 49°N, Sweden except South, Switzerland):

$E_{N2O\_DIR} = –1.3 + 0.03 * \text{N rate} + 280 * \text{soil N} + \text{emission}_{HIST}$

C. Grassland in temperate and sub-boreal climate

$E_{N2O\_DIR} = 2.4 + 0.015 * \text{N rate}$

where:

E_{N2O_DIR}	: Direct N_2O emission (kg N_2O-N ha^{-1} yr^{-1}).
N rate	: N application rate (kg N ha^{-1} yr^{-1}). Valid range: 0 – 500.
soil C	: Organic carbon content of topsoil (g g^{-1}). Valid range: 0.005 – 0.082.
sand	: Sand content of topsoil (g g^{-1}). Valid range: 0.015 – 0.857.
soil N	: Total N content of topsoil (g g^{-1}). Valid range: 0.0007 – 0.0025.
emission_{HIST}	: Fixed emission factor for the cultivation of organic soils (histosols) (kg N_2O-N ha^{-1} yr^{-1}). Values: for mineral soils = 0; for histosols = 7 under grassland & cereals and = 10 under vegetables & root crops.

Bouwman-N_2O (IFA/FAO, 2001; Bouwman et al., 2002b). Applicable to N fertilisers listed in Table 4.A.5. Scope: World.

$E_{N2O\_DIR} = \exp[0.4114 + (\text{factor}_{TYPE*RATE} * \text{N rate}) + \Sigma \text{ factors}]$

where:

E_{N2O_DIR}	: Direct N_2O emission (kg N_2O-N ha^{-1} yr^{-1}).
N rate	: N application rate (kg N ha^{-1} yr^{-1}).
$\text{factor}_{TYPE*RATE}$	: Fertiliser specific emission factor (–). See table 4.A.5.
Σ factors	: Sum of factor values for crop type, soil texture, soil organic carbon content, soil pH, soil drainage conditions and climate (–). See table 4.A.5.

B. Indirect N_2O emission

IPCC (IPCC, 1996). Applicable to reduced and oxidised N (=reactive N) inputs to surface water. Scope: World.

$E_{N2O\_INDIR} = 0.01 * [NH_3 + \text{NO emission}] + 0.025 * \text{N leached}$

where:

E_{N2O_INDIR}	: Indirect N_2O emission (kg N_2O-N ha^{-1} yr^{-1}).
NH_3 + NO emission	: NH_3 and NO emissions to atmosphere from fertiliser use (kg N ha^{-1} yr^{-1}) Default value: 0.1 * synthetic N applied + 0.2 * organic N applied.
N leached	: N lost via leaching and run-off (kg N ha^{-1} yr^{-1}). Default value: 0.3 * N applied.

Method uncertainty

Uncertainties are highest for *IPCC-N_2O* (±70 to 80%; IPCC, 1996). For *Bouwman-N_2O* they range from –40% and +70% (Bouwman et al., 2002b). *Freibauer-N_2O* has an uncertainty range of ±41% and ±65% for arable land (Equations A. and B. in Box 4.2) <±100% for grassland (Equation C.) and <±50% for the cultivation of histosols (for furthe detail refer to Freibauer, 2002).

Robustness against aggregation biases

IPCC-N_2O and *Freibauer-N_2O* are linear and thus robust against aggregation biases. *Bouwman-N_2O* is an exponential function of (a) the product of the N application rate times a fertiliser specific factor and (b) the sum of factor class values for crop type, soil texture, organic carbon, drainage conditions and climate. The relation between N_2O emission and N application rate is approximately linear (r^2 of >0.97) within any 200 kg N ha^{-1} yr^{-1} window. This means that biases due to aggregation or averaged input data will be very small up to a spread of ±100 kg N ha^{-1} yr^{-1} around the mean. The influence of the sum of factor class values is exponential. Below a sum of factor class values of –0.8 it may also be regarded linear. For values above –0.8 emissions should be computed individually for each factor combination and be aggregated afterwards (frequency-weighted mean).

Indirect N_2O emissions

Indirect N_2O emissions from agriculture occur when reactive nitrogen compounds (NH_3, NO_x, NO_3) lost from fields are re-deposited on soils or cycled in downstream ecosystems. Re-deposited N compounds are assumed to have the same effects on soil N_2O emissions as synthetic fertiliser. N compounds in aquatic ecosystems are subject to denitrification in groundwater, rivers, estuaries and costal marine ecosystems, during which N_2O is produced (Seitzinger and Kroeze, 1998). For reviews on indirect N_2O emission see IPCC (1996), Mosier et al. (1998) and Seitzinger and Kroeze (1998).

The IPCC (1996) proposes a simple emission factor approach based on the analysis of published emission figures from different terrestrial and aquatic environments. Originally, NH_3, NO and NO_3 emission estimates, which are required as input, are computed with the EMEP and IPCC methods or default values, respectively. However, any other method may be eligible as well.

Knowledge on regulating factors and their effects is still expanding and the approach is not yet very detailed. The high method uncertainties reflect this: They are –80% / +100% for the emission caused by N re-deposition and –75% / +380% for the emission caused by N losses to water. The method is linear and robust against aggregation biases, if default values for gaseous and leaching losses are used.

Nitric oxide (NO)

NO_x from agricultural soils is mainly emitted as NO that, on entering the atmosphere, rapidly reacts with O_3 to form NO_2 (Lerdau et al., 2000; Mosier, 2001). Like N_2O, NO is formed as a free intermediate product during nitrification and denitrification (Delmas et al., 1997), although typically nitrification appears to be the dominant source in arable soils in temperate climate with a soil pH above 5 (Skiba et al., 1997). Consequently, factors enhancing mineralisation, e.g. tillage or manuring, have a pronounced effect on NO emissions. For a review on NO emissions we refer to Skiba et al. (1997). Compared to NH_3 and N_2O, measurement data on NO emissions is scant and the available data is extremely variable.

Estimation methods

EMEP-NO is a simple emission factor approach with a fixed emission factor. It is a somewhat preliminary method to estimate national emissions within international monitoring frameworks and mainly draws on previous work by Yienger and Levy (1995), Skiba et al. (1997) and Veldkamp and Keller (1997). *Bouwman-NO* bases on the same methodological approach as the Bouwman et al. NH_3 and N_2O models (IFA/FAO, 2001), although the available dataset for NO was considerably smaller than for the other two methods (99 data points; available at http//:arch.rivm.nl/ieweb/ieweb/databases/). *Bouwman-NO* covers synthetic fertilisers and animal manures only and is not valid for returned harvest residues, legume N or green manure. Both methods are described in Box 4.3.

Method sensitivity to regulating factors

EMEP-NO suggests that the only way to reduce NO emissions is reduced N input. It is

Box 4.3 Estimation methods for NO emissions

EMEP-NO (EEA, 2001). Applicable to all N inputs. Scope: Europe.

$$E_{NO} = 0.003 * \text{N rate}$$

where:

E_{NO} : NO emission (kg NO-N ha^{-1} yr^{-1}).

N rate : N application rate (kg N ha^{-1} yr^{-1}).

Bouwman-NO (IFA/FAO, 2001; Bouwman et al., 2002b). Applicable to fertilisers listed in table 4.A.5. Scope: World.

$$E_{NO} = \exp[\,-1.527 + (\text{factor}_{TYPE*RATE} * \text{N rate}) + \Sigma \text{ factors}\,]$$

where:

E_{N0} : NO emission (kg NO-N ha^{-1} yr^{-1}).

N rate : N application rate (kg N ha^{-1} yr^{-1}).

$\text{factor}_{TYPE*RATE}$: Fertiliser specific emission factor (–). See table 4.A.5.

Σ factors : Sum of factor values for soil organic carbon content and soil drainage conditions (–). See table 4.A.5.

thus not very suitable for environmental impact studies that aim at optimising management. *Bouwman-NO* is more detailed. It is most sensitive to soil organic carbon and drainage conditions. Fertiliser type and application rate have pronounced effects at rates > 200 kg N ha^{-1} yr^{-1}. Neither method takes climate or land use into account. This shortcoming may, however, be tolerable when considering the small share agriculture contributes to the total anthropogenic NO_x emission (see Table 4.1).

Method uncertainty

For *EMEP-NO* the authors assume a very large uncertainty, spanning one order of magnitude (EEA, 2001). Taking into account more recent estimates, however, it is likely that the uncertainty is less and ranges within ±250%. Bouwman et al. (2002b) do not specify an uncertainty range because the number of complete records in their database was too small to estimate the uncertainty. It is, however, likely that the uncertainty will be less than that of *EMEP-NO* since the number of data points as well as the number of model parameters is higher.

Robustness against aggregation biases

EMEP-NO is linear and hence robust against aggregation biases. *Bouwman-N_2O* is an exponential function of (a) the product of the N application rate times a fertiliser specific factor and (b) the sum of input factor class values for soil organic carbon and drainage conditions. Similar to *Bouwman-N_2O*, the relation between NO emission and N application rate is approximately linear (r^2 of >0.95) within any 200 kg N ha^{-1} yr^{-1} window. The influence of the sum of factor class values is exponential. Below 1.8 it may also be regarded linear. At values above 1.8 emissions should be computed individually for each factor combination and be aggregated afterwards (frequency-weighted mean).

Nitrate (NO_3)

NO_3 is mainly lost via downward and lateral fluxes of soil water, containing solved NO_3 (Schachtschabel et al., 1984). These are highly variable throughout space and time and dependent on a large number of environmental and management factors. For a detailed description of nitrate loss pathways see Haag and Kaupenjohann (2000).

Estimation method

We here introduce a method suggested by Brentrup et al. (2000), amending an approach of the German Soil Science Society (DBG; Gäth and Wohlrab, 1994). It consists of a mass balance for soil N in combination with a simplified soil water balance (Box 4.4).

Method sensitivity to regulating factors

DBG-NO_3 relates to most regulating factors indirectly, via the N-balance surplus. Some factors taken into account are difficult to retrieve

- Site-specific data on N-mineralisation and N-immobilisation/NH_4 fixation is seldom available. It is often assumed that in- and effluxes to the soil N pool are balanced and compensate each other (Brentrup et al., 2000). This is a useful assumption in the long run and if management is constant. However it will not hold in case of management changes or massive disturbance of the soil C and N dynamics like land conversion or land melioration. In such cases, both significant influxes as well as effluxes to the soil N pool may occur (Haag and Kaupenjohann, 2000; Nieder and Richter, 2000).
- Data on N deposition will usually be taken from the literature, or, for European countries having joint the Convention on Long-Range Transboundary Air Pollution, from the EMEP database (available at http://www.emep.int/index_pollutants.html). Note that recent studies indicate that gaseous deposition and foliar plant uptake, which are not detected by the regular deposition measurements ('bulk deposition') may be significant. In areas with intensive agricultural production, the total atmospheric N input thus could reach orders of 50 to 60 kg N ha^{-1} yr^{-1} (Ferm, 1998; Jenkinson, 2001; Weigel et al., 2000), which is about twice the bulk deposition rates reported normally.
- The gaseous N losses consist of NH_3 volatilisation and denitrification losses. We introduced methods to estimate NH_3 volatilisation. Denitrification losses include NO

Box 4.4 Estimation methods for NO_3 emissions

DGB-NO_3 (Gäth and Wohlrab, 1994; Brentrup et al., 2000). Scope: Germany.

E_{NO3} = N balance * $factor_{EX\_FREQ}$

where:

E_{NO3}	: NO_3 emission (kg NO_3-N ha^{-1} yr^{-1}).
N balance	: N balance surplus (kg N ha^{-1} yr^{-1}). N balance = [Σ N inputs – Σ N outputs] if Σ N inputs > Σ N outputs, else N balance = 0.
Σ N inputs	: N fertilisation + N mineralisation + N deposition + symbiotic N fixation + N in seeds (all kg N ha^{-1} yr^{-1}).
Σ N outputs	: Gaseous N losses + N immobilisation and NH_4 fixation + N output with product (all kg N ha^{-1} yr^{-1}).
$factor_{EX\_FREQ}$	: Flow weighting factor for the soil water exchange frequency (–). $factor_{EX\_FREQ}$ = min{[drainage / field capacity]; 1}.
drainage	: Drainage water leaching from the rooting zone (mm yr^{-1}). For Germany: drainage = 0.86 * P_A – 111.6 * [(P_A – P_W) / P_W] – 241, where P_A is the annual and P_W the winter (01 Oct – 31 Mar) precipitation (both in mm yr^{-1}).
field capacity	: Effective field capacity in the rooting zone (mm). (Standard values for different soil types given in Gäth and Wohlrab, 1994 and Hennings, 1994).

and N_2O but mainly consist of molecular N (N_2). N_2 is environmentally not relevant and therefore not within the focus of estimation method development. Following von Rheinbaben (1990), Brentrup et al. (2000) suggest to account for denitrification losses with 10% of the N application rate. For this paper we re-evaluated the data collated by von Rheinbaben (1990). It encompasses experiments lasting from one week to over one year. Since the duration of denitrification measurements has a pronounced effect on N_2O and NO losses (Bouwman, 1995; Bouwman et al., 2002b), we assumed the same was true for total denitrification. Hence we excluded measurements covering less than one year from the data set, reducing it to 22 data points for fertilised and 10 data points for unfertilised plots. Distributions of both sets are skewed towards lower values. The mean emission of the fertilised plots is 15% of the N applied, the median is 10%. Using log-transformed data to decrease the influence of extreme values, least square fitting and back-transformation yields the following equation:

$$N_{\text{denitrification}} = \exp(1.38 + N_{\text{applied}} \times 0.006) \approx 4 \times \exp(N_{\text{applied}} \times 0.006) \tag{4.3}$$

(n=22, r^2=0.43)

where $N_{denitrification}$ (kg N ha^{-1} yr^{-1}) is the total denitrification loss and $N_{applied}$ is the fertilisation rate (kg N ha^{-1} yr^{-1}). The model is valid from 50 to 600 kg N ha^{-1} yr^{-1}. The intercept of 4 kg N ha^{-1} yr^{-1} is consistent with the data from the 10 unfertilised plots, which have a mean of 5.3 kg N ha^{-1} yr^{-1} and a median of 2.5 kg N ha^{-1} yr^{-1}.

Method uncertainty

We used a tier 2 approach (stochastic simulation) to estimate the uncertainty of the N-balance surplus in *DBG-NO*$_3$, which is –90% and +120%. This is in line with empirical data for sandy soils and findings from field applications of the method (Bouwer et al., 1997). Note, that the method is less accurate for heavy soils (Stefan Gäth, University of Gießen, 2003; personal communication). Likewise, the uncertainty may be slightly higher if the soil water exchange frequency is below 1.

Robustness against aggregation biases

In its original form, *DBG-NO*$_3$ is a linear function of N inputs and outputs. Equation 4.3 for total denitrification losses is exponential. However, *DBG-NO*$_3$ plotted against denitrification is approximately linear within any 200 kg N ha^{-1} yr^{-1} window (r^2>0.96). Similarly there are non-linear responses to winter precipitation and field capacity if drainage rates do not exceed the field capacity. The response to winter precipitation is approximately linear within any 200 mm yr^{-1} window for values greater 200 mm yr^{-1}. The response to field capacity may be approximated linearly in the range between 40 mm and drainage rate.

Phosphorus (P)

From an agronomic point of view, the amounts of P lost from fields are often negligible. They are, however, highly relevant from an environmental perspective. P losses from agricultural fields occur mainly in particulate form (i.e. through soil erosion) and through P dissolved in surface and surface near run-off water (Reynolds and Davies, 2001). Leaching is generally not considered a major pathway for P losses, since P is quickly adsorbed in soils. However, P leaching has recently been reported for heavily over-fertilised sites in the UK (Addiscott and Thomas, 2000) and Germany (Jan Siemens, University of Hohenheim, 2001; personal communication). For a detailed review on P losses from fields we refer to Reynolds and Davies (2001).

Estimation methods

We here introduce separate methods for particulate and dissolved P (Box 4.5). The total P loss is the sum of both pathways, its uncertainty can be computed by tier 1 (error

Box 4.5 Estimation methods for P emissions

Total P emission = particulate + dissolved P emission

A. Particulate P (Auerswald, 1989; Auerswald and Weigand, 1999). Scope: Germany.

E_{P_PART} = SL * ER * total P * SDR

where:

E_{P_PART}	: Emission of particulate P (kg P ha^{-1} yr^{-1}).
SL	: Soil loss through water erosion (t dry soil ha^{-1} yr^{-1}). Estimate with standard methods, e.g. RUSLE (Renard et al., 1997) or ABAG (Schwertmann et al., 1987; Hennings, 1994)
ER	: Nutrient enrichment ratio for P (–). ER = 2.53 * SL $^{-0.21}$.
total P	: Total soil P content (g P kg^{-1}). Expressed as a function of CAL-P (g kg^{-1}) [a]: total P = 0.6 + 5.9 * CAL-P.
SDR	: Sediment delivery ratio to account for sedimentation between field border and watercourse (–). For Germany expressed as a function of watershed size (km^2): SDR = –0.02 + 0.385 * [watershed size] $^{-0.2}$. Set SDR = 1 to calculate total amount of P leaving field.

B. Dissolved P (Werner et al., 1991). Scope: Germany.

E_{P_DISS} = run-off * solved P * RDR * 0.01

where:

E_{P_DISS}	: Emission of dissolved P (kg P ha^{-1} yr^{-1}).
run-off	: Run-off volume (mm yr^{-1}). Expressed as a function of annual drainage (Hennings, 2000): run-off = drainage (mm yr^{-1}) * $2*10^{-6}$ * $(P_A\text{-}500)^{1.65}$, where drainage = 0.86 * P_A – 111.6 * [(P_A – P_W) / P_W] – 241. P_A is the annual and P_W the winter (01 Oct – 31 Mar) precipitation (both in mm yr^{-1}).
solved P	: P concentration in run-off water (mg P l^{-1}). Expressed as a function of CAL-P (g kg^{-1}) [a]: solved P = 0.1 * exp[6.9 * √(CAL-P)] (Auerswald & Weigand, 1999).
RDR	: Run-off delivery ratio to account for water retention between field border and water course (–). Default value: 0.5. To calculate total amount of P leaving field Set RDR = 1.

[a] Plant available P after Schüller (1969) (calcium acetate extraction).

propagation rules) as give above. Both methods are conceptually mechanistic and combine a soil P concentration estimate with a transportation rate estimate.

Particulate-P calculates P losses from long-term (>25 years) average soil erosion rates, as computed with the RUSLE (Renard et al., 1997) or the ABAG, its German analogue (Schwertmann et al., 1987; Hennings, 1994). It accounts for soil P contents and for its enrichment in the eroded material due to selective transportation and sedimentation. *Dissolved-P* calculates P losses analogously from the run-off volume. Both methods were originally developed for estimating P delivery to watercourses on the landscape scale. Thus they contain a reduction factor (SDR and RDR, respectively) to account for sediment and water retention on the way from field to the watercourse. To calculate the amount of P *leaving the field* these factors should be set to one.

Method sensitivity to regulating factors

Particulate-P reflects all relevant factors fairly well. It is important to note that it does not account for the different bio-availability of different P fractions within the eroded material (cf. Reynolds and Davies, 2001). The run-off component of *Dissolved-P* is not sensitive to site properties, which play an important role in determining the run-off volume (Rode, 1995). The same is true for field management that can quite effectively control run-off losses, e.g. through soil cover, good soil structure or establishment of vegetated filter strips. Estimating run-off with more detailed methods, e.g. the USDA-SCS (now USDA-NRCS) Curve Number Method (CN-method, www.nrcs.usda.gov), could mitigate this weakness. The CN-method is also frequently used outside the US, Bach et al. (2000) describe a version adapted to German conditions. Neither of the two partial methods reflects the risk of freshly applied manure being washed off by rain.

Method uncertainty

We used tier 2 to estimate the uncertainty (±30%) for the nutrient enrichment (ER_P) in *Particulate-P*. The uncertainty of erosion estimates (±30%) was taken from Schwertmann et al. (1987). Assuming a fixed sediment delivery ratio (SDR), the total method uncertainty for *Particulate-P* is ±45% (computed by tier 1).

The uncertainty of *Dissolved-P*, calculated by tier 1, is ±110%. This is assuming: ±50% for estimating the P concentration in the run-off (P_{sol}); and ±100% for the run-off volume. The run-off delivery ratio (RDR) was assumed to be fixed. The uncertainty estimates for both methods do not yet encompass *model* uncertainty as such, i.e. the fact that the model may yield inaccurate estimates is not covered. This uncertainty component could only be determined by testing the methods against a large data set, which was not available. The full method uncertainties will thus be larger than the above figure.

Robustness against aggregation biases

Particulate-P is a linear combination of four factors. The enrichment ratio (ER_P) is a logarithmic function of soil loss but it can be approximated linearly (r^2>0.99). The sediment delivery ratio (SDR) is, for single fields, approximately one. For catchments greater than 1 km² it is approximately linear. Below that, data should thus be stratified if watershed sizes differ by more than more than one order of magnitude. Note that estimates of soil loss (SL) with RUSLE or ABAG are highly sensitive to differences in input factors (Auerswald, 1987).

Dissolved-P responds non-linearly to soil P content (P_{CAL}) and to annual and winter precipitation (P_A and P_W). It may however be approximated linearly for P_{CAL} within any 0.2 g kg^{-1} window (r^2 >0.97). Alternatively, data stratification and computation of frequency-weighted means (see introduction) avoids aggregation biases. This is suitable if input data are already aggregated and given in the format of soil P classes, as e.g. in official German statistics. For P_A linear approximation holds within any 300 mm window if $P_A \geq 600$ mm and $P_W \geq 300$ mm. P_W may be approximated linearly within any 200 mm window if $P_W \geq 200$ mm

Synopsis of estimation methods

An overview of the methods and their performance is given in Table 4.4. Simple emission factor methods do not sufficiently account for the effects of different management practices. They suggest that the only way to decrease emission is to reduce the nutrient application rates. But also more detailed methods tend to account for environmental factors and site properties rather than for differences in management. For the task of optimising management with regard to nutrient losses it would be desirable to have methods with a stronger focus on management.

Generally, the set of methods proposed by Bouwman et al. to estimate gaseous losses of NH_3, N_2O and NO are more management sensitive than others. In some points these methods appear agronomically less informed than their expert-knowledge based counterparts. However, the sheer mass of measurement data they ground on may well compensate for this shortcoming. Also it is important that mechanistic mapping of real processes is not the prime objective of the Bouwman et al. methods. They are simple *statistical* models to make robust predictions on the basis of empirical data describing today's fertilisation practices.

A general shortcoming with all of the introduced methods for gaseous N losses is that they are not explicit about non-manure organic fertilises, such as composts, castor oil cake, colza meal, horn meal, crushed pulse seeds or Neem cake. Experimental data on these emissions are sparse and it remains unclear, whether methods do cover them sufficiently or not. Non-manure organic fertilises are frequently used in organic horticulture in Europe as well as in developing countries. Because of the high public interest in comparing conventional and organic systems and the importance of recycling and using local nutrient sources, this gap should be closed.

Interestingly, the methods for the gaseous emissions are less uncertain than the ones for non-gaseous emissions. This may be mere coincidence but could have methodological reasons as well: The methods for losses to water are conceptually mechanistic, i.e. they attempt to mimic the actual processes that lead to the nutrient losses by multiplying a flow component with a concentration component. We suppose that this will add extra uncertainty, stemming from the inevitable simplicity of the model. Conversely, the methods for gaseous losses are empirical heuristics (black box models), obtained by an 'informed fitting exercise' and expert judgement. Although this has less explanatory power, it may be more efficient in obtaining reliable predictions.

In all of the introduced methods aggregation biases can be controlled and reduced to a negligible level. However, all methods except simple emission factor approaches will produce biased results if applied to averaged data or if results were further aggregated. Therefore the implications of non-linear relations must be checked carefully before applying such methods.

Case study: Spinach production in Borken

We applied the introduced methods to estimate nutrient losses from a spinach and a maize production system in the County of Borken, Northwest Germany (Table 4.5). The area is characterised by intensive agriculture with high livestock densities (indoor keeping of mainly pigs and cattle). Maize is the dominant crop in rotations (40 – 50%). Growing vegetables (mainly spinach) for a frozen foods factory that operates in the area since the 1960s allows farmers to diversify both rotations and income sources.

Spinach is a crop with high nutrient demand and high residual N after the harvest. Since there is a risk of microbial contamination from animal manures, spinach receives synthetic N only. This causes additional N input to the regional agricultural system, which is already characterised by high N and P surpluses from livestock keeping. Spinach does usually not receive P fertilisation, because manuring keeps soils high in P. However, the crop provides little soil cover, making fields susceptible for run-off and erosion. Spinach cropping is thus likely to have environmental implications. We here compare a standard spinach production system, in which spinach is cropped twice to the same field every four years, with a standard maize system. We used long-term average data (given

Table 4.4 Performance of emission estimation methods: Sensitivity to environmental and management factors, uncertainty and robustness against aggregation biases. Lower uncertainty limits have been cut at –100% to avoid negative emission values. "Data stratification" denotes separate calculation of emissions for different input class combinations.

Estimation method	Sensitivity to regulating factors[b]						Uncertainty		Robustness against aggregation bias[c]
	Site properties	Soil properties	Climate/ weather	Land use system	Fertilisation	Total	Lower limit	Upper limit	
$EMEP-NH_3$ (synthetic)	[a]	+	+	–	++	low	–50	50	Yes
(organic)	[a]	–	+	++	+++	medium	–50	50	Limited [1]
$Bouwman-NH_3$	[a]	++	+	+++	+++	high	–15-35	25-35	Limited [2]
$IPCC-N_2O$	[a]	–	+	++	+	low	–80	80	Yes
$Freibauer-N_2O$	[a]	+	+++	+++	+	medium	–15-65 [d]	30-65 [d]	Yes
$Bouwman-N_2O$	[a]	+++	+	++	++	medium	–40	70	Limited [3]
EMEP-NO	[a]	–	–	–	+	low	–100	250	Yes
Bouwman-NO	[a]	++	–	–	++	low	–100	<250	Limited [4]
$DBG-NO_3$	+	+	+	+++	++	medium	–90	120	Limited [5]
Particulate-P	++	+++	+++	+++	+	high	–100	110	Limited [6]
Dissolved-P	–	+	+++	–	+	low	–100	100	Limited [7]

[a] Not relevant.

[b] Regulating factors as given in table 2. Symbols: – = not sensitive; +/"low" = sensitive to ≤ ⅓ of management and environmental factors; ++/"medium" = sensitive to ≤ ⅔; +++/"high" = sensitive to > ⅔.

[c] See text for details.

[d] Uncertainty for grassland up to ±100 %.

[1] Stratify input data by incorporation times. Temperatures may be averaged if they spread over three classes or less. Else stratify.

[2] Stratify input data if different factor classes for crop type, fertiliser type, application mode soil pH, CEC or climate apply.

[3] Quasi-linear for N application rates within any 200 kg N ha-1 yr-1 window. Quasi-linear if the sum of factor class values (including the fixed model factor) is <–0.8. Else stratify.

[4] Quasi-linear for N application rates within any 200 kg N ha-1 yr-1 window. Quasi-linear if the sum of factor class values (including the fixed model factor) is < 1.8. Else stratify.

[5] Denitrification rates: if computed after Equation 3 then quasi-linear for N application rates within any 200 kg N ha^{-1} yr^{-1} window, else linear. Winter precipitation: quasi-linear within any 200mm window for values ≥200mm. F and drainage rate, constant for values greater drainage rate. Outside these ranges stratify.

[6] Quasi-linear for watershed sizes > 1km². Below that stratify.

[7] Plant available P: quasi-linear within any 0.2 g P kg^{-1} window. Annual precipitation: quasi-linear within 300 mm window if annual precipitation ≥ 600 mm and winter precipitation ≥ 300 mm. Winter precipitation: Winter preci values ≥ 200 mm. Outside these ranges stratify.

in Table 4.5). Emission estimates were calculated according to the method description in Boxes 4.1 to 4.5.

Table 4.6 shows the emission estimates from the two systems. NH_3 is the only emission where there are pronounced differences between crops and almost none between methods. *Bouwman-NH_3* has slightly higher certainty than *EMEP-NH_3*. As opposed to that, direct N_2O emission estimates differ largely between methods (factor 3 – 4) but little between crops. *Freibauer-N_2O* produces the lowest estimates. Results from *IPCC- N_2O* and *Bouwman-N_2O* are almost identical. The differences between crops are small compared to those between methods. Indirect N_2O emissions are in the same order of magnitude as direct ones. The total N_2O emission (direct plus indirect) ranges from 2.7 to 5.8 kg N_2O-N ha^{-1} in the spinach system and from 2.6 to 4.7 kg N_2O-N ha^{-1} in the maize system, depending on method choice.

NO emission estimates differ strongly between methods (by nearly factor 3) but not between crops. Likewise, there are only small differences in the P emissions of the two systems. *Dissolved-P* does not differentiate between the two systems, a weakness that is

Table 4.5 Site and management details for the case study production systems in County of Borken, Northwest Germany.

Climate		Soils	
Annual precipitation	760mm	Texture	sandy(78% sand)
Winter precipitation (01 Oct–31 Mar)	395mm	pH	5.5–6.5
Mean temperature	9.7°C	Organic carbon	1.7%
		CEC (cmol kg^{-1})	12–16
		Drainage conditions	good
N deposition	20kg N ha^{-1} yr^{-1}	Plant available P [a]	0.135g P kg^{-1}

	Spinach	Maize
Management		
Sowing	Mar–Aug (twice)	Mid-May
Tillage	conventional	conventional
Irrigation	yes	no
N fertilisation (kg N ha^{-1} yr^{-1})	108UAN (sprayed) 94CAN (broadcast)	25liquid NP (injected) 144mixed slurry (brdc.&inc.) [b]
Soil erosion (t dry soil ha^{-1} yr^{-1})	1.1	1.3
Soil water exchange frequency (yr–1)	>2.0	>1.0
Yield		
Fresh product (t ha^{-1} yr^{-1})	42	10
N off-take (kg N ha^{-1} yr^{-1})	131	105

[a] CAL-P after Schüller (1969). Value shown is frequency weighted mean of soil P contents in Borken from Lammers (1999).
[b] Mixed pig/cattle slurry (each 50%) with TAN = 63% of total N, broadcast at 15°C ambient temperature, incorporated within 4 hrs.

due to its low sensitivity to environmental and management factors. This weakness may, however, be neglected because of the small contribution of dissolved P to the total P loss (roughly 10%).

Method choice has the most pronounced effect in N_2O and NO emission estimates. Indirect N_2O and NO are the ones where least research has been undertaken in the past. In the case of NO, this may be tolerable, considering the small share field emissions contribute to the total anthropogenic NO emission. In the case of indirect N_2O, the development of more sophisticated estimation methods appears crucial. The methodological differences between the N_2O models are intriguing, because the rather simplistic IPCC methodology and the relatively complex Bouwman method yield very similar results, while the Freibauer method, which is intermediate in complexity, differs.

NH_3 emissions in the maize system are by 60 – 80% higher than in spinach, which is plausible because of the use of animal manure. Spinach is likely to emit slightly more N_2O than the maize system. Direct N_2O accounts for 50 – 60% of the total emission in both systems. Nutrient losses to water are similar for both systems. Note however, that neither the additional risk from poor irrigation management in spinach nor the additional run-off risk from freshly applied manure in maize are accounted for.

Table 4.6 Nutrient emission estimates for the case study systems. Figures in brackets indicate uncertainty ranges.

Emission type, method	Emission estimate (kg ha^{-1} yr^{-1})			
	Spinach		Maize	
NH_3-N				
EMEP	9.6	[4.8–14.4]	17.2	[8.5–25.7]
Bouwman	10.5	[6.7–14.1]	16.5	[13.5-20.8]
N_2O-N				
direct				
IPCC	3.5	[0.7–6.4]	3.1	[0.6–5.6]
Freibauer	1.0	[0.6–1.4]	1.0	[0.6–1.4]
Bouwman	4.0	[2.4–6.9]	3.1	[1.8–5.3]
indirect [a]				
IPCC (default)	1.7	[0.4–7.7]	1.6	[0.4–6.7]
IPCC (Bouwman/DBG)	1.8	[0.5–8.0]	1.6	[0.4–8.2]
NO-N				
EMEP	0.6	[0.0–1.5]	0.5	[0.0–1.3]
Bouwman	1.6	[0.0–5.5]	1.3	[0.0–4.4]
NO_3-N	67	[7–147]	56	[6–123]
P [b]	3.9	[0–8.4]	4.5	[0–9.5]
particulate	3.8	[0–8.2]	4.4	[0–9.3]
dissolved	0.1	[0–0.2]	0.1	[0–0.2]

[a] 'Default' and 'Bouwman/DBG' refer to the estimation methods used to determine NH_3, NO and NO_3 emissions.

[b] P = particulate P + dissolved P. Differences in sums are due to rounding.

Potential mitigation strategies for N_2O emissions in spinach could include fertiliser selection (replace UAN solution by solid nitrate fertiliser) or incorporation of the UAN solution. Both measures would be reflected by *Bouwman-N_2O* but not by *IPCC-* and *Freibauer- N_2O*. Strategies to reduce the NH_3 volatilisation in maize could involve surface near application techniques and/or timelier incorporation. Both of them would be reflected by *EMEP-NH_3* while *Bouwman-NH_3* only accounts for the application mode. Reducing NO_3-leaching could involve measures to reduce inputs, such as more N efficient varieties, fertiliser selection and fertilisation scheduling, and ‘end-of-the-pipe’ solutions like growing catch crops after harvest. All of these measures would be reflected by the *DBG-NO_3* N balance approach. Finally, strategies to reduce P losses could involve reducing excess P pools in the soil and improved soil protection and infiltration to reduce erosion and surface run-off. Soil P is reflected by the methods discussed here. Soil protection would be accounted for through the RUSLE/ABAG erosion estimate in *Particulate-P* but not in *Dissolved-P*.

Chapter summary and conclusions

Agricultural fields are a source of ammonia (NH_3), nitrous oxide (N_2O), nitric oxide (NO), nitrate (NO_3) and phosphorus (P) emissions that are environmentally relevant. Environmental impact studies often rely on rather simple methods to quantify these emissions, because data, cost and time constrains do not allow for more detailed investigations. In this paper we analysed frequently used methods with regards to three qualifying criteria: (1) sensitivity to environmental and management factors, (2) uncertainty and (3) robustness against aggregation biases.

Methods that are adequately sensitive to environmental and management factors are available for all emissions discussed here, except for NO and dissolved P. This seems, however, tolerable because the agricultural share in the anthropogenic NO emissions is very low (refer to Table 4.1) and so is the share of dissolved P in the total P losses.

The methods discussed here are quite robust against aggregation biases, although non-linear methods must not be applied blindly to data that have been or are to be aggregated. In doubt, data stratification can be used to obtain approximately unbiased results. Uncertainties of the methods for gaseous N losses tend to be low ($\ll \pm 100\%$), while those for N and P losses to water are rather high ($\geq \pm 100\%$). As the amount of published measurement data on N and P losses to water is vast, these uncertainties can most likely be further reduced. A potential way to do so is a statistical approach similar to that by Bouwman et al. (2002a, 2002b, 2002c) for NH_3, N_2O and NO emissions. This would also provide researchers with a complete and methodologically consistent set of methods to estimate nutrient emissions from agricultural fields. Method choice matters for N_2O and NO emission estimates, as the case study showed.

We conclude that estimation methods for NH_3 and direct N_2O are available, which are satisfactory in terms of all of the three qualifying criteria used here, namely sensitivity to environmental and management factors, certainty and robustness against aggregation biases. With exception of certainty, this is also true for NO_3 and (particulate) P. Methods for indirect N_2O, NO, and dissolved P emissions are both highly uncertain and rather insensitive to important regulating factors. This is however tolerable because NO and dissolved P play a comparatively small role compared to other sources and because indirect N_2O emission are hardly manageable for agriculturalist, except for reduction of primary emissions.

Appendix to Chapter 4

Table 4.A.1 EMEP/CORINAIR emission factors for ammonia volatilisation form applying synthetic fertilisers (EEA, 2001). Emission factors refer to total fertiliser N.

	Group I [a]	Group II [b]	Group III [c]
Ammonium nitrate and calcium ammonium nitrate	0.03	0.02	0.01
Ammium sulfate	0.15	0.10	0.05
Anhydrous ammonia	0.04	0.04	0.04
Urea ammonium nitrate solution	0.08	0.08	0.08
Urea	0.20	0.15	0.15
Combined ammonium phosphate (generally DAP)	0.05	0.05	0.05
Other complex NK and NPK fertilisers	0.03	0.02	0.01

[a] Warm-temperate, soil pH usually >7 (Greece, Spain)
[b] Temperate, warm-temperate, soil pH usually around 7 (Italy, France UK, Ireland, Poland, BeNeLux)
[c] Temperate, cool-temperate, soil pH usually <7 (Scandinavia, Germany, Switzerland, Austria)

Table 4.A.2 EMEP/CORINAIR emission factors for Germany (Döhler et al., 2002) for ammonia volatilisation from applying organic fertilisers, depending on temperature and incorporation. Emission factors are expressed as proportion of total ammoniacal N (TAN).

Application mode [a]	Temp.	Emission factor					
		Incorporation within					
	°C	1 h	4 h	6 h	12 h	24 h	48 h
Cattle slurry							
bc, so	5	0.03	0.10	0.14	0.22	0.26	0.30
bc, so	10	0.06	0.18	0.25	0.32	0.36	0.40
bc, so	15	0.10	0.26	0.35	0.43	0.46	0.50
bc, st	25	0.20	0.65	0.78	0.85	0.90	0.90
th, so	5	0.01	0.06	0.09	0.15	0.22	0.26
th, so	10	0.03	0.10	0.14	0.22	0.31	0.36
th, so	15	0.04	0.15	0.20	0.30	0.39	0.46
th, st	25	0.10	0.35	0.47	0.70	0.80	0.90
Pig slurry							
bc, so	5	0.01	0.04	0.05	0.08	0.09	0.10
bc, so	10	0.025	0.06	0.08	0.12	0.16	0.20
bc, so	15	0.04	0.09	0.11	0.16	0.21	0.25
bc, st	25	0.15	0.37	0.47	0.60	0.67	0.70
th, so	5	0.01	0.02	0.03	0.045	0.06	0.07
th, so	10	0.01	0.04	0.05	0.08	0.11	0.14
th, so	15	0.02	0.06	0.08	0.11	0.14	0.18
th, st	25	0.09	0.19	0.25	0.37	0.48	0.55

[a] bc = broadcast, th = trailing hose,
so = onto bare soil, st = onto stubbles.

Table 4.A.3 EMEP/CORINAIR emission factors for Germany (Döhler et al., 2002) for ammonia volatilisation from applying organic fertilisers, depending on land use, manure type, application mode and incorporation. Emission factors are expressed as proportion of total ammoniacal N (TAN).

Manure type and application mode [a]		Emission factor Incorporation					
		none,	within				
		>24 h	1 h	4 h	6 h	12 h	24 h
Arable land							
Cattle manure	bc	0.90	0.09	0.45			0.90
Cattle slurry	bc	0.50	0.10	0.26	0.35	0.44	0.46
Cattle slurry	th (s)	0.46	0.04	0.15	0.20	0.30	0.39
Cattle slurry	th (l)	0.63					
Cattle slurry	th (h)	0.35					
Pig manure	bc	0.90	0.09	0.45			0.90
Pig slurry	bc	0.25	0.04	0.09	0.11	0.16	0.21
Pig slurry	th (s)	0.18	0.02	0.06	0.08	0.11	0.14
Pig slurry	th (l)	0.25					
Pig slurry	th (h)	0.13					
Poultry manure	bc	0.90	0.00	0.18			0.45
Urine	bc	0.20	0.02	0.07			0.18
Grassland							
Cattle slurry	bc	0.60					
Cattle slurry	th (l, h)	0.60					
Cattle slurry	ts	0.60					
Cattle slurry	os	0.54					
Pig slurry	bc	0.30					
Pig slurry	th (l)	0.21					
Pig slurry	th (h)	0.30					
Pig slurry	ts	0.15					
Pig slurry	os	0.12					

[a] bc = broadcast, th = trailing hose, ts = trailing shoe, os = open slot, (s) = onto bare soil, (l) = into low canopy (<0.3 m), (h) = into high canopy.

Table 4.A.4 Typical contents of total ammoniacal N (TAN = NH_4-N + NH_3-N) of various organic fertilizers (value rounded). Values taken from Brentrup et al. (2000), Sommer and Hutchings (2001) and Döhler et al. (2002).

Manure type	Dry matter (%)	N content (kg t^{-1})	TAN-fraction
Cattle manure	25	5	0.10-0.25
Cattle slurry	8	4	0.40-0.55
Cattle lqd. manure/urine	2	2.5	0.80-0.90
Calf slurry	3	3.5	0.55
Pig manure	23	6	0.10-0.40
Pig slurry	6	5	0.65-0.70
Pig liquid manure/urine	n.a.	n.a.	0.85-0.90
Poultry manure	57	3	0.20-0.65
Poultry slurry	18	10	0.50-0.70
Horse & sheep manure	n.a.	n.a.	0.40

Table 4.A.5 Factor classes and factor class values for the Bou w man et al. emission models for ammonia, nitrous oxide and nitric oxide.
The NH_4 model is
$E_{NH4\text{-}N} = R*EXP(F_1+F_2+F_3+F_6+F_7+F_9)$;
the N_2O model is $E_{N2O\text{-}N} = EXP(C+F_1+F_2*R+F_4+F_5+F_6+F_8+F_9)$;
the NO model is
$E_{NO\text{-}N} = EXP(C+F_2*R+F_5+F_8)$;
where E is the emission rate of the particular N species in kg N ha^{-1} yr^{-1} and R is the application rate in kg N ha^{-1} yr^{-1} (after Bouwman et al., 2002a; Bouwman et al., 2002b; IFA/FAO, 2001).

Factor, factor class	Factor value		
	NH_3– model	N_2O– model	NO– model
Fixed factor (C)		0.411	-1.527
Crop type (F_1)			
Grass	-0.158	-1.268	
Grass-clover		-1.242	
Legume	-0.046	-0.023	
Other upland crops	-0.047	0.000	
Wetland rice	0.000	-2.536	
Fertiliser type (F_2) [a]			
Ammonium sulfate	0.429	0.0051	0.0056
Urea	0.666	0.0051	0.0061
Ammonium nitrate	-0.350	0.0061	0.0040
Calcium ammonium nitrate	-1.064	0.0037	0.0062
Calcium nitrate	-1.585	0.0034	0.0054
Anhydrous ammonia	-1.151	0.0056	0.0051
Other ammon. based fertilisers		0.0051	0.0056
Other nitrate based fertilisers		0.0034	0.0054
N solutions	-0.748		
Urea ammonium nitrate solution	0.000	0.0053	0.0004
Monoammonium phosphate	-0.622	0.0039	0.0055
Diammonium phophate	0.182	0.0039	0.0055
Other compound NP and NPK	0.014	0.0039	0.0055
Compound NK	-1.585		
Ammonium bicarbonate	0.387	0.0051	0.0056
Animal manure	0.995	0.0021	0.0016
Animal manure plus synthetic N		0.0042	0.0055
Urine	0.747	0.0051	0.0061
Grazing	-0.378		
Application mode (F_3)			
Broadcast	-1.305		
Incorporate	-1.895		
Apply in solution	-1.292		
Broadcast or incorporate, then flood	-1.844		
Broadcast to floodwater at panicle initiation	-2.465		
Soil texture (F_4) [b]			
Coarse		-0.008	
Medium		-0.472	
Fine		0.000	
Soil organic carbon content in%(F_5)			
SOC ≤ 1.0		0.000	0.000
1.0 < SOC ≤ 3.0		0.140	0.000
3.0 < SOC ≤ 6.0		0.580	2.571
SOC > 6.0		1.045	2.571
Soil pH (F_6)			
pH ≤ 5.5	-1.072	0.000	
5.5 < pH ≤ 7.3	-0.933	0.109	
7.3 < pH ≤ 8.5	-0.608	-0.352	
pH > 8.5	0.000	-0.352	
Soil cation exchange capacity in cmol kg^{-1} (F_7)			
CEC ≤ 16	0.088		
16 < CEC ≤ 24	0.012		
24 < CEC ≤ 32	0.163		
CEC > 32	0.000		
Soil drainage (F_8)			
Poor		0.000	0.000
Good		-0.420	0.946
Climate (F_9) [c]			
Temperate	-0.402	0.000	
Tropical	0.000	0.824	

[a] Multiply with N application rate for N_2O and NO model.

[b] ‘Coarse’ includes sand, loamy sand, sandy loam, loam, silt loam and silt; "Medium" includes sandy clay loam, clay loam and silty clay loam; "Fine" includes sandyclay, silty clay and clay.

[c] For NH_3: "Temperate" = temperatures <20°C,"Tropical" = ≥20°C
For N_2O and NO: "Temperate" = temperate oceanic and continental, cool tropical, boreal and polar/alpine; "Tropical" = (sub-) tropical, subtropics winter/summer rains, tropics, warm humid, tropics warm seasonal dry.

CHAPTER 5

Evaluating land use related impact indicators for sustainability*

Abstract

In the past decade, numerous indicators and indicator sets for sustainable agriculture and sustainable land management have been proposed. Land managers are often interested in comparing not only different management systems on an indicator by indicator basis, but also to compare individual indicators against a threshold or, in order to study trade-offs, among each other. To this end it is necessary to (1) transform the original indicators into a comparable format and (2) score these transformed indicators against a sustainability function.

This paper introduces an evaluation scheme for land use related impact indicators, which was designed to accomplish these tasks. It links into a larger framework for sustainability assessment of land use systems (see Chapter 2), which is briefly presented. The evaluation scheme introduced here comprises (1) a standardisation procedure, which aims at making different indicators comparable. To this end, they are first normalised by referencing them to the total impact they contribute to and then corrected by a factor describing the severity of this total impact in terms of target exceedance. This standardisation procedure borrows conceptually from Life Cycle Assessment (LCA) Impact Analysis methodology; (2) a valuation procedure, which judges the individual standardised indicators with regards to sustainability.

This methodology is then tested on an indicator set for the environmental impact of a spinach production system in North Western Germany. The method highlights mineral resource consumption, greenhouse gas emission, eutrophication and impacts on soil quality as most important environmental effects of the studied system.

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We then explore the effect of introducing weighting factors, reflecting the differing societal perception of diverse environmental issues. Two different sets of weighting factors are used. The influence of weighting is, however, small compared to that of the standardisation procedure introduced earlier.

Finally, we explore the propagation of uncertainty (defined as a variable's 95% confidence limits) throughout the standardisation procedure using a stochastic simulation approach. The uncertainty of the analysed standardised indicator was by a factor of 2.0 to 2.5 higher than that of the non-standardised indicators.

Introduction

Agriculture is one of the human activities most tightly connected with land (cf. Matson et al., 1997; Fields, 2001; Tilman et al., 2001), but virtually any human enterprise is associated with land use or land occupancy. Even virtual activities rely on physically real space to host power generation and hardware. Land occupancy is thus a trait characteristic of most human activities.

Numerous indicators and indicator sets for sustainable agriculture and sustainable land management have been proposed in the past years (Niu and Khan, 1993; Izac and Swift, 1994; Stockle et al., 1994; Smyth and Dumanski, 1995; Bockstaller et al., 1997; van Mansvelt, 1997; Smith and McDonald, 1998; Wackernagel and Yount, 1998; Cornforth, 1999; Halberg, 1999; Lewandowski et al., 1999; Eckert et al., 2000; Sands and Podmore, 2000; Reganold et al., 2001; Stevenson and Lee, 2001; for a review see Chapter 2). Land managers are often interested in comparing not only different management systems on an indicator-by-indicator basis, but also in comparing individual indicators against a threshold (Syers et al., 1995) or, in order to study trade-offs, among each other. To this end it is necessary to firstly transform the original indicators into a comparable format and to secondly submit these transformed indicators to a sustainability scoring function (cf. Chapter 2).

This paper presents an evaluation method for indicator sets that describe the impact of land use systems on sustainability. The method is suited for evaluating impact indicator sets. Impact indicators are here defined as measures of a land use system's contribution to certain 'threats to sustainability' (Smith and McDonald, 1998) or 'constraints to sustainability' (Stockle et al., 1994), i.e. indicators that in the OECD indicator classification scheme would be classified as Pressure/Driving Forces indicators (OECD, 1998a, 2000). This confines the method to indicator sets that describe 'bads', which are, however, often more salient and policy relevant than 'goods' and less subject to differing perceptions (Costanza, 1993; Ludwig et al., 1993; Jamieson, 1998).

The method presented here links into a broader framework of indicator based sustainability assessment (Chapter 2), which can be conceptualised in three stages, as shown in Figure 2.1. In **Stage 1** the set-up of the actual indicator set is addressed, i.e. first identifying the problems to be assessed (i.e. the actual matters of interest, also called 'indicanda') and then attributing indicators that well describe these indicanda. **Stage 2** comprises a standardisation procedure (to make different indicators comparable) and the actual sustainability valuation procedure, which assigns a sustainability value to each indicator value. This paper focuses on Stage 2 and the method presented here will address these two steps: indicator standardisation and sustainability valuation. **Stage 3** finally deals with developing a strategy for improvement and with making improvements visible.

The method presented here includes elements inspired by Life Cycle Assessment (LCA) methodology. It was, however, extended to meet a number of specific requirements for land uses evaluation in the context of sustainability. In particular, the method was developed to

- be appropriate for environmental, social and economic indicators alike, i.e. be suitable for all dimensions of sustainability;
- allow for comparing very different land uses systems, such as agriculture and commerce;
- acknowledge the fact that sustainability issues emerge at very different spatial scales, ranging from m^2 or even less to hundred thousands of km^2;
- separate the descriptive and normative elements of sustainability evaluation as clearly as possible.

The latter criterion – keeping descriptive and normative elements separate – acknowledges the fact that science and scientists play a dual role in the sustainability debate: On one hand they are a part of society and policy decisions do affect their individual environments. On the other hand they are to inform societal and political decision making processes. The first role is connected to the normative stratum of the sustainability debate, the second to the descriptive. There is a fine line between these two roles and they are not always easy to separate. In fact, they are rather extreme poles of a broad continuum than clear-cut opposites. Any normative decision implies certain descriptive elements and vice versa (Hoyningen-Huene, 1999). In order to assure the quality of scientific information, we hence hold that it is important to be explicit about normative elements and about the limitations of descriptiveness (cf. Funtowicz and Ravetz, 1993; Tacconi, 1998). For the method presented here, this means that it does not attempt to engage in normative debates with allegedly scientific arguments. It is meant to inform decision making processes and to fuel further debates but not to generate ‘absolute facts’.

This paper is structured as follows: First the standardisation and sustainability valuation procedures are introduced in detail. In order to enhance the transparency, we highlight and discuss underlying assumptions and implications of the method. We then apply the presented method to evaluate an actual impact indicator set, which was developed to assess a spinach cropping system, covering some 1,000 ha in the County of Borken, Northwest Germany. As the case study data are subject to large uncertainty, we also assess the propagation of uncertainties through the standardisation procedure onto results by using stochastic simulation. Finally, case study results are discussed and main findings are summarised.

The evaluation scheme

According to the above framework, evaluation comprises two separate steps: Indicator standardisation and sustainability valuation.

Indicator standardisation

The standardisation procedure introduced here aims at making different indicators comparable. It transforms each indicator into a dimensionless index by multiplying it with two factors: The first one relates the impact of the system under investigation (e.g. phosphate losses from agricultural fields) to the overall pressure causing the issue that the indicator describes (e.g. the total phosphate load discharged into the watershed the field is situated in). The second factor describes the severity of this issue itself (e.g. eutrophication of the watershed caused by phosphate). The standardised indicator for any issue i is then:

$$std\, I_{S\,i} = I_{S\,i} \times NF_i \times SF_i \tag{5.1}$$

with

$std\, I_{S\,i}$ Standardised indicator for a particular issue i (dimensionless)

$I_{S\,i}$ Indicator value describing the impact of land use system S on issue i (any dimension × ha^{-1} yr^{-1})

NF_i Normalisation factor for issue i (dimension is reciprocal to that of $I_{S\,i}$)

SF_i Severity factor for issue i (dimensionless).

The normalisation factor, NF_i, is the reciprocal of the indicator calculated for an average hectare of the spatial scale, on which the issue emerges:

$$NF_i = \left(\frac{I_{tot\,i}}{A_{tot\,i}} \right)^{-1} \tag{5.2}$$

with

$I_{tot\,i}$ Indicator value describing the total impact, to which land use system *S* contributes and which causes issue *i* ([same dimension as $I_{S\,i}$] × yr^{-1})

$A_{tot\,i}$ Total land area of the scale level, at which issue *i* typically emerges (ha).

The severity factor, SF_i, is the ratio of an actual and a critical impact level (i.e. what was called a distance-to-target ratio in Chapter 3):

$$SF_i = \begin{cases} 0 & \text{if } L_{act\,i} / L_{crit\,i} < 1 \\ L_{act\,i} / L_{crit\,i} & \text{if } L_{act\,i} / L_{crit\,i} \geq 1 \end{cases} \tag{5.3}$$

with

$L_{act\,i}$ Actual level of the impact causing issue *i* (any dimension)

$L_{crit\,i}$ Critical level, beyond which impacts cause irreversible or severe damage (same dimension as $L_{act\,i}$). Must be ≠ 0.

Multiplication with the normalisation factor, NF_i, equals dividing the average annual per ha impact of the land use system under investigation by the average total annual per ha impact it contributes to. The product $I_{S\,i} \times NF_i$ can thus be interpreted as the factor, by which the total impact would change if the particular land use under investigation would be extrapolated to the issue-specific area.

The issue-specific area $A_{tot\,i}$ is the geographic extent of the spatial scale, at which the particular issue emerges and is normally assessed. It is here assumed to be predetermined by conventions of the pertinent scientific disciplines. E.g. the impact of pesticides on aquatic organisms would normally be assessed on a regional scale, whereas the emission of greenhouse gases affects the global level.

Ideally, the actual and critical impact levels are assessed using the same methodology as the indicator values. In that case $L_{act\,i} = I_{tot\,i}$ and Equation 5.1 reduces to

$$std\, I_{S\,i} = I_{S\,i} \times A_{tot\,i} \,/\, L_{crit\,i} \tag{5.4}$$

In practice it is, however, often difficult to find information on critical impact levels that are methodologically consistent with the data required for indicator calculation (e.g. differences in base year or inventory method). In such cases may $L_{act\,i} \neq I_{tot\,i}$.

The severity factor, SF_i, accounts for the fact that different issues may be differently pressing. It can be interpreted as the factor, by which the actual impact level exceeds the critical one. (This is similar to what in LCA is – misleadingly – called ‘distance-to-target weighting’; Brentrup et al., 2004; Müller-Wenk, 1996). If the actual impact level is below the critical impact level, there is obviously not an issue and SF_i is consequently set to zero.

Sustainability valuation

The following valuation function then assigns the standardised indicators to three discrete sustainability classes:

$$val\, I_{S\,i} = \begin{cases} \text{sustainable} & \text{for } 0 \le std\, I_{S\,i} < cl \\ \text{critical} & \text{for } cl \le std\, I_{S\,i} < 1 \\ \text{unsustainable} & \text{for } 1 \le std\, I_{S\,i} \end{cases} \tag{5.5}$$

where

$val\, I_{S\,i}$ is the sustainability classification for of land uses system *S* with regards to a issue *i*, and

cl is a lower limit <<1 (dimensionless).

Note that the class ‘sustainable’ should, strictly speaking, be named ‘not unsustainable’, because conceptually the method assesses constraints or limitations to sustainability, not sustainability as such: Absence of known constraints of sustainability, as measured with impact indicators, does not allow for inferences on the sustainability of a system, because other constraints may not yet be known (cf. Chapter 3 for a discussion of the feasibility of statements on sustainability). The term ‘sustainable’, although less precise, is here used for reasons of simplicity.

The sustainability valuation procedure can easily be extended to a probabilistic version to account for risk and uncertainty. In the probabilistic version we may demand that a standardised indicator be below or above the class limits (*cl* and 1) with a certain probability, say, 67 or 95%. This does, however, require information on the probability distribution of the standardised indicators, which is often difficult to obtain.

Example

Before discussing the assumptions and implications of this evaluation scheme we shall illustrate it by an example: Consider the terrestrial eutrophication potential of a hypothetical land use system in Central Europe. Eutrophication in Europe is usually assessed at the continental scale (e.g. Posch et al., 1995). This corresponds with the re-deposition patters of reactive N species emitted in Europe (Ferm, 1998; Mosier, 2001): About 50% of the NH_3 is re-deposited near the source and more than 90% within 1,000 km. About 75% of the NO_X is re-deposited within 2,000 km from the source.

Assume our hypothetical system emits 60 kg NO_X-eq. ha^{-1} yr^{-1}. The normalisation factor, $NF_{eutroph\_terr}$, would be computed from the area on which the problem is assessed ($A_{tot\ eutroph\_terr}$), here Europe (982.2×10^6 ha), and the average annual emission from that area (45 Gt NO_X-eq. yr^{-1}): $NF_{eutroph\_terr}$, is thus 982.2×10^6 ha / 45 Gt NO_X-eq. yr^{-1} = 0.02 (kg NO_X-eq.)$^{-1}$ ha yr.

The critical emission level (critical loads concept, Posch et al., 1995) for Europe is equivalent to 22 kg NO_X-eq. ha^{-1} yr^{-1} (given the target of not exceeding critical loads in more than 5% of the ecosystems; Amann et al., 1999). The actual level of 45 Gt NO_X-eq. ha^{-1} yr^{-1} was given above. The severity factor, $SF_{eutroph\_terr}$, is thus 45 / 22 = 2.05. The standardised indicator is then

$$I_{S\_eutroph\_terr}\times NF_{eutroph\_terr}\times SF_{eutroph\_terr} = 60\times0.02\times2.05 = 2.46.$$

Since this is greater 1, it would be classified 'unsustainable' according to the above sustainability valuation function.

Basic assumptions and implications

Any standardisation and evaluation method is value-laden, in that it carries implicit values due to choices (e.g. of concepts and data) and fundamental assumptions. In order to assure transparency we shall highlight some of the implications, assumptions and limitations of the presented method before turning to a case study example.

In the text to follow, the index *i*, denoting a particular issue, has been dropped for easier reference.

Indicator standardisation

- Above we interpreted the product $I_S\times NF$ as the factor, by which the total impact would change if the land use under investigation was extended to the specific spatial scale of the issue, A_{tot}. The severity factor was interpreted as the factor, by which the actual impact level exceeds the tolerable level. The result of the total standardisation procedure is thus the factor, by which the total impact would exceed the tolerable level if the land use under investigation would be extended to the total area, A_{tot}. Of course this is usually neither a policy option nor realistic at all. Rather, this assumption should be seen as a useful proposition in order to compare diverse land use systems in a standardised way.

- The evaluation scheme presented here treats all land use types equally. Accordingly, it subjects all land uses to the same evaluation rules, regardless of social and political priorities, mitigation potentials or other criteria that could justify differentiated treatment of different systems. In this context it is important to note that (a) the total impacts I_{tot} include impacts from *all* land use types and sectors (e.g. not only those from agriculture) and that (b) the issue-specific areas A_{tot} comprises *all* usable land (e.g. not only agricultural land). This may at first appear counter-intuitive. However, using agricultural area only would imply attributing the impacts of all other land uses, such as industry and traffic, to agriculture as well. This is certainly not adequate, because agriculture is not responsible for impacts from other sectors. Likewise, taking into account only agricultural impacts and agricultural area would reduce the evaluation to an isolated analysis of the agricultural sector. This is not adequate either, because the source of an impact does not matter with regards to the effect: E.g. will sensitive ecosystems react to nitrogen immissions, regardless of whether they stem from agriculture of traffic.

- If I_S changes, I_{tot} must change also, because I_S feeds into it. Thus I_{tot} is a function of I_S. For large scale issues, the absolute contribution of the system under investigation will usually be marginal and changes in I_{tot} due to changes in I_S can be neglected. But for small-scale issues, such as field-level impacts on soil, I_S contributes a substantial proportion to I_{tot}. For these issues I_{tot} needs to be re-computed if I_S changes.

- The critical impact level L_{crit} could either be based on policy decision and negotiations (e.g. international conventions) or on scientific grounds (e.g. critical load concepts). We here propose to use the latter category. Although negotiated solutions and participatory techniques have become popular in the context with sustainability, they are inadequate when biophysical realities are involved, as these are not negotiable. In some instances science-based critical levels may not be available. Policy-based values could then be used as a proxy. They should, however, be clearly marked as such.

In this context it is important to note that of course ‘scientifically chosen’ critical impact levels are not free of values either. However, they root in the broader discussion within the pertinent disciplines and are subject to the usual mechanisms of scientific quality control (e.g. peer reviewing).

- The computation of the severity factor as $SF = L_{act} / L_{crit}$ implies a linearity assumption: Doubling L_{act} will double SF, i.e. the severity of the particular issue i increases linearly within creasing L_{act}. This is certainly not always realistic. However, for the time being, the real cause-effect relations are usually unknown or highly uncertain. The potential bias caused by the linearity assumption will increase as L_{act} / L_{crit} diverges from one. Severity factors much greater than 1 should thus be treated with great caution.

- Another fundamental assumption is that the normalisation factor and the severity factor can compensate each other. This is stringent for the ideal cases where $L_{act} = I_{tot}$ (see above). It may, however, bias results if proxy impact levels are used and $L_{act} \neq I_{tot}$. The bias will increase as, either L_{act} / L_{crit} or $I_S \times NF$ deviate extremely from 1. These cases should, again, be treated very cautiously.

- A constraint to the method is that the standardisation procedure requires L_{crit} values to be unequal zero. This makes it e.g. unsuitable for indicators with zero tolerances. In many cases, redefinition of the indicator may provide a solution to this problem. A special case is that of input-output balances: Often the target is to balance inputs with outputs, i.e. the aim is a zero residue. In that case severity factors can be computed by using the input-output ratio as the actual impact level, L_{act}, and 1 (equalling balanced inputs and outputs) as the critical level, L_{crit}.

Sustainability valuation

- It is important to note that the sustainability valuation procedure proposed here works on the level of the individual indicator: It allows for statements on sustainability only with regard to the particular issue that the indicator describes. It does not judge an 'overall sustainability' of the entire land use system.

- The sustainability classification used in the valuation procedure requires explanation. It grounds onto the premise that it should be possible to identify (a) a range, within which an indicator could be regarded definitely sustainable (or 'not unsustainable', to be precise) and likewise (b) a range, within which it is definitely unsustainable. We here argue that the former is the case at $std\, I_S = 0$ and the latter at $std\, I_S \geq 1$. The range between these two 'definite' realms cannot be attributed to either one and is here called 'critical'. The argumentation for the two 'definite' realms is as follows:

(a) Obviously, if $I_S \times NF \times SF = 0$, the impact of a land use system is no threat to sustainability, because either $SF = 0$ or $I_S = 0$, i.e. there is either no impact or no issue (NF should, for logical reasons, never be zero if I_S is not, because the latter feeds into the total impact, I_{tot}, which is the denominator of NF). In addition there could be situations where the impact of the land use system under investigation is very small compared with other systems, i.e. $I_S \times NF$ is very close to zero. By convention, there could be a value, below which an impact is regarded negligible. The method presented here provides for this by introducing a lower limit or cut-off value, cl, that could be set to, say, 0.1. Then, if $std\, I_S < cl$ (and not just if $std\, I_S = 0$) the particular indicator would be classified 'sustainable'. The choice of cl is of course arbitrary and mainly defendable for practicable reasons.

(b) The argument for classifying indicators with *std* $I_S \geq 1$ as 'definitely unsustainable' is less straightforward at first sight. It grounds on the premise that, with regards to an individual indicator, a situation where $L_{act} > L_{crit}$ is unsustainable. Consider the marginal situation $I_S \times NF = SF = 1$ and recall the ideal case, in which $L_{act} = I_{tot}$ (see above). This would mean that the actual impact level equals the critical one and our land use system contributes just as much to the issue as others do. If we now increase $I_S \times NF$, this means that I_{tot}, into which I_S feeds, would increase (if only marginally for large scale issues). If $L_{act} = I_{tot}$, the overall situation worsens consequently and shifts SF to the unsustainable realm, i.e. a situation where $L_{act} > L_{crit}$. (In practice, these changes will often be marginal and negligible in absolute magnitude. It is thus often allowable to treat I_{tot} and L_{act} as constant. This does, however, not vitiate the above argument.) Similarly, if again $I_S \times NF = 1$ and we now increase SF to above 1 this would mean that there is a threat to sustainability, to which our land uses system contributes equally to others. This situation would, again, be classified 'unsustainable'.

Aggregation and weighting

It is often desirable to aggregate several indicators into broader thematic indices. The standardisation procedure described here subjects indicators to a uniform transformation that yields dimensionless and comparable figures. These can readily be aggregated to higher-level thematic indices by simply adding them. This can be done repeatedly on various hierarchical levels: E.g. for a certain land use system, various environmental indices can be aggregated to a single environmental index, which then can be aggregated with a social and economic impact index to yield a total sustainability index for the system.

Apart from the technical aspect of aggregation stands the normative one: Any aggregation scheme treats different items as comparable. The adequacy of such comparisons certainly requires discussion. This discussion involves, however, strongly normative decisions (cf. Schäfer et al., 2002) and leading this discussion is not the intention of this paper. We here address aggregation, because there seems to be a considerable demand for aggregated indices to guide policy, land managers and research.

Within such an aggregation procedure, there might be a desire to give some issues more weight than others, according to societal or policy preferences. (Note that the severity factor reflects the severity of the individual issues in terms of target exceedance, but does not *weight* them in relation to each other!) E.g. issues related to human health could be regarded more important than those related to ecosystem integrity; issues of individual interests could be appraised less important than those affecting a broader public; irreversible damage could be judged more important than short-term impacts.

These examples show that any weighting involves decisions about priorities, values and the judgement of trade-offs, to which there are no scientific answers. This is not to say that science does not play a role in informing decisions on values and priorities, but it cannot make them. Weighting functions thus need to be agreed within the individual context that the method is applied in.

Given a set of agreed weighting factors is available, they can easily be incorporated into the aggregation procedure proposed here: Before adding them to an aggregated index, the individual standardised indicators would simply be multiplied with their corresponding weighting factor:

$$Index_{\lambda} = \sum_{j} \left(Index_{j,\lambda-1} \times WF_{j,\lambda-1} \right) \qquad (5.6)$$

where

$Index_{\lambda}$ is the Index on aggregation level λ

$Index_{j,\lambda-1}$ is the index (or standardised indicator) for theme (or issue) j on the aggregation level λ–1

$WF_{j,\lambda-1}$ is a weighting factor for theme (or issue) j on the aggregation level λ–1.

Case study: Spinach cropping in Northwest Germany

Background

The following section illustrates the above evaluation scheme by an example. It uses an indicator set developed to assess the sustainability of a spinach production system in the County of Borken, North Western Germany. Some one hundred contract growers in Borken grow spinach for a local frozen foods factory on roughly 1,000 ha land per year. Other crops in the rotations include fodder maize (40 – 50%) and cereals as well as sugar beets, potatoes, fine herbs and other vegetables. The area is characterised by intensive agriculture and livestock production. The climate is temperate oceanic with a mean temperature of 9.7°C and an annual precipitation of 760 mm. Soils are sands and loamy sands with pH values around 6.0. Further details on management and local conditions are given elsewhere (Chapter 1).

Spinach is cropped in a one-in-four years rotation and grown twice on the same field in the contact year. Sowing and harvesting dates are mandated by the factory's fieldsmen, as are amount and timing of fertilisation and pest control measures. Contract growers are responsible for carrying out all crop management measures and for irrigation. Harvest and transportation are contracted to a third party company, on behalf of the factory.

Fertilisation is carried out after soil testing, pest control follows an integrated pest management (IPM) scheme. A nitrogen base dressing is applied as liquid sprayed fertiliser (urea ammonium-nitrate solution), top dressings are applied as calcium ammonium-nitrate. Usually, no P is applied, because of high soil P contents as a consequence of intensive manuring. Herbicides are usually applied in repeated split rates to increase efficacy. Insecticides, if necessary, are selected to minimise environmental impacts. Fungicides are not applied. Resistant varieties are cropped during periods of high pressure of downy mildew.

Contracts are made with each individual grower for a particular field on a yearly basis. During the contract year they grant the factory's fieldsmen control of crop management. However, between contract years, the factory has little control of the land management. Due to these specifics there is a particular interest to assess the impact of the single crop spinach on sustainability (rather than that of the full rotation).

Methods and data

Due to the scope of the study only environmental impacts were assessed, but the approach can be extended to social and economic impacts analogously. Relevant environmental issues were identified through a structured inventory of potential issues from the literature followed by a systematic selection for relevance within the actual context (Chapter 3). Severity factors as the ones described above were used as a decision criterion in this selection: Issues with a severity factor less one were appraised not relevant for agricultural production in the County of Borken, the ones with a factor greater one were classified relevant. Along with a set of environmental impact indicators given in Table 5.1, three economical impact indicators (cost, labour and risk) as well as yield were recorded. These three indicators were, however, not standardised and valuated.

We used the inventory model described in the Appendix to this Chapter to generate the input data for indicator calculation. Figure 5.1 shows the scope of the inventory, data sources are described in the Appendix to this Chapter and the technical notes to Table 5.2, severity factors were taken from Chapter 3 and are documented there in detail. Some issues were not represented due to lacking suitable or practicable indicators: They include human exposition to pesticide and emission of noise and odours. Pesticide use with regards to ecotoxicity was evaluated after Burth et al. (2000) and Gutsche and Enzian (2002), as described in Chapter 3 and classified not relevant according to the above criterion.

Usually, environmental indicators are subject to rather high uncertainty, e.g. due to poor data quality, natural variability between objects, measurement and model errors and incomplete knowledge about the phenomena under investigation. To explore the propaga-

Table 5.1 Indicators used to describe the environmental impacts of a spinach production system in the County of Borken, Northwest Germany. For details on indicator calculation refer to Annex I.

Issue	Assessment scale	Indicator	unit *(ha^{-1} yr^{-1})
A Soil fertility related issues			
Soil loss	Field	Soil loss/formation ratio	dimensionless
Damage to soil structure	Field	Soil compaction index	dimensionless
Acidification/alkalinisation	Field	Proton input/output ratio	dimensionless
Accumulation of contaminants	Field	Heavy metal accumulation index	dimensionless
Depletion of soil organic matter (SOM)	Field	SOM output/input ratio	dimensionless
B Resource related issues			
Fossil fuel consumption	World	Non-renewable energy demand	GJ primary energy-eq.
Consumption of minerals	World	Mineral potassium consumption	t K_2O-eq.
Land occupancy	Region	Land naturalness degradation (Naturalness Degradation Potential)	dimensionless
Water consumption	River basin	Water consumption	m³
C Emission related issues			
Global warming	World	Emission of greenhouse gases (Global Warming Potential)	t CO_2-eq.
Summer smog/ground level ozone	Continent	Emission of NMVOCs (proxy for ozone precursors)	t NMVOC
Terrestrial eutrophication	Continent	Emission of eutrophying substances (Terrestrial Eutrophication Potential)	t NO_X-eq.
Marine eutrophication	Marine catchment	Emission of eutrophying substances (Aqautic Eutrophication Potential)	t PO_4-eq.

tion of uncertainties associated with the input data, a stochastic simulation approach (cf. IPCC, 2000) was used to calculate uncertainty ranges for three of the standardised indicators: For Soil erosion, Marine eutrophication and Greenhouse gas emissions. Uncertainty ranges of the input data to these indicators were taken from the original data sources. The analysis was limited to these three indicators due to insufficient because for the others information on input data uncertainty was unavailable. Triangular distributions (Johnson and Kotz, 1999) were used as an (finite interval) approximation of the true (but unknown and semi-finite interval) probability density function for each input variable (cf. Johnson, 1997).

Calculations were carried out with the GenStat® statistical software package (GenStat Committee, 2003). For each indicator, 10,000 values were generated. Uncertainty ranges, defined as the interval between the 2.5% and the 97.5% percentile, were then determined from the resulting distribution.

To assess the impact of introducing a weighting step into the evaluation scheme, we used two (imaginary) sets of weighting factors (Table 5.3). Set A is based on the appraisal of five spinach contract growers who had been involved in a sustainability project for three years and were asked to rank the importance of the issues listed in Table 5.1 on a 1 to 5 scale. Mean scores are used as weighting factors here. Set B is based

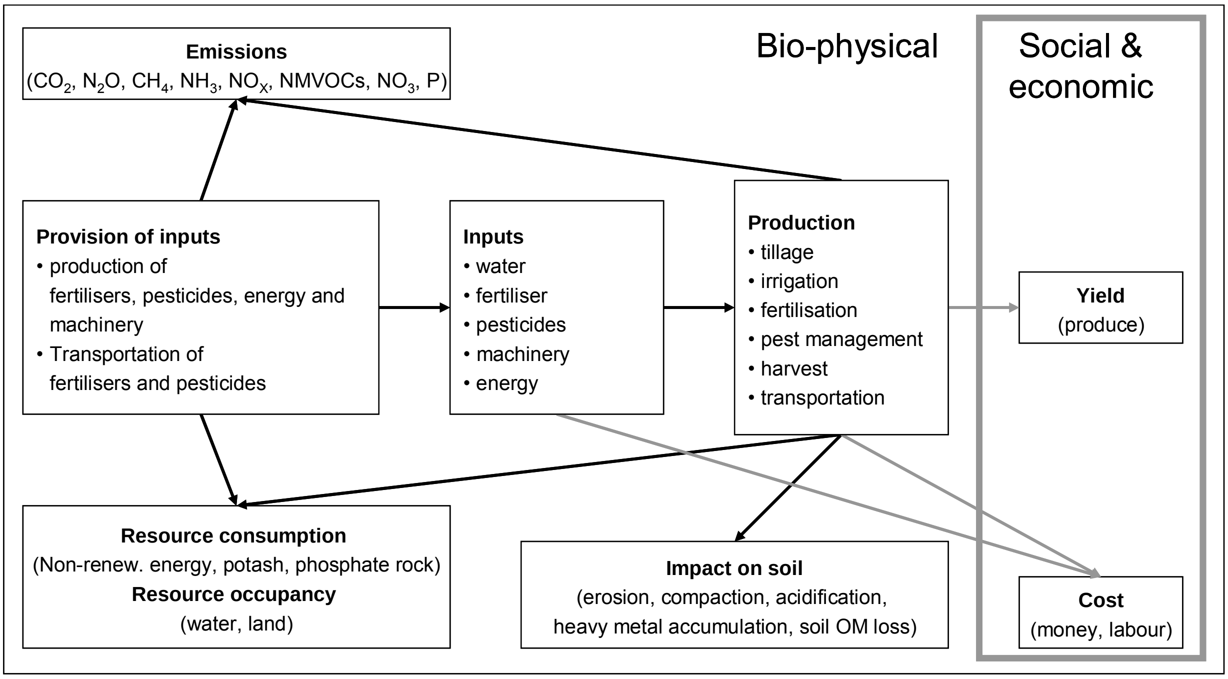


Figure 5.1 Scope of the impact inventory in the case study. Black arrows indicate biophysical, grey arrows social and economical impacts. Refer to Appendix to Chapter 5 for data and model used.

on the frequency by which issues are named in a sample of 44 publications on sustainable agriculture (publications reviewed in Chapter 3). For Set B, the percentage of references that appraised a particular issue as relevant to sustainable agriculture was taken as a weighting factor. To make the two sets comparable, each set was transformed by dividing the individual values by the average of the particular set, i.e. the transformed weighting factors are centred around one.

Note, that the weighting factor sets used are by no means representative and have not been agreed among any group of societal actors. They simply resemble 'weighting factor sets as they could be' that are simply used to test the sensitivity of evaluation results to different weighting schemes.

Case study results and discussion

Table 5.2 shows the normalisation and severity factor used, the standardised indicators and the sustainability valuation results. Among the standardised indicator values, the high value for mineral K consumption springs to eye. It is due to the very high normalisation factor, which reflects that agriculture consumes much more K than other land uses. The same would hold for mineral P if P fertiliser was applied.

In fact, agriculture is the single dominant user of mineral P and K (e.g. in the US, it consumes over 90% of the total P and K production; USGS, 2001). Given present consumption rates and presently known (economically exploitable) reserves, these resources will be consumed within 90 (P) and 320 (K) years. This is alarming, since there are virtually no substitutes for these resources and agricultural production – which provides over 99% of the human diet (Vitousek et al., 1997) – is highly dependent on adequate replenishment of soil nutrient pools. Interestingly, the issue of rapid consumption of mineral resources is hardly raised in the literature on sustainable agriculture. Even more interestingly, the case of finite fossil *fuel* resources is, although (a) the agricultural sector contributes far less than 2% to the global energy consumption (WRI, 2003a), (b) the potential for substitution is much higher and (c) agricultural production is, compared to P and K fertilisation, less vitally dependent on it.

Concerns about the dwindling of fossil fuel resources and their implications for equitable global development were among the most important drivers of the emergence of the concept of sustainable development since they have been articulated in the report 'The Limits to Growth' to the Club of Rome (Meadows et al., 1972). This might explain their prominence in the literature on sustainable agriculture. It is, however, hardly explicable, why the depletion of mineral P and K have received that little attention.

We thus hold the high value the method assigns to mineral K consumption defendable. One has also to bear in mind that the standardised indicator does not yet include societal priorities that can be incorporated through weighting factors.

The results also suggest that the spinach production system in the County of Borken contributes overly to global warming and to eutrophication and impairs soil quality. Further analysis of these impacts reveals that nearly 75% of the system's global warming potential stem from fertilisation (half direct field emissions on lime application and N fertilisation and half indirect emissions from fertiliser production and distribution). The remaining greenhouse gases stem from production and maintenance of agricultural machinery (~15%) and diesel and electricity requirements for field operations (~10%). Two thirds of the terrestrial eutrophication potential are due to field NH_3 emissions, diesel combustion and field emissions of NO each contribute another 10%. Nearly 70% of the marine eutrophication are due to field NO_3 losses and 30% stem from field P losses. The re-deposition of gaseous emissions plays a negligible role. Heavy metal accumulation has been recognised as a major long-term threat to arable soils in Germany (UBA, 2001a; UMK-AMK-LABO-AG, 2000), with copper, zinc and lead being the most critical ones. The metals in danger of accumulation in the studied spinach system are chromium, which originates almost entirely from lime fertilisers, and lead, stemming half from calcium ammonium nitrate fertiliser and half from lime. Soil organic matter decomposition in spinach is due to intense cultivation (ploughing two times a year, cultivator use) in combination with lacking organic fertilisation. Soil compaction is mainly (70%) caused by narrow and high pressure tyres used to apply fertilisers and spray.

Table 5.2 Indicators and indicator standardisation for the case study (standardised Indicator *std* I_S = *NF* x *SF* with *NF* = A_{tot} / I_{tot}). For documentation of severity factors see Table 3.2, for indicator calculation refer to Annex I.

Indicator* Name	Assessment scale	Unit	Value I_S ('unit' $ha^{-1} yr^{-1}$)	Standardisation: Normalisation factor I_{tot} ('unit' yr^{-1})	A_{tot} (ha)	*NF* ('unit'$^{-1}$ ha yr)	I_S x *NF* (–)	Severity factor *SF* (–)	*std* I_S (–)	Technical Notes
Soil loss/formation ratio	Field	dimensionless	1.10	0.96	1.0 †	1.05	1.15	1.0	1.10	1
Soil compaction index	Field	dimensionless	1.74	1.68	1.0 †	0.60	1.04	1.7	1.74	2
Proton input/output ratio	Field	dimensionless	0.98	1.67	1.0 †	0.60	0.59	1.8	0.98	3
Heavy metal accumulation index	Field	dimensionless	3.44	2.48	1.0 †	0.40	1.39	2.5	3.44	4
Soil OM output/input ratio	Field	dimensionless	2.39	1.09	1.0 †	0.92	2.19	1.1	2.39	5
Non-renewable energy demand	World	GJ prim. en.-eq.	51.7	$35.6*10^9$	$13.1*10^9$ a	0.04	1.89	1.1	2.10	6
Mineral potassium consumption	World	t K_2O-eq.	0.31	$26.5*10^6$	$13.1*10^9$ a	491	152.23	1.1	169.14	6
Land naturalness degradation	Region	dimensionless	0.80	$91*10^3$	$141.6*10^3$ b	1.55	1.24	1.2	1.45	7
Water consumption	River basin	m^3	517.1	$31.6*10^9$	$19.9*10^6$ c	0.001	0.32	2.5	0.81	8
Emission of greenhouse gases	World	t CO_2-eq.	7.16	$30.2*10^9$	$13.1*10^9$ a	0.43	3.09	2.2	6.75	9
Emission of NM VOCs	Continent	t NMVOC	0.002	$16.8*10^6$	$982.2*10^6$ d	60.0	0.12	2.9	0.35	10
Emis. eutrophying substances (terrestrial)	Continent	t NO_X-eq.	0.087	$44.9*10^6$	$982.2*10^6$ d	21.8	1.90	2.0	3.90	11
Emis. eutrophying substances (marine)	Marine catchment	t PO_4-eq.	0.041	$1.6*10^6$	$85*10^6$ e	51.7	2.12	1.1	2.39	12

* Refer to Annex I for indicator descriptions.

† On-site impacts referenced to one ha of specific land use.

[a] Global land area (excluding permanent ice. FAO, 2001; Olsen et al., 1983).

[b] Land area of County of Borken (LDS, 2001).

[c] Land area of Rhine-Maas basin (WRI, 2003b).

[d] Land area of Europe (including Russian Federation West of Urals; FAO, 2001; Stolbovoi et al., 2002).

[e] Land area of North Sea catchment (OSPAR, 2000).

Technical Notes

1 *Underlying goal*: Protect soil fertility. *Target*: Erosion rate less than soil formation rate. *Data source and method*: Own calculations (Table A.1, Annex II), erosion rates after Universal Soil Loss Equation (Renard et al., 1997) as adapted for Germany by Schwertmann et al. (1987) and Hennings (2000). Soil formation rates of 1 t ha^{-1} yr^{-1} assumed (average rates given by Troeh et al., 1998).

2 *Underlying goal*: Protect soil fertility. *Target*: No exceedance of soil mechanical resistance (equivalent to pressure at which, at given soil moisture, soil pore volume declines below 8% (Drew, 1992, Horn et al., 1996; Paul, 1999)) *Data source*: Own calculations (Table A.2, Annex II); expert judgement (Uppenkamp, 2003, pers. comm.); machinery and tyre specifications from manufacturers' catalogues

3 *Underlying goal*: Protect soil fertility. *Target*: Proton input rate less than proton output rate. *Data source*: Own calculations (Table A.3, Annex II). Fertiliser use and nutrient off-take from official local statistics (LDS, 2001; LK WL, 2002) and expert judgement (Ferdinand Pollert, 2003, pers. comm.). Atmospheric deposition rates of acidifying substances from EMEP, 2002b.

4 *Underlying goal*: Protect soil fertility. *Target*: Input rates of Cd, Cr, Cu, Hg, Ni, Pb and Zn less than their output rates. *Data source*: Own calculations (Table A.4, Annex II), fertiliser use and yield data from official local statistics (LDS, 2001; LK WL, 2002) and expert judgement (Ferdinand Pollert, 2003, pers. comm.). Plant heavy metal uptake from own data, Bergmann (1992), Delschen and Leisner-Saaber (1998), and UBA (2001). Atmospheric deposition and leaching rates of heavy metals from UBA (2001a) and EMEP (2002b).

5 *Underlying goal*: Protect soil fertility. *Target*: Soil organic matter (SOM) decomposition rate less than input rate. *Data source*: Own calculations (Table A.5, Annex II), SOM losses and reproduction from Leithold et al. (1997). Organic fertiliser use, yields and harvest residues from official statistics (LDS, 2001; LK WL, 2002) and expert judgement (Ferdinand Pollert, 2003, pers. comm.).

6 *Underlying goal*: sustainable resource use (i.e. prevent rapid and total depletion). *Target*: Consumption rate decreasing at a decennial rate to 90% of the previous decade (base-line year 1999). Based on the premise that (a) a sustainable use would not allow for complete depletion and (b) extraction should be phased out subsequently (power law) to allow for substitution or other adjustment (Dresselhaus and Thomas, 2001). *Data source*: Energy consumption from EIA (2001), K consumption from USGS (2001). No mineral P fertiliser used in case study system. Lime reserves are vast and therefore not regarded limiting to sustainability.

7 *Underlying goal*: Nature conservation (leave habitat/land resources to wildlife). *Target*: Diverse cultural landscape capable of sustaining human land use and biodiversity as defined by SRU (1985, 1994), i.e. 10 % (semi-) natural habitat within a diverse landscape with largely extensive agriculture. *Data source*: Naturalness Degradation Potential for different land uses from Brentrup et al. (2002), landscape type and infra-structure data from official statistics (LDS, 2001; Kreis Borken, 2002)

8 *Underlying goal*: Nature conservation (leave habitat/resources to wildlife). *Target*: Human water abstraction not threatening ecological water flow requirement as defined by Smakhtin et al. (2004). *Data source*: German water consumption assumed representative for the total Rhine-Maas catchment area. Human population figures in catchment from WRI (2003b), multiplied with Per capita water consumption from UBA (2002).

9 *Underlying goal*: Stop global warming. *Target*: Reduce energy related greenhouse gas emissions to stabilise atmospheric CO_2-concentrations at 450 ppmv in 2100 (cf. IPCC, 2001).

Data source: Global CO_2 emissions from fossil fuel combustion and cement manufacturing from WRI (2003a). Global N_2O and CH_4 emission estimates from EDGAR (1995 data; changes between 1995 and 2000 are considered negligible; Olivier et al., 2002). (CO_2 emission from the decomposition of soil organic matter not included due to lack of consistent data).

10 *Underlying goal*: Protect human health, prevent vegetation damage.*Target*: For human health protection: Less than 0.1 ppm*hours exceedance of AOT60 (8-hour average) (WHO-EH; Amann and Lutz, 2000). The target to prevent vegetation damage was assumed met if that for human health is met (Amann and Lutz, 2000). *Data source*: NMVOC emission from EMEP (2002b).

11 *Underlying goal*: Protect sensitive terrestrial ecosystems. *Target*: Exceedance of critical loads critical loads in less than 5% of ecosystems (Posch et al., 1995; UBA, 1996). *Data source*: NH_3 and NO_X emission from EMEP (2002b).

12 *Underlying goal*: Protect sensitive aquatic ecosystems. *Target*: Reduction of nutrient input by 50%, as compared to 1985 (CONSSO, 2000). *Data source*: North Sea catchment N and P emissions calculated from per ha emission data (EEA, 2000) and catchment size information (OSPAR, 2000) (1990's data). Atmospheric N deposition (1998 data) from EMEP (2002a). Target levels for freshwater ecosystems were assumed to be met if those for marine ecosystems are met.

We here again emphasise that the method presented here does not allow for statements on the overall sustainability of the whole land uses system: First of all, the sustainability valuation takes place on the level of the individual indicator. Second, the method concentrates on negative impacts and does not quantify positive contributions (e.g. maintenance of cultural landscapes through agriculture). If these positive contributions were conceptualised as benefits and the negative impacts as costs, we might find these costs quite acceptable compared to the produced benefits. The fact that single indicators of the analysed system are in the unsustainable realm should thus be interpreted as an indication of action hot spots, but it does not allow for inferences on the whole system's sustainability or unsustainability.

In the case study, the normalisation factors, *NF*, show much greater spread than the severity factors, *SF*. The final standardised indicator values are highly correlated with the normalised indicator values, $I_S \times NF$, but show no correlation with the severity factors.

The same is true for the weighting factors, as Table 5.3 shows: The weighted results are highly correlated with the unweighted ones ($r > 0.99$). (This correlation is supported by the large value pairs for K consumption. Deleting these extreme values, weighted and unweighted values are still correlated with $r = 0.94$ and $r = 0.64$ for weighting factor sets A and B, respectively).

Interestingly, both weighting factor sets largely concur in which issues are more and which ones are less important than the average: Both sets judge soil compaction, fossil fuel consumption, water consumption and eutrophication as more important and soil

Table 5.3 Effect of weighting on standardised indicator values. Weighting factors used: (A) frequency of occurrence in the literature (percentage); (B) score for importance assigned by local growers (see text for details). To make sets comparable, the original values of each set were transformed by dividing them by the average of the set.

Issue	Set of weighting factors				Standardised indicators values		
	Set A, Frequency in literature		Set B, Growers' appraisal		unweighted	weighted with	
	%	transformed	score	transformed	(Table 5.2)	Set A	Set B
Soil loss	50	(2.1)	4.2	(1.3)	1.1	2.3	1.4
Damage to soil structure	34	(1.2)	4.2	(1.2)	1.7	2.1	2.1
Acidification/alkalinisation	23	(0.8)	3.0	(0.9)	1.0	0.8	0.9
Accumulatio of contminants	16	(0.5)	3.0	(0.9)	3.4	1.7	3.1
Depletion of soil organic matter	23	(0.8)	4.2	(1.2)	4.2	1.9	2.9
Fossil fuel consumption	32	(1.1)	3.6	(1.1)	2.1	2.3	2.3
Consumption of minerals	9	(0.3)	3.6	(1.1)	169.1	50.7	186.1
Land occupancy	7	(0.2)	1.8	(0.5)	1.5	0.3	0.7
Water consumption	30	(1.0)	3.6	(1.1)	0.8	0.8	0.9
Global warming	20	(0.7)	2.6	(0.8)	6.8	4.7	5.4
Summer smog/ground level ozone	5	(0.2)	2.6	(0.8)	0.3	0.1	0.3
Terrestrial eutrophication	68	(2.3)	4.0	(1.2)	3.9	9.0	4.7
Marine eutrophication	68	(2.3)	4.0	(1.2)	2.4	5.5	2.9

Table 5.4 Uncertainty ranges of the non-standardised and standardised indicator values of three indicators, derived by stochastic simulation.

Indicator	Uncertainty ranges (%) Non-standardised indicator value		Standardised indicator value	
	lower	upper	lower	upper
Soil loss/formation ratio	–30	+30	–65	+55
Emission of greenhouse gases	–40	+20	–30	+100
Emission of eutrophying substances (aquatic)	–50	+70	–65	+230

acidification, heavy metal accumulation, land occupancy, global warming and ground level ozone as less important than the average. Only two of twelve issues – soil organic matter depletion and mineral K consumption – are judged differently in the two sets of weighting factors. In both of these cases, the weighing based on the frequency of occurrence in the literature leads to less, the farmers' appraisal to higher-than-average weighting.

The uncertainty ranges for the standardised indicators, obtained by stochastic simulation, are shown in Table 5.4. For all three indicators, the uncertainty of the standardised value ($std\, I_S$) was by a factor of 2.0 to 2.5 higher that that of the non-standardised indicator value (I_S). Although these results cannot be extrapolated to the other indicators, for which no simulations were carried out, they do indicate that the standardisation procedure inflates uncertainty substantially. As tracking the uncertainty through the standardisation process reveals, this inflation of uncertainty was mainly caused by the division operations, especially, where the uncertainty of divisors was high.

Chapter summary

We introduced an evaluation scheme for impact indicators, consisting of a standardisation and valuation procedure. It links into a larger framework for sustainability assessment of land use systems, as presented in Chapter 1.

The standardisation procedure comprises a normalisation step, to make different indicators comparable, and a severity weighting step, to account for the differing target exceedances in different issues. The sustainability valuation procedure assigns the standardised indicators to three discrete sustainability classes. This methodology was designed to meet four specific requirements, namely: (1) to be applicable to environmental, social and economic indicators alike; (2) to allow for comparing very different land use systems; (3) to account for the specific spatial scale at which different issues emerge; and (4) to be clear about the descriptive and normative elements of the evaluation.

This methodology was used to evaluate an indicator set for the environmental impact of a spinach production system in North Western Germany. The results indicate that mineral resource consumption (P and K fertilisation) is the most pronounced impact. The results also suggest that the system contributes overly to global warming and to eutrophication and impairs soil quality. These findings can probably be generalised to most intensive crop production systems in temperate Europe.

The case study results also suggest that the differences between standardised indicators are most strongly determined by the differing relative contribution of the land use system under investigation to the overall problem, as described by the normalisation factor. Relatively small impact had both the severity of the underlying issue (as expressed by the severity factor) and differing societal priorities (as described by two tentative weighing factor sets).

Standardised indicators 'inherit' uncertainty from the input data through error propagation. In the case of three indicators that were analysed through stochastic simulation, the uncertainty of the standardised indicator was by a factor of 2.0 to 2.5 higher than that of the non-standardised indicator. This inflation of uncertainty throughout the standardisation procedure was caused by division and multiplication operations with uncertain figures.

Appendix to Chapter 5

Impact inventory model

The impact inventory used the following model:

$$\mathbf{E}_{m \times n} = \mathbf{A}_{m \times m} \times \mathbf{B}_{m \times n} \tag{5.A.1}$$

where

$\mathbf{E}_{m \times n}$ is an *m* row *n* column matrix in which each row denotes an activity and each column an impact. ‘Activities’ are all items listed under ‘Inputs’ and ‘Production’ in Figure 5.1, and their sub-categories (not shown; e.g. different fertilisers under the ‘fertiliser’ heading). ‘Impacts’ are all emissions, non-renewable inputs etc., which are in Figure 5.1 listed in the boxes ‘Emissions’, ‘Human and eco-toxicity’, ‘Resource consumption and Occupancy’, ‘Impact on soil’ and ‘Cost’.

$\mathbf{A}_{m \times m}$ denotes a diagonal matrix containing in its principal diagonal the number that each of the *m* activity is carried out or applied in the production system (e.g. the number of cultivation passes or kg of a certain fertiliser applied).

$\mathbf{B}_{m \times n}$ is an impact factor matrix that contains in each column the activity-specific impact per unit of the *m* different activities, e.g. kg of CO_2 emitted per litre diesel fuel used. The data used in the impact factor matrix are documented below.

The vector of column-sums

$$\mathbf{v} = \left(\sum_i^m e_{i1} + \sum_i^m e_{i2} + \ldots + \sum_i^m e_{in} \right) \tag{5.A.2}$$

contains the totals of each of the *n* impacts, e.g. the total CO_2-emission of the land use system under investigation.

Impact data

Impact factors used in the inventory model were taken from the following sources:

Energy consumption and emission data for the production and transportation fertilisers was taken from Patyk and Reinhardt (1997), solid waste generation and water and land consumption from the TEMIS 2.18 data base (Öko-Institut, 1998). Heavy metal contents of fertilisers are from UMK-AMK-LABO-AG (2000) and UBA (2001). Fertiliser costs are average local retailer prices in 2001.

Gaseous emissions from N-fertiliser application were calculated after Bouwman et al. (2002a, b), using additional factors published in IFA/FAO (2001). Emissions from lime fertiliser application are from Patyk and Reinhardt (1997). Nutrient losses via run-off and leaching were calculated after Auerswald (1989), Auerswald and Weigand (1999), Gäth and Wohlrab (1994) and Werner et al. (1991), as described in Chapter 4.

Energy consumption for the production of pesticides was taken from Gaillard et al. (1998) as published by Brentrup (2003). Related emissions and resource consumptions were taken from Patyk and Reinhardt (1997) complemented by data from Öko-Institut (1998), as for fertilisers. Pesticide costs are average local retailer prices in 2001.

Machinery emission and resource consumption due to production and maintenance were taken from Gaillard et al. (1998) as published by Brentrup (2003). Related emissions and resource consumptions were taken from Patyk and Reinhardt (1997) complemented by data from Öko-Institut (1998), as for fertilisers. Total machinery life cycle emissions were attributed according to economic life-spans from KTBL (2002), from where also diesel consumption, cost, labour demand and duration of operations were taken. For harvest and transportation these information were taken from the third party contractor (Stephan Pothmann, 2003, personal communication). Costs include machine depreciation (including interest), maintenance, insurance and lubricants. Costs for spinach transportation from field to factory were not included, as they are not payable by the growers. For all machinery, storage is excluded in both environmental impacts as well as cost, as a farm's endowment with buildings were not considered variable or affected by spinach production and is therefore not within the scope of the study (see Figure 5.1). For irrigation, environmental impacts due to machinery production and maintenance were accounted for tractor use, but not for the irrigation equipment itself, due to lacking data.

Farm endowment with machinery and implements, machinery specifications, machine weight, tyre specifications and tyre inflation pressures reflect common local conditions according to expert opinion (Dr. Norbert Uppenkamp, 2003, personal communication), backed by data collected on pilot farms and published producer information (tyre and machinery catalogues).

Emissions and resource consumption/occupancy due to electricity, diesel and other fuel production, distribution and combustion were taken from Patyk and Reinhardt (1997) complemented by data from Öko-Institut (1998).

The acidification potential of fertilisers (N fertilisers only) and atmospheric deposition (SO_4 and total reduced and oxidised N) was determined stoichiometrically based on H^+-equivalents (Van Breemen et al., 1984). It was assumed that: (a) N fertilisation was subject to NH_3 volatilisation, calculated after Bouwman et al. (2002a) and IFA/FAO (2001); (b) NH_X was fully nitrified and only NO_3 was taken up by plants; (c) only 50% of the NO_3 taken up by plants lead to proton consumption (unbalanced cation-anion balance because of excess cation uptake). Denitrification was estimated with Equation 4.3, as described in Chapter 4.

Costs for land use assumed base on average local lease (Ferdinand Pollert, 2003; personal communication). Note, that CO_2 emissions do not include C from the decomposition of soil organic matter, because no reference data of sufficient quality were available.

CHAPTER 6

Recommendations for spinach production in Borken

Environmental impacts: Sources and mitigation options

This chapter explores the findings from Chapter 5 in some more detail with the aim of quantifying reduction potentials and making recommendations for spinach production in the County of Borken.

As discussed in Chapter 5, environmental impacts with a standardised indicator value greater than one indicate a threat to sustainability. In our case study, these are the following ten impacts (standardised indicator values in parentheses):

A. Soil related issues

- Soil loss (1.1)
- Damage to soil structure (1.7)
- Accumulation of contaminants (heavy metals) (1.7)
- Depletion of soil organic matter (2.4)

B. Resource related issues

- Fossil fuel consumption (2.1)
- Consumption of minerals (potassium) (169.1)
- Land occupancy (1.5)

C. Emission related issues

- Emission of greenhouse gases (6.8)
- Terrestrial eutrophication (3.9)
- Marine eutrophication (2.4)

The total environmental impact (i.e. the sum of the standardised indicator values) is dominated by the *Resource related issues*, because of the very high value for mineral potassium consumption (this was discussed in detail in Chapter 5). Excluding it, the *Resource related issues* contribute least to the total impact and the *Emission related* ones most.

Sources of environmental impacts

In order to identify mitigation options we first analysed the sources and drivers of the environmental impacts. Using the impact inventory model introduced in Chapter 5, we determined the percentage contribution of the different agricultural activities to (a) each individual indicator value and (b) the total environmental impact. In the model, the term 'activity' refers both to the use of inputs, such as fertiliser, and to actual operations, such as ploughing (for details refer to the Appendix to Chapter 5).

The percentage contribution of an activity was determined by deleting it from the model (i.e. setting the pertinent row of the activity matrix to zero) and recording the change in indicator values and total impact, respectively. The contributions of the different activities to the individual indicators are shown in Table 6.1 a and the contribution to the total environmental impact in Table 6.1 b.

Mitigation options and reduction potentials

In the following section we discuss mitigation options for the different environmental impacts and try to quantify the reduction potentials for the different impacts. Results are summarised in Table 6.2.

Soil loss – water erosion

Soil erosion was estimated with the German adaptation of the Revised Universal Soil Loss Equation (RUSLE; Renard et al. 1997). Given constant rainfall erosivity (R-factor in the RUSLE), erosion is most pronouncedly influenced by the field topography (LS-factor), which could vary by two orders of magnitude (influencing the RUSLE result to the same degree), and the soil type (K-factor). Both factors can be managed by field selection (topography) and in part also by introducing buffer strips to divide the erosive slope length. Direct management as relating to soil cover (C-factor) and direction of tillage (P-factor) has a less pronounced influence on RUSLE estimates.

The SL-factor could be reduced by about 80% through avoiding sloping fields, the C-factor by about 25% through ensuring sufficient soil cover before and after the spinach crop. In total, the standardised indicator for erosion could be reduced by over 80% to below 0.2.

Table 6.1 a Percentage contributions of different input uses and agricultural operations to individual indicators. '–' denotes values <0.1%, empty cell denote no recorded impact.

		Soil loss	Damage to soil structure	Soil acidification	Accum. of contaminants	Depletion of soil org. matter	Fossil fuel consumption	Consumpt. of minerals (K)	Land occupancy	Water consumption	Global warming	Summer smog/ozone	Terrestr. eutrophication	Marine eutrophication	Costs	Labour
N fertilisers	CAN			42	4		7			5	27	3	26	31		6
	UAN			58			9			6	29	3	58	39		6
K fertiliser					1		6	100		<1	3	1	<1			10
Lime					95		2			<1	11	<1	<1			3
Pesticides							–		–	–	–	–	–			13
Plough							9		–	<1	4	16	2		15	11
Cultivate			13				6		–	<1	3	7	1		8	6
Sow			8				12		–	<1	5	14	2		11	11
Fertiliser application			35				3		–	–	1	4	1		7	3
Spray pesticide			30				5		–	<1	2	4	<1		13	5
Irrigate							8		–	87	4	5	1		6	23
Harvest, field transport			14				14		–	<1	3	21	3			
Road transportation							19		–	<1	8	18	3			
Fieldmen's cars							<1		–	–	<1	2	<1			
Landuse*		100				100			100					30	40	
Sum		100	100	100	100	100	100	100	100	100	100	100	100	100	100	100

* Including landuse-related erosion and run-off.

Table 6.1 b Percentage contributions of different input uses and agricultural operations to the total environmental impact. Except for the first row, the effect of K consumption was excluded because of its overriding magnitude.

		Soil loss	Damage to soil structure	Soil acidification	Accum. of contaminants	Depletion of soil org. matter	Fossil fuel consumption	Consumpt. of minerals (K)	Land occupancy	Water consumption	Global warming	Summer smog/ozone	Terrestr. eutrophication	Marine eutrophication	Sum
Total	with K	1	1	1	1	1	1	87	1	<1	3	<1	2	1	100
	without K	4	7	4	7	9	8		6	3	26	1	15	9	100
N fertilisers	CAN			2			1				7		4	3	17
	UAN			2			1				8		9	4	23
K fertiliser											1				1
Lime					6						3				9
Pesticides															<1
Plough							1				1				2
Cultivate			1								1				2
Sow			1				1				1				3
Fertiliser application			2												3
Spray pesticide			2								1				3
Irrigate							1			3	1				5
Harvest, field transport			1				1				1				4
Road transportation							2				2				4
Fieldmen's cars															<1

Table 6.2 Summary of mitigation potentials for single environmental impacts (see text for details). Effects on the total environmental impact are given with and without including mineral potassium consumption, because of its overriding magnitude

Issue	Standardised indic. value			Mitigation options	Reduction of total envir. Impact (%)	
	actual (—)	potential (—)	reduction (%)		K consumption incl.	excl.
Soil loss	1.1	0.2	82	Avoid sloping fields. keep soil covered before / after spinach	0	4
Damage to soil structure	1.7	1.2	31	Wider tyres / lower inflation pressures for ‘light’ work	0	2
Soil acidification	1.0	–	–			
Accum. of contaminants	1.7	0.4	77	Lime and fertiliser selection	1	5
Depletion of soil OM	2.4	0.0	100	Balance OM inputs and outputs	1	9
Fossil fuel consumption	2.1	1.9	8	Substitute diesel with plant oil	0	1
Consumption of minerals	169	0–169	50	Use recycled K	43	
Land occupancy	1.5	1.4	4	Introduce 10% set-aside	0	<1
Water consumption	0.8	–	–			
Global warming	6.8	5.9	13	Fertiliser selection to minimise N_2O emissions	0	3
Summer smog / ozone	0.3	–	–			
Terrestrial eutrophication	3.9	2.4	38	Fertiliser selection / application to minimise gaseous N losses	1	6
Marine eutrophication	2.4	0.9	61	Minimise leaching (cover crops) Minimise erosion (as above)	1	6

Note, that only water erosion was assessed. Plant damage due to erosion (abrasion) has occasionally been observed in one specific site, but the amount of soil dislodged by wind erosion is assumed to be negligible.

Damage to soil structure – compaction

Soil compaction is determined by two components: The pressure put onto the soil surface by agricultural machinery and the soil’s mechanical resistance against this pressure at a given soil moisture.

The soil’s mechanical resistance depends mainly on soil texture and the organic matter content. The effect of texture is dominant in soils typically occurring in County of Borken (Klaus Seidel, 2001; personal communication).

The soil pressure caused by machinery is mainly due to operations carried out with light duty (62 kW) tractors by the growers, i.e. cultivator use and pesticide and fertiliser application. In the standard production model it is assumed that these tractors have narrow rear tyres (12 inches/310 mm) with high inflation pressures (240 kPa). The above operations are responsible for 78% of the compaction potential, another 8% is caused by sowing. The contribution of harvesting and on-field transport is 14%. Ploughing with an

88 kW tractor (540 mm rear tyres at a pressure of 90 kPa) does not contribute to soil compaction.

Technical optimisation potentials for the harvesting equipment have largely been realised during the past few years (improved tyres and transportation system) and there seems to be little room for further technical improvements. There is, however, a technical potential for improving the machinery used by growers, i.e. using wider tyres and load-adjusted inflation pressures. This could reduce the standardised indicator for soil compaction by about 30% to 1.2.

Avoiding soils with a less stable texture, such as pure sands and silty sands, could substantially reduce soil compaction. The flexibility to avoid wet field conditions is usually limited, because all field operations need to be carried out timely and on schedule with little temporal tolerance. In addition, soil moisture is maintained at 70% of the field capacity or above to ensure optimum conditions for spinach.

Soil acidification

Under present conditions, soil acidification is not in the unsustainable realm (this holds as long as acid inputs do not exceed 13.5 kmol H^+-eq. ha^{-1} yr^{-1}). Soil pH is controlled regularly and can be adjusted through liming before the spinach crop. Avoiding HN_4 fertilisers could reduce the proton input substantially.

Accumulation of contaminants – heavy metals

95% of the contamination potential stems from high chromium (Cr) and lead (Pb) loads in lime fertilisers. CAN contains Pb, which contributes 4% to the total contamination potential. The heavy metal contents of different types and sources of fertilisers differ greatly: The Cr contents of limestone calcium carbonate (*Kieselsaurer Kalk*) are about 30 times lower than those of lime from smelting slag (*Konverterkalk*). However, contents also differ as much as by factor 3 – 10 between different sources of the same type (LABO-AG, 2000). NH_4 based N fertilisers contain substantially less Pb than CAN. Selecting lime and N fertilisers for low Cr and Pb contents could hence reduce the standardised indicator for soil contamination by about 80% to 0.4.

The debate on allowable heavy metal inputs to agricultural soils in Germany is ongoing, especially in relation to legal thresholds for heavy metal loads of composts and sewage sludge (UBA, 2001). Note that there is, compared with the leaching rate, also an excess atmospheric deposition of Pb, i.e. there is a net annual input of 5 – 6 g Pb per ha from outside the agricultural sector.

Depletion of soil organic matter

Soil organic matter (SOM) is decomposed under spinach due to intensive tillage (repeated ploughing and cultivator use). The net loss under spinach cropping is in the order of 1 t ha^{-1} yr^{-1}. Under the present system there is neither sufficient replacement in the year of spinach cropping nor elsewhere during the rotation. Introducing organic matter at the same rate in the year of spinach cropping (or elsewhere in the rotation) could fully compensate losses, i.e. reduce the standardised indicator for SOM loss to zero. For a comprehensive analysis of compost application as an option refer to the scenario COMPOST later in this chapter.

Decreasing tillage intensity is another option for reducing SOM losses. Past experiences suggest that ploughing is necessary to prepare an optimum seedbed for spinach and to rectify soil compaction. There is, however, presently an experimental programme under way to explore the potentials for reduced tillage (refer to Chapter 7 for details).

Historic sporadic measurements from spinach fields suggest that the SOM loss during spinach cropping could be much higher than the default decomposition values used here (adopted from Leithold et al., 1997). To establish a reliable baseline for SOM dynamics, fields should be monitored regularly and long-term trends be computed.

Fossil fuel consumption

17% of the fossil fuel consumption is due to direct fuel use (one half during harvest and transport and one half during field operations carried out by the growers). 24% is consumed as indirect energy input during fertiliser production. About 60% is due to production and maintenance of machinery and implements. Irrigation (electrical pump) accounts for another 8%. Thus, 83% of the fossil energy consumption is embodied in agricultural inputs and machinery.

Decreasing the standardised indicator for fossil fuel consumption to below one would require consumption to be more than halved. This is equivalent to reducing the supply chain energy consumption by at least 40%, which is probably not realistic because the amount of energy used there is beyond the control of Iglo or the growers. We estimate the potential for reducing direct diesel consumption to be below 10%. Refer to the scenario FUEL later in this chapter for the possibility of substituting diesel with plant oil.

Consumption of minerals – potassium

The K consumption in spinach production is entirely due to the use of mineral K fertilisers. Mitigation strategies should focus on finding sources of recycled K. Potential sources of recycled K are composts, manures or beet vinasse, a residual product from the sugar and alcohol industry. Although these will originally also stem from mineral K deposits, using recycled K would keep nutrients within the agricultural system and thereby help slowing down the exploitation of mineral K deposits.

When K is recycled, it depends on the accounting method how the total environmental impact of consumption is divided between and charged to the repeated uses of the same nutrients. We here assumed that for non-renewable resources the impact 'resource depletion' is fully attributed to primary use that motivates the initial resource exploitation. Under theses circumstances, all subsequent uses would not be charged with resource depletion. It is, however, also arguable that recovered nutrients could be set off from the charge on the first use, in which case it does not matter whether the resource is recycled or not. Depending on the accounting method, the reduction potential is 100% (full attribution to first use) to nil (attribution by consumption).

Land occupancy

Although nearly all inputs and field activities are associated with some form of land occupancy (e.g. for production facilities, disposal of production waste etc.), these contribute less than 1% to the total land occupancy of the spinach production system. The main source for land occupancy is the production site itself.

Reducing the standardised indicator for land occupancy would either require greatly reducing the over-all intensity of agricultural use, which is not compatible with irrigated intensive vegetable production; or leaving parts of the field unused. Converting 10% of the production to extensive set-aside (Naturalness Degradation Potential = 0.5; Brentrup et al., 2002) causes the land naturalness degradation to decrease by 5%. Higher reductions would require larger areas of set-aside, which interferes heavily with the production purpose of the land. The mitigation potential is thus very limited – we here assume 5% – and unlikely to exceed a single digit percentage.

Water consumption

Irrigation makes up 87% of the total water use. Another 11% is used during fertiliser production and transportation, the remaining 2% during production of the other inputs. Present water consumption (520 t ha^{-1} yr^{-1}) is not in the unsustainable realm (this holds unless total water consumption does not exceed 640 t ha^{-1} yr^{-1}). For monitoring and documentation purposes it is recommended that growers record irrigation water consumption.

Global warming

59% of greenhouse gas emissions is due to fertilisation (49% N fertilisation plus 10% liming). 11% of the greenhouse gases is emitted during fertiliser production and transportation. Another 18% is due to indirect energy consumption for production and maintenance of machinery and implements. Only 12% of the greenhouse gases originate from direct fuel combustion and electricity use (irrigation pump).

Field emissions of N_2O could be reduced through fertiliser choice (prefer nitrate-based N fertilisers), field selection (ensure good drainage conditions) and through minimising N leaching losses. Total fertiliser induced N_2O emissions could probably be reduced by some 35% (also refer to scenario NITROGEN later in this chapter). This yields a 13% reduction in the total standardised indicator for greenhouse gas emissions, which does not suffice to diminish it to below one. Emissions from other sources are difficult to reduce: CO_2 emissions after liming inevitably occur and liming is necessary to regulate the soil pH for spinach cropping. All other sources of greenhouse gases are associated with the supply chain of agricultural inputs.

Summer smog/ground level ozone

The emission of NM VOCs (2 kg ha^{-1} yr^{-1}) is not in the unsustainable realm (this holds as long as it does not exceed 7 kg ha^{-1} yr^{-1}). 85% of the emission of NM VOCs stems from fuel combustion during field operations (direct energy input).

Terrestrial eutrophication

Two thirds of the terrestrial eutrophication potential are caused by NH_3 volatilisation after N fertilisation. 9% stems from field emissions of NO and another 12% from direct fuel use (diesel combustion). The remaining 12% stems from NO_X emissions along the supply chain of agricultural inputs and machinery.

The reduction potential for NH_3 and NO volatilisation from mineral N fertilisers is in the order of 50%, which could be realised through fertiliser choice (prefer nitrate-based fertilisers) and application mode (timely incorporation). The scenario NITROGEN below analyses these options. The other sources are difficult to manage, except through reducing input quantities. The standardised indicator for terrestrial eutrophication could thus be reduced by some 35% to 2.4.

Marine eutrophication

Two thirds of the marine eutrophication is caused by N leaching and run-off. 30% is due to P losses in eroded soil. P run-off and re-deposition of gaseous N losses are negligible.

Spinach is harvested before physiological maturity and therefore always leaves an N residue. To prevent this residue from being leached, 'end-of-the-pipe' solutions are required, as e.g. establishing a cover crop after spinach harvest: Cover crops sown after spinach take up 30 to 150 kg N ha^{-1} yr^{-1} (see Table 6.5). It is, however, difficult to judge the fate of the N taken up by a cover crop, because it may freeze off during winter and thus subject the taken up N to leaching. Also, N leaching may occur between incorporation of the cover crop and N uptake by the following crop. Assuming that all residual N after the second spinach harvest was taken up and was fully available to the following crop, the marine eutrophication potential could be reduced by 34%.

P losses via erosion could be reduced by preventing soil erosion (see above) and by reducing the P concentration in soils. The latter option would have to focus on field selection for spinach growing, as P is presently not applied to spinach (soil P levels are usually sufficient for spinach). Preventing soil erosion could reduce the standardised indicator for marine eutrophication by another 27%. Preventing both P losses and over winter leaching of N would thus result in a 61% reduction to 0.9.

Labour

Total labour demand for growers is 13.2 hrs ha^{-1} yr^{-1} (excluding harvest and transportation, which are done by a third party contractor). Growers use 57% of their labour input for soil preparation and sowing, 33% for fertilisation and spraying pesticides and 10% for irrigation. The potential for saving time during soil preparation seems limited, because establishing a spinach crop without ploughing has proven difficult in the past. Detailed investigations into reduced tillage are undertaken presently (refer to Chapter 7 for details).

The time requirements for fertilisation and even more so spraying are highly variable. Reducing the number of field-passes for weed control probably bears the largest savings potential. This requires working towards a low weed pressure throughout the entire rotation. Another possibility for saving field passes is the combination of operations.

Cost

Total costs for growers, excluding harvest and transportation, are 1,512 EUR ha^{-1} yr^{-1}. 40% of this cost is land related (opportunity costs or lease, respectively), 24% is fertiliser and pesticide costs (without application), 18% is for soil preparation and sowing, 14% for irrigation. Only 5% of the total cost is for fertiliser application and spraying pesticides.

Assuming a raw material price of 70 EUR (alternatively 50 EUR) per tonne and an annual yield of 42 t ha^{-1}, growers earn 2,940 EUR (2,100 EUR) per hectare. After subtracting the total cost of 1,512 EUR this results in a gross income of 108 EUR (45 EUR) per hour of labour.

Scenarios

The mitigation potentials identified above only look at a single indicator and do not account for trade-offs with other indicators. To test the over-all impact of some of the suggested mitigation options, we develop four scenarios. Each scenario tests specific mitigation options against the standard production scheme (as described in Chapters 1).

Scenario TYRES

Background

The intention of this scenario was to explore the effect of substituting narrow and high pressure tyres used during fertiliser spreading and spraying. Two alternatives were tested: (1) Substituting 12 inches rear tyres with twin tyres (two times 12 inches) on the 62 kW tractor; and (2) using the larger (and heavier) 88 kW tractor for fertilisation and spraying, which is equipped with wider rear tyres (540 mm/22 inches).

Assumptions

Option 1 (twin tyres): Labour demand increases by 0.5 hrs for installation and de-installation of the twin tyres (0.9 hrs per ha at standard field size of 5 ha). The wider tracks reduce the cropped area proportionally by 4%. We assume that the yield loss is in the same order of magnitude. Note, that this is, however, a worst-case assumption: The yield penalty will most probably be less due to edge effects (more vigorous plant growth in rows next to tracks because of reduced competition). Purchasing and maintenance costs for the twin tyres are not accounted for. Neither are Fuel savings due to better traction, life cycle emissions from their production of the tyres and the additional weight on the field, which factors were regarded negligible.

Option 2 (larger tractor with wide tyres): Fuel consumption increases slightly to 2.0 l per ha for spreading fertiliser and spraying because of the larger engine of the 88 kW tractor. Cropping area is reduced proportionally to track width by 3%. Again we assume a worst-case yield loss in the same order of magnitude.

Results

Both options decrease the soil compaction potential: Option 1 by 36% and Option 2 by 29% (Figure 6.1 a shows results for Option 2). However, the standardised indicator value for soil compaction remains above one in both cases. Option 1 (twin tyres) reduces compaction more effectively than Option 1 but there are trade-offs in terms of labour (+7%) and yield (–4%). Negligible trade-offs (below 1%) in Option 2 are related to higher diesel demand. Note, that the methodology used to assess soil compaction (see Chapter 5 and Table A.2) is relatively coarse. Differences between the two options may be an artefact of method uncertainty and insignificant.

Using wider rear tyres for fertilisation and spraying could thus reduce the soil compaction potential by approximately 30%. Using twin tyres could be associated with higher labour demand. Yield penalties are below 5%. The overall environmental impact is reduced by 2%.

Scenario FUEL

Background

This scenario explores the effect of using plant oil (rapeseed oil) as a substitute for diesel fuel for tractors. 'Biodiesel' from rapeseed oil (rapeseed oil methyl-ester) is often discussed as an alternative to fossil fuel but it consumes twice as much fossil energy during production as pure rapeseed oil (Öko-Institut, 2002). Biodiesel is therefore environmentally less favourable than unrefined plant oil, although costs for adjusting tractor engines to biodiesel are considerably lower.

Assumptions

Economics: Costs for adjusting tractor engines increase tractor cost by 5%. Cost for plant oil and (subsidised) "farm" diesel are equal, at 0.60 EUR (reference year 2000). Note, that diesel prices are presently higher.

Plant oil production: Oil seed rape is produced in the Ostholstein area, Northern Germany, according to the local best practice (Box 6.1, Heitmann, 2004, personal communication). For agricultural production, the same equipment as for spinach production was assumed. Yields are 4 t seeds per ha and yr with 93% dry matter contents. The distance to the oil mill is 65 km. Oil content of the seeds is 40% of the fresh weight, oil density 920g per cm^3 (TFZ, 2004). This yields an oil proceed of 435 l per t fresh seeds. According to TFZ (2004) oil production consumes 1 GJ of electricity per t of seeds, which was assumed to be average German energy mix (after Patyk and Reinhard, 1997). Water consumption during rape oil production was assumed to be negligible. Likewise, storage and transportation from the mill to the consumer is not accounted for.

Environmental impacts: On-site (field level) impacts of rape production, such as erosion, compaction, acidification etc., were not accounted for. Off-site environmental impact were standardised by the same methodology used for spinach production (Chapter 5). Water consumption was not relevant, because rape is not irrigated. As opposed to spinach production, ecotoxicity of pesticides was relevant in rapeseed. Based on local spraying practices, a Normalised Pesticide Use Intensity Index (*Normierter Behandungsindex* after Gutsche and Enzian, 2002) for rape of 4.75 was calculated. The regional Normalised Pesticide Use Intensity Index of 4.69 was computed from crop data (Box 6.1) and region specific data cited by Roßberg et al. (2002), assuming a rotation of 33% winter oilseed rape, 50% winter wheat and 17% winter barley. Regional targets for pesticide use reduction were calculated as for spinach (Chapter 3).

Standardised indicators for rape production were determined as described for spinach in Chapter 3. To reference them to a per litre of plant oil basis, the standardised indicators for rape production were divided by the oil yield (1,740 l ha^{-1} yr^{-1}; from data given above). The result was then multiplied with the fuel demand for spinach production. The fuel demand for spinach production was corrected for the difference in energy contents between rapeseed oil and diesel (0.0351 GJ and 0.0364 GJ per litre, respectively).

CO_2 emissions from combustion of the plant oil were not accounted for, because the carbon is photosynthetically fixed. SO_X emissions are two orders of magnitude lower than those of diesel (TFZ, 2004) and were therefore neglected. NO_X, NH_3 and NM VOC emissions were assumed to be the same as for diesel.

Box 6.1 Crop data assumed for growing oilseed rape in Ostholstein.

Tillage

Plough
Sow (drill combined with rototill)
Stubbles left after harvest until drilling of next winter crop

Fertilisation (N – P – K – S; kg ha^{-1} yr^{-1})

	N	P	K	S
Autumn	30 –	26 –	108 –	15
Spring	90 –	0 –	0 –	0
	42 –	0 –	100 –	50
Summer	46 –	0 –	0 –	0
Lime (autumn)	1200 kg CaO			

Pest management		(Name, % of allowed rate applied)*
Autumn	Herbicide	('Nimbus', 83%)
	Herbicide	('Agil', 75%)
	Fungicide	('Caramba', 33%)
	Insecticide	('Fastac SC', 100%)
Spring	Fungicide	('Caramba', 33%)
	Insecticide	('Fastac SC', 100%)
Flowering	Fungicide	('Cantus', 50%)

Yield

4 t ha^{-1} yr^{-1} (93% dry matter)

*'Agil': 100g l^{-1} Propaquizafop
'Cantus': 500g kg^{-1} Boscalid
'Caramba': 60g l^{-1} Metconazol
'Fastac SC': 100g l^{-1} alpha-Cypermethrin
'Nimbus': 250g l^{-1} Metazachlor, 33.3 g l^{-1} Clomazone

Results

Results are shown in Figure 6.1 b. Substituting diesel with rapeseed oil reduces the consumption of fossil fuel by 8%. There are, however, significant trade-offs: The terrestrial eutrophication potential increases by 27%, marine eutrophication by 12%. The land naturalness degradation potential increases by 13%, mineral K consumption by 11%. Due to the spraying regime in oil seed rape, ecotoxicity of pesticides becomes a relevant environmental impact.

In summary, although the use of plant oil reduces the impact on fossil fuel resources negative effects prevail in this scenario: The overall environmental impact increases by 7% due to the production of rapeseed oil, with its high inputs of fertilisers and pesticides. Assuming present (2004) prices instead of those from 2000, plant oil has a competitive advantage against diesel, which could override the higher cost for adjusting tractor engines to plant oil in the long term. Differences in total cost will, however, hardly exceed the ±5% range.

It was mentioned above that the cumulated energy demand (Öko-Institut, 2002) for refined rapeseed oil (‘biodiesel’) is twice as high as that for unrefined rapeseed oil. Biodiesel is thus an even less favourable substitute.

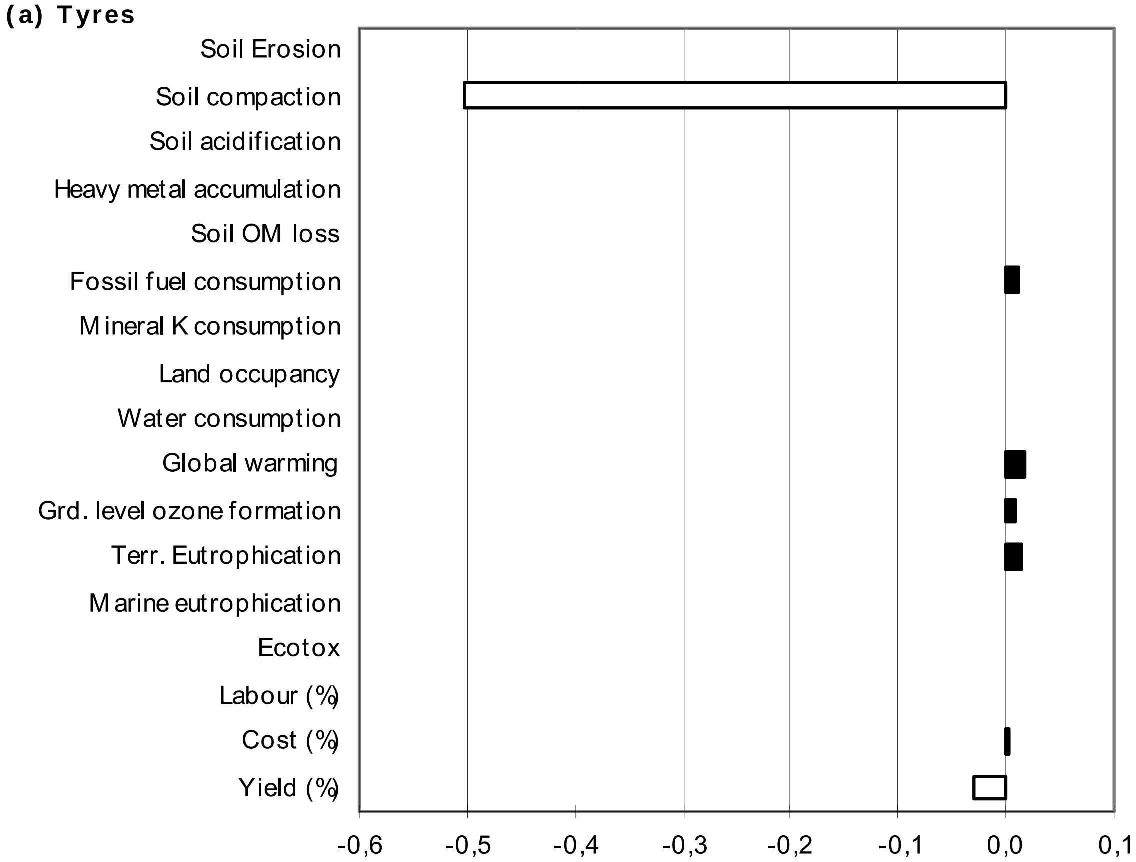
Scenario NITROGEN

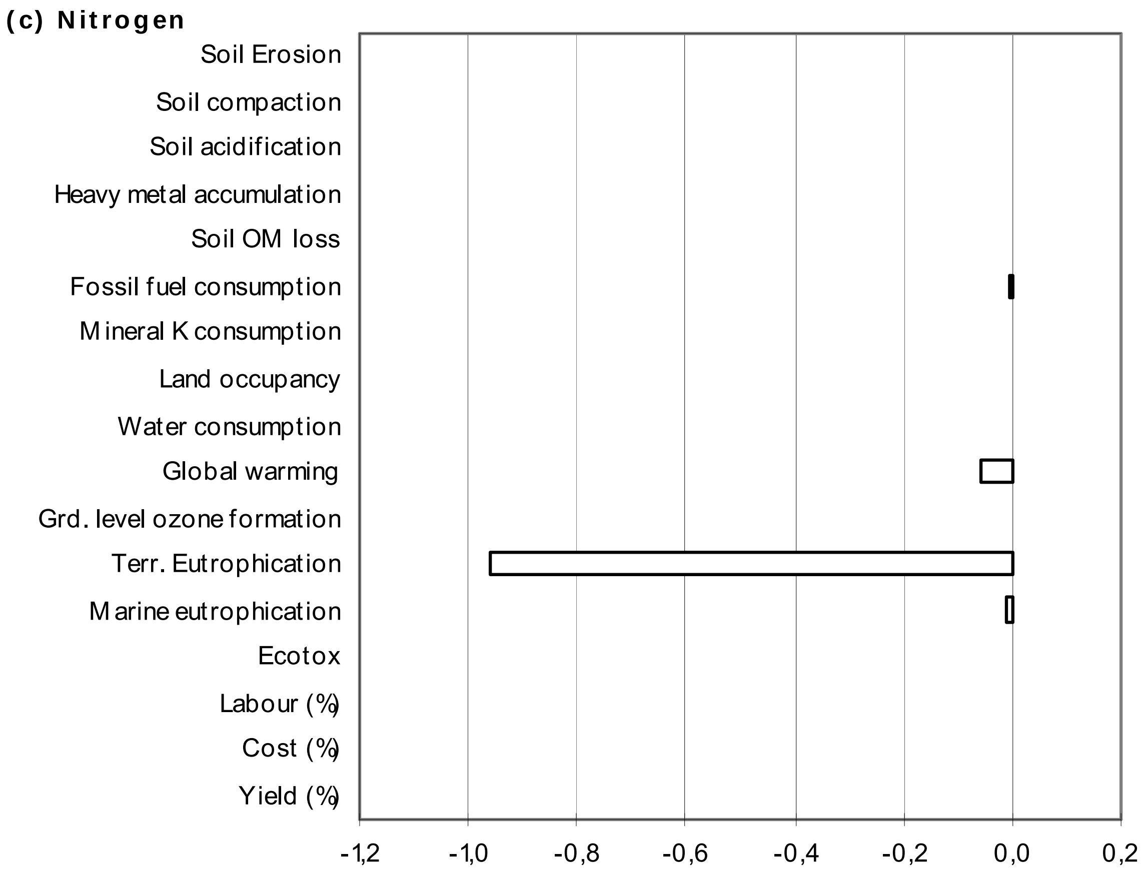
Background

This scenario explores the effects of reducing gaseous N losses through fertiliser choice and incorporation of N fertilisers. The present practice is to spray a liquid base dressing of urea ammonium nitrate (UAN) directly after sowing and apply a top dressing of calcium ammonium nitrate (CAN) after the second leaf has fully emerged. Several options for reducing volatile N losses were tested: (1) incorporating UAN, (2) substituting UAN with CAN, and (3) targeted application with reduced quantity and incorporation of UAN.

Assumptions

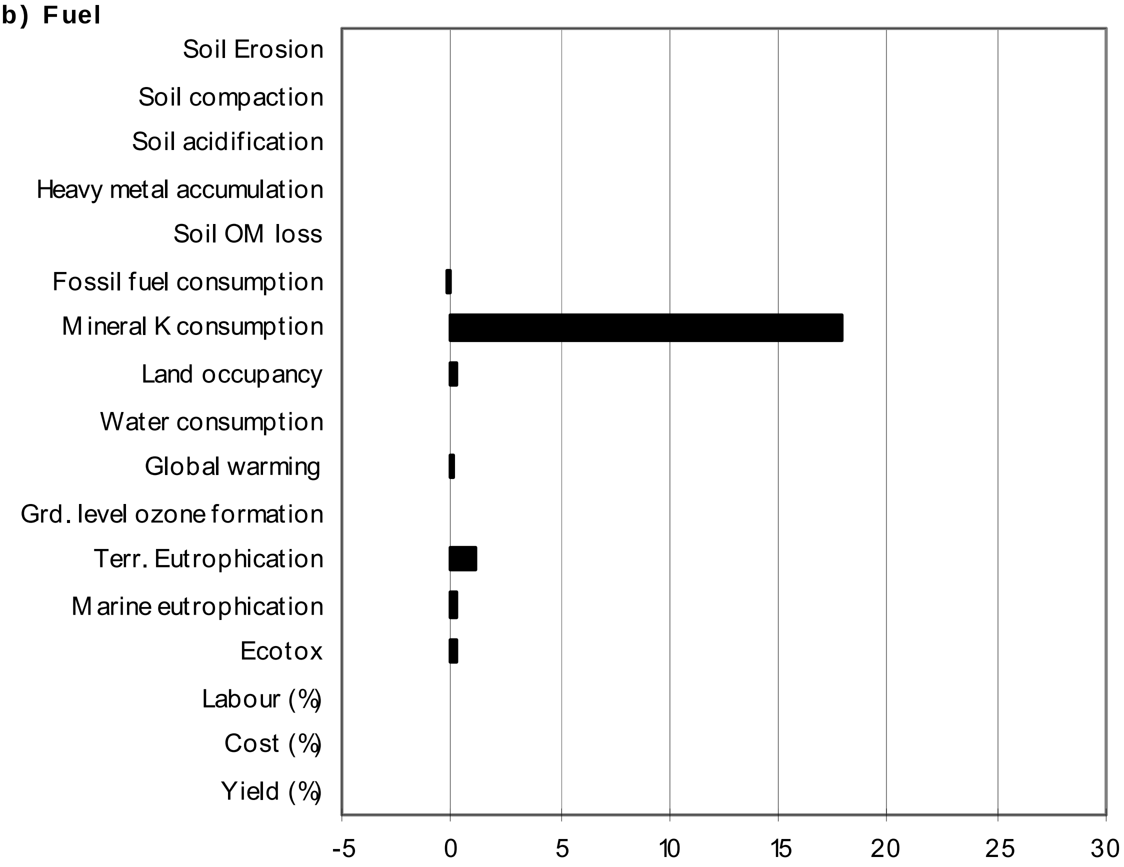
Option 1 (incorporating UAN): Presently, UAN is sprayed a few days after sowing. Reversing this order and spraying UAN directly before sowing automatically leads to incorporation by the drilling machine following the sprayer. The N application rate was slightly adjusted (–5%) to account for the saved N. A potentially herbicidal effect of UAN is not accounted for in this scenario.



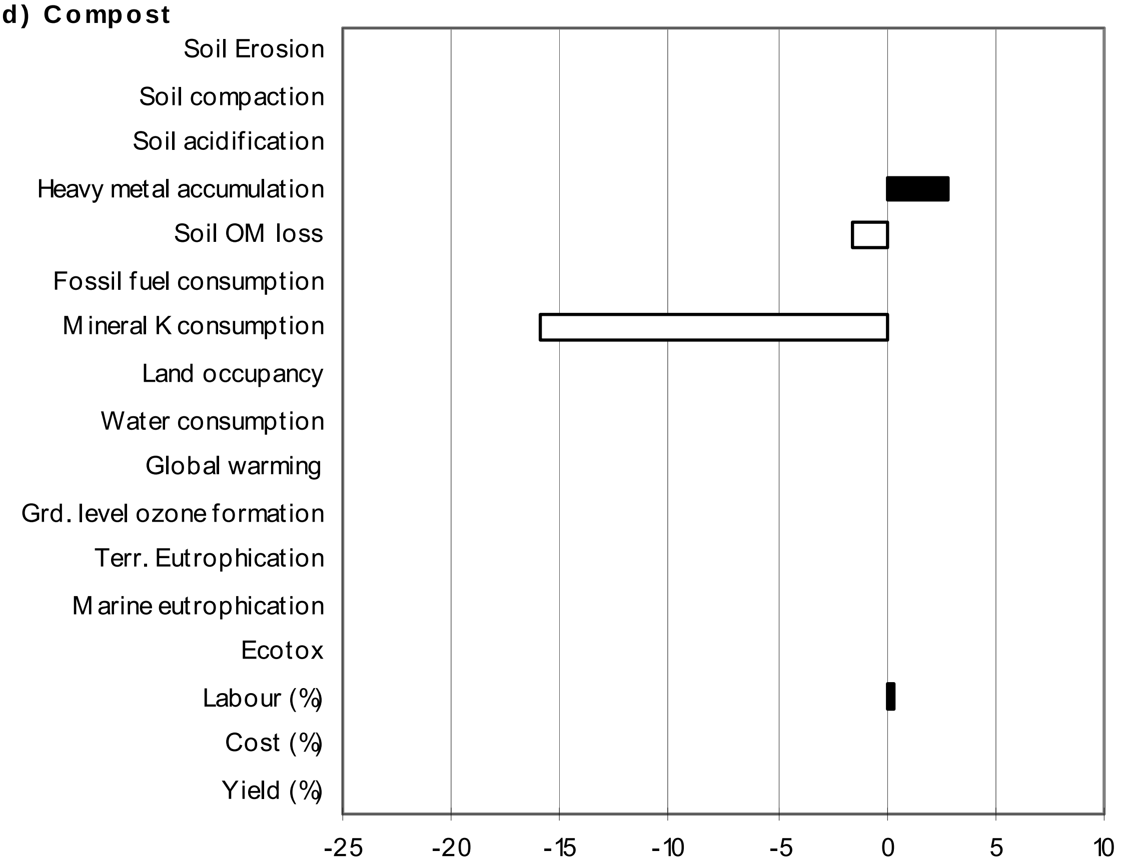


Figures 6.1 a – d Changes in indicator values for the four scenarios compared to the units standardised indicator (dimensionless), changes in economic indicators in %.

(b) Fuel



(d) Compost



standard spinach production scheme. Changes for environmental impacts are given in
Note the different scales in the four plots.

Option 2 (substituting UAN with CAN): UAN could be substituted with CAN, which would be applied with a normal fertiliser spreader before sowing. Again the application rate was slightly adjusted by 5% to account for the saved N.

Option 3 (target application and incorporation of UAN): UAN could be band-sprayed or dripped into the crop row during sowing. It was assumed that bands of a width of half a crop interrow were treated, i.e. the rate was reduced by 50%. This option assumed that a mounted sprayer or dripper is combined with the drilling machine that applies the fertiliser into the row before the drill passes. As no comparable technology was known to take data from, the technical implications of this option (especially costs, labour demand and soil compaction) were not further assessed.

Results

Option 1 – spraying UAN before sowing – results in a reduction of the terrestrial eutrophication potential by 25% (Figure 6.1 c). There is also a slight reduction in greenhouse gas emissions (decreasing indirect N_2O emissions). There are no trade-offs.

Option 2 – substituting UAN with CAN and incorporating it – reduces terrestrial eutrophication by 37%. There are small trade-offs with both labour (+0.5 hrs) and costs (+15 EUR), which are due to differences in fertiliser prices and application techniques.

Option 3 – targeted application of UAN at halved rate and with incorporation – has the most pronounced positive effects: It reduces terrestrial and marine eutrophication by 39% and 46%, respectively, and decreases greenhouse gas emissions (CO_2-equivalents) by 15%. There are also small positive effects on fossil fuel consumption, emission of ozone precursors and water consumption as well as on cost (–27 EUR).

Note again that possible herbicidal effect of applying UAN a few days after sowing in Option 1 and technical implications in Option 3 are unaccounted for.

In summary, preventing gaseous N losses is an effective means of reducing the overall environmental impact: Incorporating UAN would reduce it by 4%, targeted application with incorporation even by 15%.

Scenario COMPOST

Background

This scenario tests the use of compost at a rate of 7.5 t FM (3.1 t DM) per ha once in a one-in-four spinach rotation. This rate was chosen to solely replenish SOM that was lost during spinach cropping and is relatively low compared to typical recommendations of compost suppliers.

Assumptions
Compost is certified green waste (*Grünschnittkompost*), sourced from a local plant (i.e. compost from plant biomass, as opposed to organic household waste). It was assumed that N, K and lime applications were reduced by the plant available nutrients (not the total amount) and the CaO-equivalents contained in the compost. A potentially increased risk of P losses from the field was neglected, because the P input with compost is negligible compared to actual soil P and plant available levels. Positive effects on soil stability and infiltration were not taken into account.

It was assumed that compost was spread with a 20 t manure spreader. The price of compost was 5 EUR per t, including transportation from the compost plant to the field. Life cycle impacts from the production and transportation of compost were assumed to be negligible. Potential positive effects of repeated compost application on crop yield are not taken into account.

Results
Figure 6.1 d shows the result of this scenario. The negative SOM balance of spinach is compensated by compost application at a relatively low rate. It reduces the standardised indicator for soil organic matter depletion by 64% to below one. Also, the consumption of mineral K is decreased by 9%, which has the most pronounced effect on the over-all environmental impact. These positive effects are, however, at the expense of increased heavy metal accumulation (+159%). There are also economic trade-offs: Labour increases by 28% (3.5 hrs), mainly due to loading compost onto the spreader, and costs increase by 5% (48 EUR).

A political disadvantage of compost application is that growers have to account for the full N contents of compost (not only the plant available N fraction) in their nutrient balance sheet, i.e. they incur higher N inputs than with mineral N to achieve the same crop available rate. This would exacerbate the high N surplus that many farms in the area have because of high livestock densities.

Discussion

We explored the mitigation potentials for the ten issues where standardised indicators are above one. In six of them mitigation potentials probably suffice for a reduction to below one. There may, however, be substantial trade-offs with other issues, i.e. improving on one indicator is at the expense of one or several others. No or moderate trade-offs may be expected for reducing soil compaction, soil erosion, heavy metal accumulation and marine eutrophication.

Mitigation potentials for greenhouse gas emissions and terrestrial eutrophication allow for a reduction of 10 – 40%, but without reducing the standardised indicator below one. For land occupancy and fossil fuel consumption there seems to be little room for improvements.

A combination of those management options that seem practicable and are likely to have no or tolerable trade-offs could reduce the total (non-potassium consumption related) environmental impact by some 20%. In spite of this overall improvement, some environmental impacts are likely to remain in a realm where they could threaten sustainability.

Improvements beyond these 20% might require more radical changes to the production system, i.e. moving away from modern intensive agriculture. This has, however, strong implications for the social and economic dimension of spinach production, which will be discussed in Chapter 7.

It is important to note that the above results do not allow for statements on the overall sustainability of the whole production system: First of all, the method used here evaluates sustainability on the level of the individual *indicator*. A genuine weighting of different environmental issues against each other (Chapter 5) was not carried out and the results do therefore not account for differences in the social or political perception of the weightiness of different issues. Improvements in one area at the expense of another (trade-offs) may well be acceptable or the result of an informed choice form a societal or political point of view.

Also note that all management options discussed here are hypothetical, i.e. there are no empirical data to corroborate the assumed effects. Also, scenarios are based on the 'standard' production system outlined in Chapter 1, which uses average data and does not account for variability between fields, farms and years.

Recommendations

Looking at the ten impacts with standardised indicators greater than one we find that they are either linked to nutrients and fertilisation (eutrophication, mineral resource consumption and heavy metal accumulation) or to soil deterioration (SOM loss, heavy metal accumulation and soil erosion). We may call the former the 'nutrient complex', the latter the 'soil complex' (heavy metal accumulation, marine eutrophication and erosion play a role in both complexes). Our above findings indicate that managing the issues in the soil complex most probably would allow for reducing their standardised indicators to below one. Conversely, issues in the nutrient complex are less easy to manage and it is doubtful whether a reduction to an environmentally harmless level is possible within confines of intensive agriculture. Based on our above findings the following recommendations are made.

Recommendations for the nutrient complex

Control the input rates of heavy metals contained in fertilisers. Different fertilisers are problematic with regards to different heavy metals: High loads of Cr and Pb are conveyed with lime from smelting slag (*Konverterkalk*), Cd with mineral P fertilisers, Pb with CAN, Cu and Zn with animal manure and slurry, which in addition can carry substantial loads of organic contaminants from stable disinfectants and veterinary medicine products (UBA, 2001). Sewage sludge has frequently high levels of a range of heavy metals (including Cd, Cr and Pb) and also often carries organic toxicants as well. Composts are often high in Pb, Cu and Zn. The amounts of heavy metals and other impurities contained in fertilisers can vary extremely between different fertiliser types and different sources of the same type (LABO-AG, 2000). In order to attain balanced heavy metal inputs and outputs, loads given in Table 6.3 should not be exceeded.

Use recycled nutrients, especially potassium. Production and transportation of nutrients consume resources, both the mineral resource that a particular nutrient is drawn from, as for K and P deposits, and other resources such as energy and water needed for processing, fixation (N) and transportation. Substituting nutrients from mineral fertilisers with recycled nutrients would reduce the exploitation of mineral K (and P) deposits. At the same time, using locally available sources of recycled nutrients could save energy for refinement and transportation.

For the time being, recycled nutrients often have major drawbacks. These involve (1) chemical properties and impurities, such as heavy metals or salinity; (2) release dynamics that are difficult to control (e.g. N release through mineralisation); (3) difficult handling and costly application because of bulkiness and physical inhomogeneity; and (4) legal regulations requiring the farmer to fully account for the total amount of nutrients contained, whereas the plant available fraction is smaller.

There are, however, also a number of positive side effects with many sources of recycled nutrients, such as additional organic matter, micronutrients or a high pH value. We here recommend investigating the possibility of beet vinasse application.

Minimise gaseous N losses during application. Gaseous losses occur during and after the application of N fertilisers. Compared to the application rate, these losses are rather small but at high application rates they easily reach 10 to 30 kg N ha^{-1} yr^{-1} (Ferm, 1998; Mosier, 2001) or more (Weigel et al., 2000). This may still be tolerable from an agronomic point of view, but it is critical from an ecological/environmental one (Posch et al., 1995; Isermann and Isermann, 1998).

Table 6.3 Heavy metal input rates of different crops grown in Borken, at which inputs exceed outputs. Exceeding these input rates leads to accumulate of heavy metals in the soil (figures account for atmospheric deposition, plant off-take and leaching).

Crop (yield ha^{-1} yr^{-1})	g ha^{-1} yr^{-1}						
Spinach (42 t, 6% DM)	2.1	22	40	0.3	17	[a]	[a]
Standard rotation (per yr) [b]	1.5	34	52	0.8	62	14	359
Grain Maize (9.7 t FM, 65% DM)	0.3	17	10	0.3	26	9	5
Silage Maize (46 t FM, 33% DM)	0.5	25	26	0.3	41	27	259
Cereals (8 t FM, 86% DM) [c]	0.4	12	36	0.3	21	[a]	120
Potatoes (40 t FM, 20% DM)	2.2	[a]	48	[a]	[a]	15	10
Sugar beets (53 t, 20% TM)	0.9	[a]	41	[a]	1	3	117
Herbs [d]	0.2	11	[a]	0.3	15	[a]	[a]

[a]Atmospheric deposition exceeds leaching and plant uptake.
[b]Standard rotation (see Chapter 1, Box 1.1)
[c]50% wheat, 50% barley.
[d]Only leaching; plant uptake neglected.

Sources: Deposition data (national averages) from UBA (2001) except for Cd, Hg and Pb (site-specific on 50 km x 50 km grid) from EMEP (2002). Leaching data from UBA (2001), plant off-take after own data, Bergmann (1992), Delschen and Leisner-Saaber (1998) and UBA (2001).

Table 6.4 Volatile N losses (kg N ha^{-1}) from applying calcium ammonium nitrate (CAN), urea ammonium nitrate solution (UAN) and mixed slurry with and without incorporation. NH_3 losses from CAN and UAN according to Bouwman et al. (2002a), losses from slurry according to Döhler et al. (2002). NO emissions according to Bouwman et al. (2002b).

	Appl. rate (kg N ha^{-1})	CAN		UAN		Slurry [a] broadcast		Slurry [a] trailing hose	
NH_3-N		incorporate							
		N	Y	N	Y	N	Y	N	Y
	35	1	0	3	1	9	4	8	3
	50	1	1	4	2	13	6	11	4
	75	2	1	6	3	20	9	17	6
	100	3	1	8	4	26	12	22	7
	150	4	2	11	6	39	18	34	11
	200	5	3	15	8	53	25	45	15
	250	6	4	19	10	66	31	56	18
NO-N	35	0.6		0.7		0.6			
	50	0.7		0.8		0.6			
	75	0.8		0.9		0.6			
	100	0.8		1.0		0.7			
	150	1.0		1.4		0.7			
	200	1.2		1.9		0.8			
	250	1.5		2.6		0.8			

[a]Mixed cattle/pig slurry with 70% total ammoniacal N. Application at 15°C, incorporation within 4 hours.

NH_3 volatilisation can be reduced through the choice of N fertilisers (avoid NH_3 based fertilisers), application mode (incorporate) and by avoiding application at high summer temperatures. Table 6.4 shows the volatilisation rates for different fertilisers and application modes. In addition, ensuring good soil structure and aeration reduces NO and N_2O emissions.

Reduce nutrient losses after harvest. Spinach is harvested before physical maturity and therefore leaves an N residue in the soil. I.e. a certain proportion (roughly one third) of the initially applied N is left unused by the crop and is prone to leaching after harvest. Due to intensive livestock breeding in County of Borken many farms are also net importers of P and intensive manuring in the past has lead to the build-up of high P levels in soils. Other than N, P is usually not applied to spinach. However, the risk of erosion and surface run-off is elevated under spinach, because the crop repeatedly leaves the soil uncovered or with only little cover. The present N fertilisation strategy has been designed in the early 1990's to best possible match the N supply to crop requirements (Schick, 1992, 1993, 1994, unpublished fertilisation trial data). The potential for reduce N losses by changing the fertilisation strategy thus seems limited.

Planting cover crops after spinach reduces both the N leaching potential, because plants take up water and N, and protects the soil against erosion and thus prevents P losses. It is important to note, that cover crops only shift N loads into the next vegetation period or to the next leaching event after a frost period. It thus depends on the cover crop species, weather and following crop, how much of the taken up N is really preserved from leaching. Table 6.5 shows N uptake of different cover crops.

Another, more fundamental approach to reducing residual N after spinach is by breeding for varieties with high nutrient uptake efficiency and varieties that are less dependent on high soil N levels to reach quality and yield targets.

Consider the risk of P losses during field selection. P losses mainly depend on the soil P content and soil erosion (see Chapter 4). Table 6.6 shows the potential P loss at different P contents and erosion levels.

Recommendations for the soil complex

Establish a reliable baseline and trend for SOM dynamics. Until the beginning of this project there was no regular monitoring of SOM contents. Historical data that was available from sporadic measurements did not yield a consistent picture. Regular measurements and information on field history and management should be used to corroborate and adjust the decomposition coefficients used here (Leithold et al., 1997), which are rather coarse and not specific to the region. Three years of systematic SOM measurements, taken before spinach show remarkably robust results: In all three years the

Table 6.5 N uptake (kg N ha^{-1}) of winter cover crops after spinach until the end of December at different sowing dates and soil mineral N supply (assuming average weather conditions and normal amount of spinach harvest residues).

Sowing date (week)	Fodder radish, winter turnip			Mustard			Rye		
	20 [a]	40 [a]	80 [a]	20 [a]	40 [a]	80 [a]	20 [a]	40 [a]	80 [a]
33	130	140	170	80	90	110	100	110	130
34	110	120	140	60	70	90	80	90	110
35	80	100	120	50	60	70	60	70	90
36	70	80	100	30	40	50	50	60	70
37	50	60	80	20	30	40	30	40	50
38	40	50	60	20	20	30	20	30	40
39	30	40	50	10	10	20	20	20	30
40	<10	<10	<10	<10	<10	<10	10	20	20
41	<10	<10	<10	<10	<10	<10	10	10	20
42	<10	<10	<10	<10	<10	<10	<10	10	10

[a] Soil mineral N in 0–30 cm (kg ha^{-1})
Source: Own data.

mean SOM contents was 2.9%, with 0.025 and 0.975 percentiles at 1.2% and 5.8%, respectively (sample size varies slightly between years and is greater than 100 fields in each year).

Find suitable sources of organic matter. Spinach cropping most likely consumes SOM because of the intensive tillage regime (ploughing two to three times, repeated cultivator use). We here used the SOM balance after Leithold et al., (1997) to estimate SOM losses and inputs. The results suggest that biomass from harvest residues and cover crops sown after spinach would usually not compensate these losses. Organic matter should thus be introduced at a rate of approximately 1 t ha^{-1} of stabilised organic matter

Table 6.6 P loss via erosion and run-off at different erosion rates (t ha^{-1} yr^{-1}) and soil P levels (estimated as described in Chapter 4).

Erosion rate	P_{CAL} [a], Class [b]								
	2	4	7	10	13	19	26	33	40
	A	B		C		D		E	
0.5	1.0	1.0	1.0	1.5	1.5	1.5	2.0	2.0	2.5
1.0	1.5							4.0	4.5
1.5	2.5	2.5	2.5	3.0	3.5	4.0	4.5	5.0	6.0
2.0	3.0	3.0	3.5	4.0	4.0	5.0	5.5	6.5	7.5
3.0	4.0	4.0	4.5	5.0	5.5	6.5	7.5	9.0	10.0
4.0	5.0	5.5	6.0	6.5	7.0	8.5	9.5	11.0	12.5
5.0	5.6							13.0	15.0

[a] Plant available P (mg P_2O_5 per 100 g soil) after Schüller (1969).
[b] Soil P content classes as used by LUFA Münster (Lammers, 1999).

(e.g. compost or solid cattle manure). It could either be applied in the year of spinach cropping or elsewhere in the rotation.

As for recycled nutrients, typical sources of organic matter, such as manure or compost often carry contaminants. Finding adequate sources for organic matter could thus be difficult. A potentially interesting source is cereal straw, which could be applied in combination with a low rate of slurry to adjust the C:N ratio.

Another option to reduce SOM losses could be reducing tillage intensity. Past experiences suggest that ploughing is necessary to prepare an optimum seedbed for spinach and to rectify soil structural damage and compaction. There is, however, presently an experimental programme under way to explore the potentials for reduced tillage.

Select fertilisers for heavy low metal loads. See above.

Encourage cultural controls of erosion. Soil erosion depends mainly on climatic, soil and topographic factors, but management does also play a role: The protection of the soil by a plant canopy or mulch is represented by the C-factor of the RUSLE (Renard et al., 1997; Hennings, 2000). For spinach, the C-factor (calculated as described in Auerswald and Kainz, 1998 and Auerswald and Schwab, 1999) for a range of different situations is closely correlated with the soil cover index (days of the year with covered soil divide by 365). Figure 6.2 shows this relation.

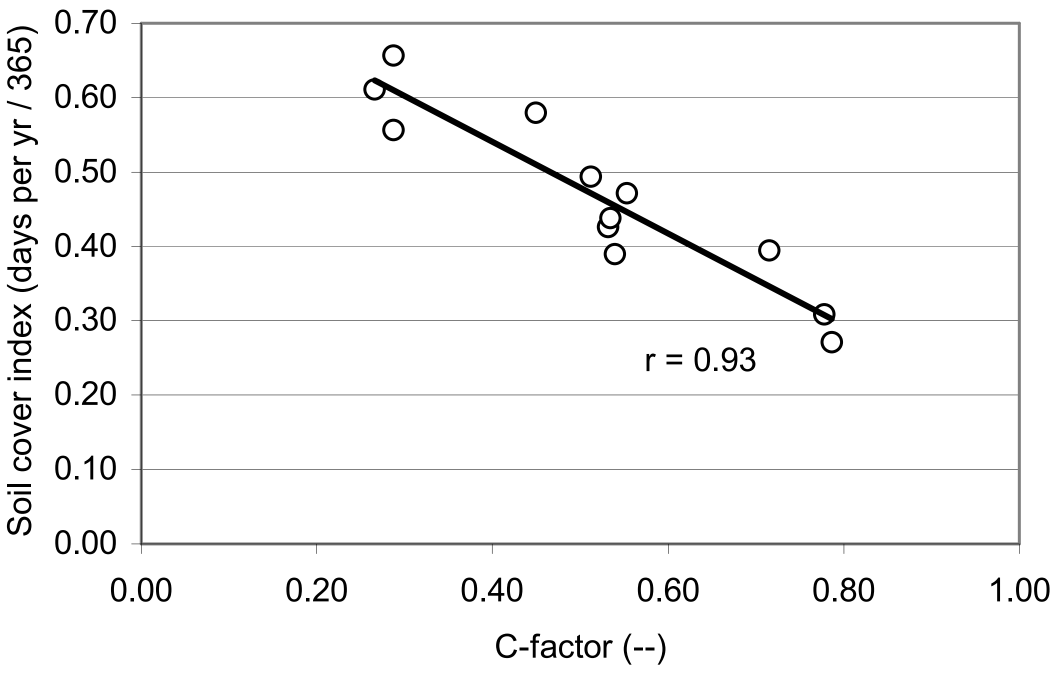


Figure 6.2 Relationship between soil cover index and the soil cover factor (C-factor) of the RUSLE in spinach grown in Borken (own data).

Another management measure with the potential to reduce erosion is to till fields across the slope. For mildly sloping fields tillage across the slope could reduce erosion by up to 50%. Long and rectangular fields are often situated parallel to the slope. In such cases it could be practicable to limit tillage across the slope to ploughing and drilling, while fertilisation and spraying could still be carried out along the slope.

A third measure to reduce erosions is to divide the erosive slope length, e.g. with a furrow or vegetated strip. Figure 6.3 shows critical slope lengths in relation to soil type and inclination (factor combinations, at which erosion rate exceeds 1.0 t ha^{-1} yr^{-1} if R = 95, C = 0.47 and P = 1.0). Finally, good soil structure and should be ensured in order to reduce susceptibility to erosion.

Consider erosion risk during field selection. Some fields have a predisposition to erosion due to soil type or topography. The risk of a particular combination of slope length,

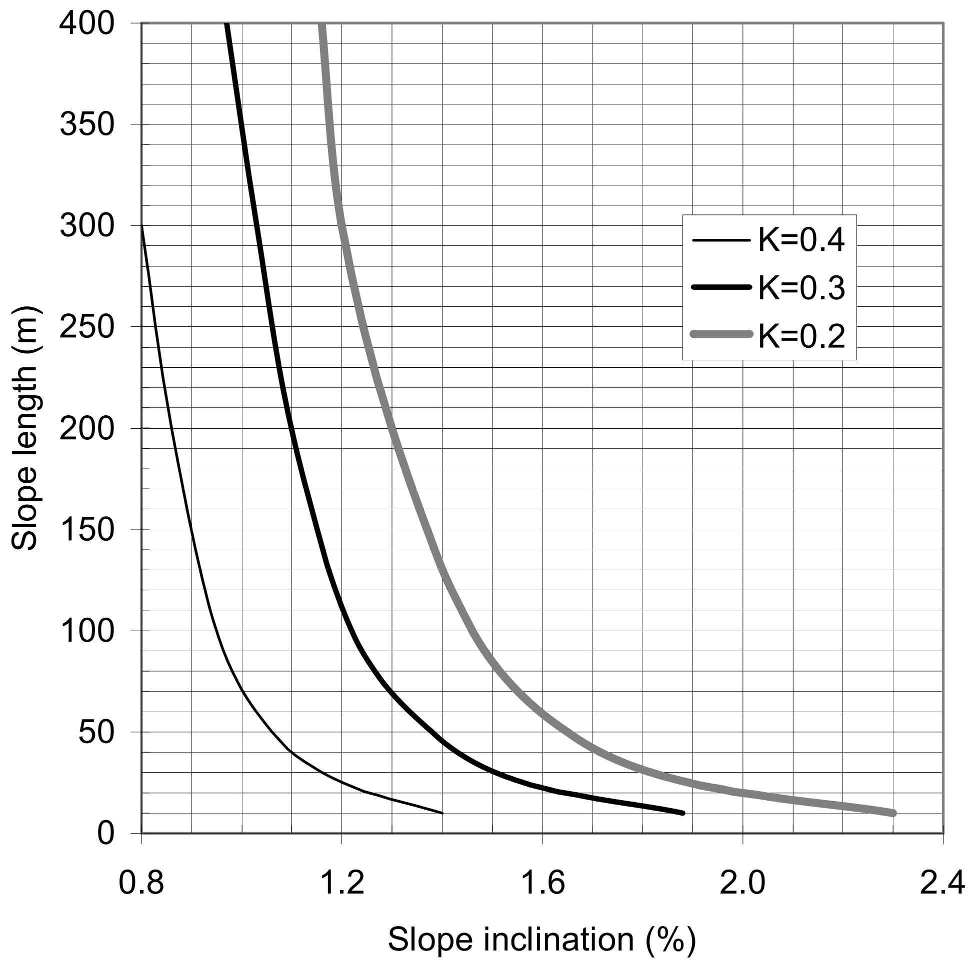


Figure 6.3 Iso-erodent lines for different soil types (K-factors of the RUSLE) as a function of erosive slope lengths (L-factor) and inclination (S-factor). Iso-erodent lines indicate factor combinations, at which the erosion rate exceed the soil formation rate of 1.0 t ha^{-1} yr^{-1} (given R = 95, C = 0.47 and P = 1.0).

inclination and soil type may be taken from Figure 6.4. Also, delayed drilling could reduce erosion: As Figure 6.4 shows, the C-factor of the RUSLE declines as drilling dates become later (this due to faster growth and canopy cover at later drilling dates).

Often, only parts of a field are susceptible to erosion. In that case, cultural controls as discussed above should be considered. If only a small area is affected it could be planted with non-crop vegetation. Finally, repeated occurrence of visible symptoms of erosion, such as erosion gullies on the field or sediments on adjacent areas should be taken as an indication to avoid spinach cropping on a particular field.

Encourage growers to use adequate tyres and inflation pressures. Our analysis showed, that narrow tyres with high inflation pressure used for cultivation, fertilisation and spraying cause nearly tree fourth of the soil compaction potential. Using wider tyres or twin tyres and load-adjusted inflation pressures could largely reduce soil compaction (see the above scenario TYRES, page 114). Load adjusted tyres inflation pressures can be obtained from producers' catalogues. As a rule of thumb, the inflation pressure should ideally be below 100 kPa, where for light tractors and under drier soil conditions (e.g. fertilisation) 150 kPa are tolerable. This is also in line with the official recommendation of the local agricultural extension service (Dr Norbert Uppenkamp, 2003, personal communication).

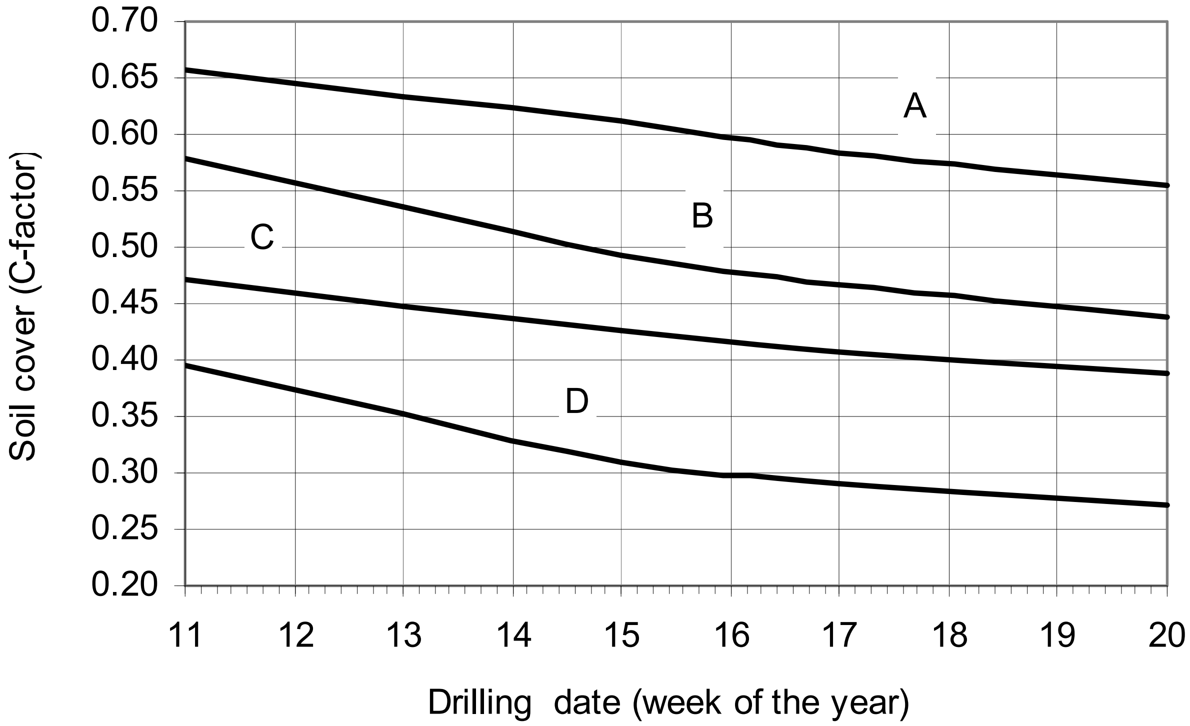


Figure 6.4 Relationship between drilling date of the first spinach crop, cover crop use and the C-factor of the RUSLE (calculated after Auerswald and Schwab, 1999): (A) without a cover crop, (B) cover crop before spinach, (C) cover crop after spinach (D) cover crop before and after spinach (given R = 95, K = 0.19, LS = 0.13, P = 1.0).

Grower self-assessment

We finally propose a voluntary grower self-assessment for contract growers as a part of a management system for sustainable spinach production. The self-assessment should ask growers to submit a small set of data. On these and information from the Iglo database a number of simple (proxy) indicators could be computed. Each participant would receive an individual report, stating indicator results and basic information on yield and soil testing results. The report should show results both in absolute figures as well as in relation to the other participants.

Voluntary grower benchmarking systems have proven to be a powerful tool to drive improvement in Unilever sustainability projects in Australia, Kenya and the US.

Chapter summary

The aim of this chapter was to link the environmental effects of spinach production in the County of Borken back to actual agricultural practices and to explore how modifying these practices would affect the environmental impact. Nutrient management and soil management were identified as the areas with the largest impacts on the environment.

We referred to two areas for improvement as the ‘nutrient complex’ and the ‘soil complex’. While issues in the nutrient complex can be reduced to a level at which they do not threaten sustainability, issues in the nutrient complex are more difficult to manage within the confines of intensive agricultural vegetable production.

Recommendations for the nutrient complex are to (1) select fertilisers for low heavy metal loads, (2) find recycled sources for K and other nutrients, (3) minimise gaseous N losses during application, (4) reduce N losses after harvest (5) and consider the risk of P losses during field selection. Additional recommendations for the soil complex are to (1) establish a reliable baseline and trend for SOM dynamics, (2) find suitable sources of organic matter, (3) encourage cultural controls of erosion, (4) consider erosion risk during field selection and (5) encourage growers to use adequate tyres and inflation pressures. In addition, a grower self-assessment is suggested, which allows growers to benchmark their own performance against that of the other growers.

CHAPTER 7

Environmental impacts in the context of sustainability

Negative impacts and the upside of it all

We applied the sustainability assessment method developed in this study to spinach production in the County of Borken in Northwest Germany. In Chapter 5, the production system's main environmental impacts were identified and evaluated. In Chapter 6 we analysed the sources of the impacts that could threaten sustainability and discussed mitigation potentials. We found that, while some of the impacts could be reduced to a level where they do not threaten sustainability, others are difficult to reduce substantially or come at the price of exacerbating other impacts.

These results do not, however, allow for statements on the overall sustainability of the whole production system: First of all, the method used here evaluates sustainability on the level of the individual *indicator*. A genuine weighting of different environmental issues against each other (Chapter 5) was not carried out and the results do therefore not account for differences in the social or political perception of the weight of different issues. Improvements in one area at the expense of another (trade-offs) may well be acceptable or the result of an informed choice form a societal or political point of view. Second, our method concentrates on environmental, and, to some degree, economic impacts. The social dimension is not accounted for. Finally, the method assesses negative impacts and does not quantify positive contributions.

In the context with sustainability, such positive contributions are often conceptualised as 'benefits' and the impacts as 'costs', i.e. they are positive and negative flows onto and from capital assets. 'Capital' is conceptualised as a compound of financial, natural, human and social capital (Pearce and Turner, 1993; Daly, 1996). In the terms of the benefit-cost construct, this study was concerned with quantifying the cost-side of land use in general and spinach production in the County of Borken in particular. In order to obtain a more complete picture of spinach production in Borken, we shall briefly discuss the benefit-side as well, namely the production of financial, human and social capital.

Benefits of spinach production in Borken

Production of financial capital

Financial capital refers to monetary assets, quite naturally expressed in monetary terms. The fact that spinach production in Borken has been commercially successful for the past 40 years can be taken as a strong indication for its capacity to create (positive) financial capital. This affects both Iglo and the roughly one hundred contract growers, for whom spinach cropping is an important source of income. In addition, there are indirect positive effects on financial capital: Iglo is a major employer in the region and sourcing goods and services locally generates a substantial flow of finance into the region (Dr Volker Schick, 2003; personal communication).

Production of human capital

Human capital refers to an individual's capacity to know and to act. It comprises knowledge, skills and awareness obtained trough formal and informal education (Wignarajah, 2001).

Compared to other arable crops, spinach is demanding, especially in terms of soil preparation, canopy establishment, weed control and irrigation management. Expertise in spinach cultivation in the Borken area has grown steadily ever since contract growing started in the early 1960s. This holds both in quantitative terms, as the grower-base has continuously been extended, and in qualitative terms, as the experience embodied within the grower community and the fieldsmen team increased substantially over the years.

Production of social capital

A common classification for social capital (Fukuyama, 1995; Putnam, 2000) distinguishes

- *Bonding social capital*, as the connectedness through trust between members of a social unit, which contributes to the social unit advancing its objectives; from
- *Bridging social capital*, as the connectedness through trust between different social units, being to their mutual benefit.

Bonding social capital refers to the degree of trust between the members of a social unit. The social unit in our case is the institution of spinach production, comprising of the Iglo factory, the contract growers and the contract harvester as core members.

Mutual trust among these core members has been build up over the past 40 years and many growers produce for Iglo in the second generation. As growers have organised themselves and are represented by the growers' board, there is an equitable distribution of market power between suppliers and buyer. For Iglo, the substitutionality of suppliers is low, because of the high degree of specialisation and skill required for spinach growing and the crop's limited transportation distance. For the growers, spinach cropping is an attractive possibility for income diversification. There is thus a mutual dependency between the growers and Iglo. The same is true for the contract harvester, who owns highly specialised custom equipment and technical and logistic experience, which Iglo depends on, but also gains a large part of his income from working for Iglo.

The mutual dependency between Iglo, growers and contract harvester evolved to an equitable partnership over the past decades. In interviews, representatives of the three actors were very content with the level of co-operation and partnership and saw no major need for improvement (Krusche, 2004; unpublished data).

Bridging social capital refers to the relations of the social unit with the outside world. The 'outside world' for Iglo's spinach operation spans various spatial scales: It comprises the local community and local interest groups as well as stakeholders on the national level and consumers. Although somewhat speculative, there is evidence that bridging social capital is sufficiently developed on the local level but less developed on the national level.

Iglo has established contacts and dialogues with a number of local stakeholders, including the local Association for the Enhancement of Nature (*Naturfördergesellschaft*), the municipalities and county administration, water suppliers and the Chamber of Agriculture. Krusche (2004, unpublished) interviewed a range of local community representatives for the importance of Iglo's spinach operation in the region and potential problems associated therewith. Most of the interviewees judged the company's presence as positive or at least as neutral. None of them specified any particular problems. We may take this as an indication that there are no pressing issues concerning bridging social capital on the local level.

Conversely, project growers expressed on various occasions concerns about the (perceived) bad reputation of agriculture in the public. In the broader sustainability discourse, these concerns may be subsumed under 'lacking contact between consumers and producers' or 'consumers' detachment from agricultural production' (cf. literature reviewed in Chapter 3). In fact, large-scale consumer studies undertaken by Iglo indicated that there were two extreme perceptions of agriculture (Peters, 2001): a positive one of romantic rural life and small-scale extensive farming and a negative one of industrialised agriculture, exploiting people and nature. Consumers tend to suspect the Iglo operation to belong to the second category. We may take this, in connection with the growers' concerns, as evidence for lacking bridging social capital on the national level.

Positive contributions to social and human capital made by the Iglo LAP

In the literature, conducting a practical sustainable agriculture project has variously been reported to have positive effects on human and social capital (Bell and Morse, 1999; Röling and Wagemakers, 1998; Lefroy et al., 2000; Bosshard, 1997; Van Mansvelt and Van der Lubbe, 1999). Typically, increasing awareness, skill and understanding among those involved are reported (human capital) as well as increasing co-operation and joint pursuit of (in part newly adopted) goals (bonding social capital) and strengthened relationships to external groups (bridging social capital). This does also hold for the Iglo LAP.

- It broadened the awareness of those involved for issues of sustainability, both in the ecological realm (understanding the production system's relations with its biophysical environment) and the socially realm (understanding the agendas of project partners and other actors and how they shape the system's social environment).
- Pilot growers and advisors developed new practical skills through field trials and new practices in a range of areas, such as erosion control, Qualitative Soil Structure Assessment (Beste, 2002), management of cover crops, grass and flower margins, irrigation management, mechanical weeding and novel pest control strategies. Two pilot growers also had farm specific biodiversity action plans developed.
- The Iglo LAP has provided a basis for learning through offering a number of training seminars and discussion groups. They were well received and appraised beneficial by the majority of participants. Themes included soil protection, pest management, biodiversity, renewable energy and nutrient management.
- Within the LAP, a clear commitment to partnership and participatory processes was made. Pilot growers and the growers' board were involved in all project decisions.
- A consumer communication initiative has been started that aims at communicating to consumers a more realistic picture of agriculture and spinach production. Among other activities, there are field and factory visits for consumers on three days per week

throughout the season (*Iglo-Land* project described in Chapter 7) and the origin of the raw material is now specified on the finished product.

- A number of information and decision support tools have been or are presently being developed to assist agricultural managers. This study describes one of these tools. Others are a GIS field database, an ISO Quality Management System for suppliers, a software module to calculate nutrient balances, a monitoring and warning system for infestations with the silver-y-moth (*Autographa gamma*), a farm-level habitat structure and quality assessment as well as a social systems analysis (see Chapter 7 for the individual sub-projects).

- Finally, spinach is a non-subsidised crop and the project is independent from any political or legislative scheme or government funding. A partnership between the food industry and its suppliers to engage in sustainable agriculture is, to our knowledge, unprecedented in Germany. The project is thus pioneering private business partnerships for sustainability within the agricultural sector. It thereby contributes to the establishment of institutional structures for implementing sustainable agriculture. According to Rutan (1994), lacking institutional structures are one of the main barriers to the implementation of sustainable agriculture.

In summary, there is evidence that Iglo's spinach operation enhances financial and human capital as well as bonding social capital. This is also true for bridging social capital on the local level. There is also evidence for lacking bridging social capital on the national level. This is, however, not a specific problem of spinach production but affects the relations between consumers and agriculture in general. The Iglo LAP has apparently reinforced the positive effects on human and social capital and has also enhanced bridging social capital.

Relating costs and benefits

Our analysis of mitigation potentials in Chapter 6 indicates that some of the environmental impacts of spinach cropping can be alleviated. However, the room for alleviating others – especially those related to the 'nutrient complex' (Chapter 6) – is limited. Although reducing some of these impacts is possible, they will most likely stay at a level that burdens the environment. This is not to say that we have to accept these environmental impacts, but we seem to encounter soon the confines immanent to the system of intensive agriculture.

This leads on to the discussion of the 'pros' and 'cons' of intensive agriculture in the broader context of societal preferences. This discussion goes beyond the scope of this study. We hypothesise, though, that if the negative impacts of intensive agriculture were conceptualised as costs of the production of certain benefits, we might find them quite acceptable: Agricultural intensivation is, among other features, characterised by decreasing labour demand (both per unit area and per unit product) and increased per hectare productivity. This development has been made possible by high yielding varieties, increasing mechanisation and high inputs of synthetic fertilisers, pesticides and irrigation (Miller and Wali, 1995; Matson et al., 1997; Tilman et al., 2002). Reversing this development would, at a given level of production, require increasing inputs of both labour and land. Both factors are expensive (land lease/cost make up 40% of the total producer's costs in spinach). Large-scale extensivation would hence result in increased product prices.

Conversely, consumer behaviour in Germany indicates a high societal preference for cheap food: The percentage expenditure for food in Germany is among the lowest in the EU (Eurostat, 2003) and the market share held by food discounters is increasing constantly (ZMP, 2003). Although few people would probably opt openly for increasing financial capital (low food expenditure) at the cost of natural capital (impacts of intensive agriculture) it seems to be widely agreed to accept this trade-off tacitly.

Summarising the results for spinach production in Borken, we may say that it produces human, social and financial capital and consumes natural capital. This resembles a situation, which is in the literature called 'weak sustainability' (Pearce and Turner, 1990; Pearce and Atkinson, 1993): Maintaining or increasing the total capital asset, but accepting that within the total asset one kind of capital – here natural capital – is drawn from. As opposed to that, supporters of 'strong sustainability' argue that natural capital cannot be substituted adequately and therefore demand another condition to be met: To attain sustainability, the total capital asset shall be maintained or increased *and* natural capital shall be maintained and increased.

This position is intuitive but it is also simplistic and clings to the economic concept of capital in a too narrow-minded way. A more adequate perspective for sustainability could be that of ecology, namely co-evolution (Noorgard, 1988; Iyer-Raniga and Treloar, 2000): All human activities, at least those taking place in the physical world, necessarily influence and alter the environment they take place in. The task is thus less one of avoiding change but rather of avoiding abrupt and radical change in order to allow ourselves and our fellow species to adapt. Likewise it is important to create viable adaptive systems that are not highly dependent on a particular set of boundary conditions (Bossel, 1999).

In this context, diminishing today's natural capital also bears the potential for creating tomorrow's natural capital. In fact, some of our nowadays most valued landscapes are the result of past human activities, which were highly destructive of the environment they took place in: Central European heath landscapes as well as Mediterranean macchia, to name just two, are the result of degradative overuse in past centuries.

This is not to argue for an *après-nous-la-deluge* attitude. Yet, to what extent and at what rate the reduction of natural capital is acceptable and for what gains it may be justified is a societal decision, not a scientific one. It is at the heart of this study to inform (and possibly fuel) the related discourse.

CHAPTER 8

Discussion of the method developed in this study

In this chapter we localise the method developed here within the ‘landscape’ of existing methods for assessing sustainable agriculture and then discuss it within this context, following the structure of the three stages of sustainability assessment identified in Chapter 2.

The ‘landscape’ of sustainability assessment schemes

Von Wirén-Lehr (2000) distinguishes between means-orientated and goals-orientated assessment schemes for sustainable agriculture and between relative and absolute ones. *Means-orientated* schemes evaluate agricultural production systems on the basis of adopting certain practices and production techniques (means). The criterion for evaluating sustainability in *goals-orientated* schemes is achieving certain environmental, economic or social targets.

Absolute assessment schemes provide set targets, against which a production system is evaluated and therefore allow for an ‘absolute’ judgement about the system’s sustainability. *Relative* schemes do not provide targets and therefore only allow for comparing different systems, options or scenarios against each other.

We used these two criteria to localise the method developed here within the ‘landscape’ of diverse assessment schemes introduced in Chapter 2 (Figure 8.1). In the discussion, there is a particular focus on two schemes that are presently discussed in Germany: KUL/USL (Eckert et al., 2000) and REPRO (Diepenbrock et al., 1999). KUL/USL uses a farm level indicator set, based on farm survey data (three year averages). REPRO is set up around a balancing and simulation model for farm matter and energy cycles. Farm and field data are used to simulate matter fluxes, from which then a set of field

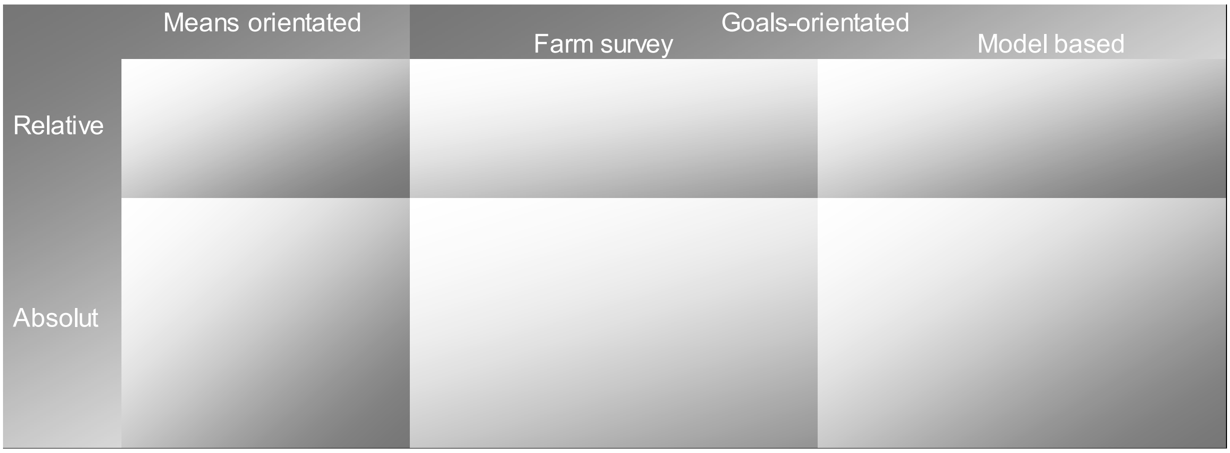


Figure 8.1 The method developed here within the 'landscape' of sustainability assessment schemes for land use systems.

and farm level indicators are computed. Both KUL/USL and REPRO use scoring functions to map indicator readings onto an interval of 1 to 10 (in USL/KUL) and 0 to1 (in REPRO). KUL/USL defines a 'tolerable range' of 1 to 6 within that interval; REPRO defines 0 as the worst and 1 as the best situation with regards to sustainability.

Bockstaller and co-workers developed a system similar to KUL/USL in France (Bockstaller et al., 1997; Bockstaller and Girardin, 2000b). They use complex compound indicators for each of the following thematic areas: (1) crop diversity; (2) rotation; (3) organic matter management; (4) phosphate; (5) nitrogen; (6) pesticide use; (7) irrigation; and (8) energy. Indicators are dimensionless indices. They are scaled to an interval from 0 to 10 in a way that 7 represents the recommended value for each indicator.

Sands and Podmore (2000) developed a system similar to REPRO for the Mid-West US. It is set up around a dynamic simulation model that is used to generate a set of indicators. They are scaled into an interval from 0 to 1. These scaled indicators are then aggregated to higher level indices.

Both KUL/USL and REPRO are – compared to the method developed here – 'mature' and have been developed by a number of people over several years. There are ongoing intensive debates about the two systems in the agronomic research community in Germany. KUL/USL has been adopted by the German Association of Agricultural Testing and Research Laboratories (VDLUFA) as the basis of a farm auditing and certification programme. The German Regionalised Agronomic and Environmental Information System (RAUMIS; Julius et al., 2003) also uses a set of modified KUL/USL indicators. REPRO has been implemented into a software tool that is sold as a part of a farm consultancy package.

Examples for relative assessment schemes in this discussion are Halberg (1999), Reganold et al. (2000) and Lefroy et al. (2001). Halberg (1999) presents a goals-orientated assessment scheme for Danish dairy farming. Reganold et al. (2001) present a goals-orientated scheme for apple production in the US (Washington State), which uses partly absolute, partly relative indicators. It is based on data from an experimental comparison of a conventional, an integrated and an organic apple production system. Lefroy et al. (2000) use the FESLM framework (Smyth and Dumanski, 1995) to develop a set of simple (mostly qualitative and semi-quantitative) indicators to assess the sustainability of smallholder farmers on sloping lands in Vietnam, Indonesia Thailand and Nepal. They use a mixed means and goals-orientated approach.

Taylor et al. (1993) take a purely means-orientated approach with absolute indicators by developing a ‘farmer sustainability index’ for Malaysian vegetable growers, which assigns different scores for the adoption of various (and supposedly more sustainable) practices. Rigby et al. (2001) follow the same approach for arable farming in England.

Comparison with other assessment schemes

Discussion of Stage 1: Indicator set construction

The method we developed in this study allows for systematically identifying indicanda that explicitly link with the sustainability discourse (Chapter 3). This ensures indicator set are genuinely relevant and reduces the risk of them being biased during construction, e.g. by the subjective views of academics; or by being the arbitrariness of societal negotiation. Using the NUSAP scheme by Funtowicz and Ravetz (1990) further allowed us to describe the theoretical, empirical and social quality of the indicandum-indicator relationship.

Including a systematic and transparent step of indicator set construction is, as far as we are aware, unique in the literature. Among the assessment schemes discussed here, only Lefroy et al. (2000) and KUL/USL provide details on indicator sets construction: Lefroy et al. used a participatory approach among local stakeholders and experts. The KUL/USL indicators were, according to its authors, derived through expert consultation and group sessions among German agricultural experts, largely from the academia. Halberg (1999) states that his selection of indicators was based on an analysis of the impact of Danish livestock keeping on relevant interests of future and present generations. Sands and Podmore (2000) give an axiomatic definition of ecological sustainability, on which they base the selection of indicators.

In fact, the system of Sands and Podmore is the only one among those reviewed that is explicit about the underlying sustainability conception and, to some degree, about the reasons and deliberations for indicator selection. However, even these authors explicitly exclude the impact of agriculture on air quality and climatic change from the scope of their system, without explaining this choice. This is difficult to understand considering the global role of agriculture as a major source of NH_3 and N_2O (IFA/FAO 2001). Halberg (1999) addresses heavy metal accumulation in arable soils in Denmark, which we also identified as a thread to sustainability in our case study in the County of Borken in Germany. According to UBA (2001a) the problem is widespread in German agriculture, but neither KUL/USL nor REPRO takes it into account. These examples show that indicator set construction is largely intransparent and often appears arbitray.

Another phenomenon is a widespread lack of consistency between sustainability perceptions and the choices of indicators. E.g. Reganold et al. (2000) and Eckert et al. (2000) make reference to a broader sustainability conception, but their choice of indicators shows only partial agreement with these basic deliberations: E.g. Eckert et al. (2000) list a number of concerns that motivated the development of KUL/USL, including global warming. The emission of greenhouse gases is, however, not an indicator in their system, nor reflected by other indicators. This is in line with a larger scale analysis by Anderson and Lockeretz (1992) who found that a lack of consistency between basic principles and the actual indicator selection was widespread in American sustainable agriculture protocols.

In other cases, indicators are ambiguous and could relate to several indicanda: The N and P budget surplus (e.g. KUL/USL; REPRO; Halberg, 1999) can be both a measure of environmental risk as well as of maintaining soil quality – both of which bear quite different implications.

In summary, there is a fundamental lack of transparency in the underlying assumptions and choices of indicators throughout published assessment schemes. It is thus difficult to compare or discus them. The method developed in this study tries to integrate systematic indicator selection into the assessment process and to make choices transparent.

Apart from transparency, this also makes our method highly flexible compared to the other methods discussed, which are bound to use pre-defined indicator sets. This means they are confined to production systems and contexts similar to those they were originally developed for and it limits their ability to cope with novel issues. Means-orientated approaches like those of Taylor et al. (1993) or Rigby et al. (2001) are even more specific than goals-orientated ones, because the set of practices that are regarded favourable will depend very much on the location and probably even the individual crop.

Thus, tailoring the indicator set to the specific requirements of a particular production system and context makes the method developed here highly flexible. It can integrate additional indicators, delete obsolete ones or replace an old indicator with a better one. The latter feature is important because scientific progress is made in many of the fields relevant for sustainable agriculture. E.g. in the case study presented here, we used NM VOCs as a proxy for ground level ozone formation. The Tropospheric Ozone Production Potential (EEA, 2000) is probably a better indicator.

Discussion of Stage 2: Indicator set evaluation

The assessment method we developed here is, according to the classification of von Wirén-Lehr (2000), an *absolute* assessment scheme. I.e. puts indicators into a normative context that allows a decision whether a particular measurement result is desirable or not, e.g. by comparing it against a threshold.

More generally, we called the step of putting indicators into a normative context *indicator evaluation*. In Chapter 5 we distinguished between an *indicator standardisation* step, which aims at making different indicators comparable, and a *valuation step*, which maps the standardised indicator values onto a sustainability function. While standardisation is largely descriptive, valuation is clearly normative and we separated them in order to ensure transparency and control of biases.

This is, again, unique in the literature. All other absolute assessment schemes discussed here use scaling functions that treat normalisation and valuation as one, typically by mapping the measurement results for diverse indicators onto a defined interval. As the nature of indicanda and the range of possible indicator measurements differ, the scaling functions also have to differ between indicators. In KUL/USL, these functions are set based on intensive expert consultation (Eckert et al., 2000). In REPRO, they are flexible and the authors propose they be negotiated at a political level among local representatives of governments and administrations, the farming community and other stakeholders. Lefroy et al. (2000) mostly use discrete qualitative indicators, threshold classes for which are chosen in a participatory process among local experts and stakeholders.

The drawback of such negotiated scaling functions is an immanent danger of conceptually flawed results, as is best illustrated by an example: Eckert et al. (2000) use the N budget as an indicator for N loading (eutrophication) of the environment. The threshold level for this indicator, however, does not correspond to the N tolerance of the environment but to (agronomically) 'inevitable minimum residues'. Agronomic deliberations are, however, completely irrelevant for the protection of sensitive ecosystems. The argument of agronomic feasibility implies certain yield expectations, i.e. a second indicandum – profitability – tacitly enters the discussion and is traded off against the apparent indicandum, N loading of the environment. I.e., the indicator N balance' relates to at least two indicanda, which should be addressed separately and have individual thresholds and reference values. Conflicts between the differing indicanda should be addressed and resolved at the level of indicator weighting and aggregation, but not at the level of standardisation.

As all other schemes discussed, the assessment scheme we developed here does not include a genuine indicator weighting step. It does, however, explicitly allow for integrating a weighting step. Conceptually, this is possible for other the absolute assessment schemes as well, although KUL/USL is the only one where advances have been made to identify weighting factors: According to its authors, this process has, however, been stopped because it had proven very difficult to agree weighting factors that find broad endorsement within the academia (Eckert, 2003, personal communication). This is not surprising. Finding genuine weighting factors is indeed ambitious because it is an entirely normative process (see Chapter 5) and even if consensus within the academia would have been reached, consensus among other social groups concerned with farming would not have been guaranteed.

Discussion of Stage 3: Strategy development

It is one of the obvious strengths of means-orientated assessment schemes that it is fairly easy to delineate practical advice. The assessment results in Taylor et al. (1993), Rigby et al. (2001) and, to some degree, Lefroy et al. (2000) are directly linked to adopting particular agricultural practices. Sustainability in these systems can thus be reached by simply adopting those practices that are 'more sustainable' (however defined). This is very useful for providing guidance, e.g. to farmers or agricultural policy makers. Simplicity comes, however, at a price: One has to pre-define whether or not a particular practice per se is sustainable or not; and regrettably few researchers and politicians will find themselves in a position to do this confidently. Indeed, the classification of 'unsustainable' and 'sustainable' practices in both Taylor et al. (1993) and Rigby et al. (2001) seems to be based on assertion more than on analysis of the effects of a particular practice.

Goals-orientated assessment schemes do not lend themselves as easily to delineating improvement strategies as means-orientated ones. Although often straightforward, the process of developing improvement strategies here is difficult to formalise. It involves an element of creativity and intuition when seeking to understand a particular outcome, identifying alternatives and finally drawing conclusions for improvement. The strengths of means-orientated assessment schemes lie predominantly in the area of policy advice. Goals-orientated ones, as the one developed here, are more useful in an academic and analytical context. In fact, they can be used to identify the generalised more sustainable practices that can then be implemented via a means-orientated scheme.

Strengths and weaknesses

Strengths

The above comparison with nine other assessment schemes highlighted two major advantages of the method developed here: It (1) explicitly addresses indicator set construction and (2) conceptually separates indicator normalisation and indicator valuation. Three further features of this method are unique: (1) its capacity to integrate over different sustainability dimensions; (2) its capacity to integrate over different spatial scales; and (3) its separation of normative and descriptive elements of the sustainability evaluation.

Capacity to integrate over different sustainability dimensions

In the past and present discourse on sustainable agriculture it is common to distinguish a number of different sustainability dimensions (Zink and Farshad, 1995). The distinction between an environmental (or ecological), a social and an economic dimension is most common. Yet, many indicator sets and assessment methods proposed in the literature stay within the supposedly safe confines of ‘hard’ natural science – i.e. the environmental dimension – and avoid the supposedly ‘soft’ economic and – even more so – social aspects (Anderson and Lockeretz, 1992; von Wirén-Lehr, 2001).

As opposed to many other assessment schemes, the method presented here is conceptually not confined to a particular sustainability dimension (cf. review in Chapter 2). Although our case study focuses on environmental impacts, the method can readily be extended to the social and economic dimension. Chapter 3 provides an inventory of social and economic issues as well, which can, by the same approach as for the environmental impacts, be used as the basis for identifying relevant issues.

As discussed before, the method developed here focuses on impacts, i.e. negative effects. The same conceptual approach could, however, be used for positive benefits as well, e.g., carbon sequestration (Robertson et al., 2000), enhancement of beneficial species (Thies and Tscharntke, 1999) or provision of food, income and jobs (Tilman et al., 2002). Instead of critical impact levels, one would have to define minimum benefit levels, such as economic break-even. It is, however, important to note that one must not set standardised indicators for ‘goods’ against those for the ‘bads’, because they each refer to conceptually distinct entities.

Similarly, the method is applicable not only to agriculture. By referencing all impacts to units area, also other land use systems than agriculture become comparable. This is important, because there are often different potential uses for the same piece of land. E.g. for wildlife conservation, nature conservationists could suggest agricultural extensivation (Matson et al., 1997). Alternatively, however, agriculture could be intensified in some part while leaving other parts to nature (Addiscott, 1995). At the same time, farmers could be more intrigued with selling their land as construction site for housing. By referencing impacts to units area, these different options can be compared.

This is, to our knowledge, unique to this method. Other methods either reference effects to units of the analysed product, such as the *functional unit* in Life Cycle Assessment (Brentrup, 2003). Or they convert all impacts to a single physical or monetary 'currency unit' (Addiscott, 1995; Wackernagel and Yount, 1998; Steinborn and Sviezhev, 2000; Tellarini and Caporali, 2000). Using a 'currency unit' is questionable, because converting as diverse issues as the loss of biodiversity or greenhouse gas emissions to monetary or energetic units is hardly adequate.

Capacity to integrate over different spatial scales

Sustainability involves multiple spatial and temporal scales. Interestingly, proposals for sustainability assessment acknowledging this fact are rare. Dumanski et al. (1998) and Smith and McDonald (1998) propose a multi-scale assessment for sustainable agriculture by assessing the same system at different spatial scales. This inflates complexity immensely. We here hold that environmental as well as social and economic problem usually emerge at distinct scales (cf. O'Neill et al., 1989). Even a multi-scale assessment is thus likely to encounter a particular problem only once, on its specific scale. In fact, if a problem is encountered repeatedly at different scales it is unlikely to be the same problem: E.g. riverine and marine eutrophication affect different organisms and ecosystems, involve different nutrients, pathways and different thresholds. It is thus useful to treat them as distinct problems.

Based on this observation we suggest rather breaking down individual issues to a single common reference scale, than scaling impacts of the system up: The latter would mean multiplying the assessment effort by the number of relevant spatial scales. Conversely, the method developed here references all impacts to a per-hectare-and-year basis, which allows for simultaneously assessing issues that originally emerge at different scales.

Conceptual separation of normative and descriptive elements

Stevenson and Lee (2001) plea for clearly separating the 'objective' (= scientific and value-free) and 'subjective' (= political and value-laden) steps of a sustainability assessment process. We here used the concepts of 'descriptive' and 'normative' instead of objective and subjective, but we share the idea of keeping steps of different normative contents separate in order to enhance transparency and credibility. The method developed here reflects this by

- acknowledging the normative nature of sustainable agriculture as a policy goal by identifying relevant issues (indicanda) based on the ongoing discourse, as represented in the literature and within the project (Chapter 3);
- explicitly describing the theoretical, empirical and social quality of data and indicator-indicandum relations (Chapters 2 and 3);
- separating indicator normalisation, severity weighting and the actual sustainability valuation, i.e. separating evaluation steps with differing normative contents (Chapter 5);
- using science-based critical impact levels in indicator standardisation, which ground on the disciplines concerned with a particular issue in order to prevent non-pertinent arguments biasing the results (Chapters 3 and 5).

It is, however, important to note that the separation between normative and descriptive elements is conceptual. In practice, the descriptive elements require choices and assumptions, which imply particular values and norms. Normativity inevitably enters the descriptive processes at various parts, as was discussed in Chapter 3.

Weaknesses

For the time being, the method developed in this study demands high temporal effort. Although the procedure itself is relatively simple and could, in the case study, largely be performed on public domain data, finding and preparing the data was time consuming. The same is true for ensuring methodological consistency and comparability between different data sources. This shortcoming could be overcome by collating reference databases. These could contain both basic data on various potential sustainability issues as well as 'off-the-shelf' normalisation factors and severity factors for particular regions. Implementation in a computer programme would further improve the potential for easy application of the method.

Coupling this method with standard simulation models (such as crop, soil, hydrological and air pollution models) would further enhance its applicability and power. As different models are preferred by different users, we suggest creating protocols for data import from various standard models.

Finally, this method lends itself to use in academia, research and policy support. As with most goals-orientated sustainability assessment schemes, agricultural practitioners may find it less useful, though, because its output does explicitly advise on how to be more sustainable. There is no immediate guidance on better practices and identifying the most effective options for more sustainable production may not be straightforward.

Conclusions

We found that the method presented here overcomes some methodological shortcoming of previously published assessment schemes for sustainable agriculture. Namely, the method's scope explicitly includes indicator set construction; it can be applied to any land use system and location; hence, it allows for comparing diverse land use systems, regardless of type, location and scale level; it allows for using locally adjusted indicator sets and can integrate over various sustainability dimensions and scales; changes in indicators set or indicator calculation method are easy to implement; it clearly separates descriptive and normative elements and thereby allows for managing normativity within the scientific process. This results in three main benefits

- Comprehensiveness
- Flexibility
- Transparency.

Establishing reference databases and implementation in a software tool could enhance the method's practicability.

References

Abelson, P.H., 1995. Sustainable agriculture and the 1995 farm bill. Nature 267, 943.

Addiscott, T.M., 1995. Entropy and sustainability. European Journal of Soil Science 46, 161-168.

Addiscott, T.M. and Thomas, D., 2000. Tillage, mineralization and leaching: phosphorus. Soil and Tillage Research 53, 255-273.

Allen, P., Van Dusen, D., Lundy, J. and Gliessman, S., 1991. Expanding the definition of sustainable agriculture. Journal of Alternative Agriculture 6, 34-39.

Amann, M. and Lutz, M., 2000. The revision of air quality legislation in the European Union related to ground-level ozone. Journal of Hazardous Materials 78, 41-62.

Amann, M., Imrich, B., Cofala, J., Gyarfas, F., Klimont, Z. and Schöpp, W., 1999. Integrated assessment modelling for the protocol to abate acidification, eutrophication and ground-level ozone in Europe. Report to the Dutch Ministry of Housing, Spatial Planning and Environment. International Institute for Applied Systems Analysis (IIASA), Laxenburg.

Anderson, M.D. and Lockeretz, W., 1992. Sustainable agriculture research in the ideal and in the field. Journal of Soil and Water Conservation 47, 100-104.

Andreoli, M. and Tellarini, V., 2000. Farm sustainability evaluation: methodology and practice. Agriculture, Ecosystems and Environment 77, 43-52.

Anonymous, 2000. Erfahrungssätze für Maschinenring-Arbeiten unter Landwirten (in German). Kuratorium der Betriebshilfsdienste und Maschinenringe in Westfalen-Lippe e.V. and Landwirtschaftskammer Westfalen-Lippe.

ASA, 1989. Decision reached on sustainable agriculture. American Society of Agronomy (ASA). Agronomy News, January 1989, 15.

Asman, W.A.H., 1992. Ammonia emission in Europe: updated emission and emission variations. National Institute for Public Health and the Environment (RIVM), Bilthoven.

Auerswald, K., 1987: Sensitivität erosionsbestimmender Faktoren (in German). Wasser und Boden 39, 34-38.

Auerswald, K., 1989. Predicting nutrient enrichment from long-term average soil loss. Soil Technology 2, 271-277.

Auerswald, K. and Kainz, M., 1998. Erosionsgefährdung (C-Faktor) durch Sonderkulturen (in German). Bodenschutz 3, 98-102.

Auerswald, K. and Schwab, A., 1999. Erosionsgefährdung (C-Faktor) unterschiedlich bewirtschafteter Weinanbauflächen (in German, with English abstract). Viticultural and Enological Science 54, 54-60.

Auerswald, K. and Weigand, S., 1999. Eintrag und Freisetzung von P durch Erosionsmaterial in Oberflächengewässern (in German). In: VDLUFA-Schriftenreihe 50. Darmstadt, pp. 37-54.

Bach, M., Huber, A., Frede, H.-G., Mohaupt, V. and Zullei-Seibert, N., 2000. Schätzung der Einträge von Pflanzenschutzmitteln aus der Landwirtschaft in die Oberflächengewässer Deutschlands (in German, with English summary). Erich Schmidt, Berlin.

Bandick, A.K. and Dick, R.P., 1999. Field management effects on soil enzyme activities. Soil Biology and Biochemistry 31, 1471-1479.

Barkmann, J. and Windhorst, W., 2000: Hedging our bets: The utility of ecological integrity. In: Joergensen, S.E. and Müller, F. (eds.): Handbook of ecosystem theories and management. Lewis Publishers, Boca Raton, FL. pp. 497-517.

BBA, 2002. Verzeichnis der regionalisierten Kleinstrukturanteile (in German). Website of the Biologische Bundesanstalt für Land- und Forstwirtschaft (BBA) (http://www.bba.de/), Braunschweig, Germany.

BBodSchV, 1999. Bundes-Bodenschutz- und Altlastenverordnung vom 12.7.1999. Bundesgesetzblatt 1999, I 1554 ff.

Becker, B., 1997. Sustainability Assessment: A Review of Values, Concepts, and Methodological Approaches. Issues in Agriculture 10. Consultative Group on International Agricultural Research (CGIAR). The World Bank, Washington, DC.

Becker, J., 1998. Sustainable development for local land uses. International Journal of Sustainable Development and World Ecology 5, 59-69.

Bell, S. and Morse. S., 1999. Sustainability Indicators: Measuring the Immeasurable Earthscan, London.

Bergmann, W. (ed.), 1992. Nutritional Disorders of Plants. Gustav Fischer, Jena.

Beste, A., 2002. Weiterentwicklung und Erprobung der Spatendiagnose als Feldmethode zur Bestimmung ökologisch wichtiger Gefügeeigenschaften landwirtschaftlich genutzter Böden (in German, with English summary). Dissertation, University of Gießen, Department of Agriculture. Gießen.

Bindraban, P.S., Stoorvogel, J.J., Jansen, D.M., Vlamming, J. and Groot, J.J.R., 2000. Land quality indicators for sustainable land management: Proposed method for yield gap and soil nutrient balance. Agriculture, Ecosystems and Environment 81, 103-112.

Binswanger, H.P., 2001. HIV/AIDS Treatment for Millions. Science 292, 221-223.

Bockstaller, C. and Girardin, P., 2000a. Agro-ecological indicators – instruments to assess sustainability in agriculture. In: M. Härdtlein, M. Kaltschmitt, I. Lewandowski and H. Wurl (eds.), Nachhaltigkeit in der Landwirtschaft. Erich Schmidt, Berlin, pp. 69-83.

Bockstaller, C. and Girardin, P., 2000b. Berechnungsverfahren "Agrarökologische Indikatoren" (in German, French version available). Laboratoire D'Agronomie INRA, Colmar.

Bockstaller, C., Girardin, P. and van der Werf, H.M.G., 1997. Use of agro-ecological indicators for the evaluation of farming systems. European Journal of Agronomy 7, 261-270.

Bossel, H., 1999. Indicators for Sustainable Development: Theory, Method, Applications – A Report to the Balaton Group. International Institute for Sustainable Development (IISD), Winnipeg.

Bosshard, A., 1997. What does objectivity mean for analysis, valuation, and implementation in agricultural landscape planning? A practical and epistemological approach to the research for sustainability in 'agri-culture'. Agriculture, Ecosystems and Environment 63, 133-143.

Bouwer, W., Gäth, S. and Frede, H.-G., 1997. Vergleich dreier Instrumente zur Abschätzung und Kontrolle der nutzungsbedingten Nitratauswaschung auf auswaschungsgefährdeten Standorten. (in German, with English summary). Zeitschrift für Kulturtechnik und Landentwicklung 38, 154-160.

Bouwman, A.F., 1995. Compilation of a Global Inventory of Emissions of Nitrous Oxide. Dissertation, University of Wageningen, Wageningen.

Bouwman, A.F., Boumans, L.J.M. and Batjes, N.H., 2002. Modeling global annual N_2O and NO emission from fertilized fields. Global Biogeochemical Cycles 16, (8-1)-(8-15).

Bouwman, A.F., Boumans, L.J.M. and Batjes, N.H., 2002a. Estimation of global NH_3 volatilization loss from synthetic fertilizers and animal manure applied to arable lands and grasslands. Global Biogeochemical Cycles 16, (8-1)-(8-15).

Bouwman, A.F., Boumans, L.J.M. and Batjes, N.H., 2002b. Modeling global annual N_2O and NO emission from fertilized fields. Global Biogeochemical Cycles 16, (8-1)-(8-15).

Bouwman, A.F., Boumans, L.J.M. and Batjes, N.H., 2002c. Emissions of N_2O and NO from fertilised fields: Summary of available measurement data. Global Biogeochemical Cycles 16, (6-1)-(6-13).

Bouwman, A.F., Lee, D.S., Asman, W.A.H., Dentener, F.J., Van Der Hoek, K.W. and Olivier, J.G.J., 1997. A global high-resolution emission inventory for ammonia. Global Biogeochemical Cycles 11, 561-587.

Brentrup, F., 2003. Life Cycle Assessment to Evaluate the Environmental Impact of Arable Crop Production. Dissertation, University of Hannover, Department of Horticulture. Cuvillier Verlag, Göttingen.

Brentrup, F., Küsters, J., Lammel, J. and Kuhlmann, H., 2000. Methods to estimate on-field nitrogen emissions from crop production as an input to LCA studies in the agricultural sector. International journal of Life Cycle Assessment 5, 349-357.

Brentrup, F., Küsters, J., Lammel, J. and Kuhlmann, H., 2002. Life Cycle Impact Assessment of land use based on the hemeroby concept. International journal of Life Cycle Assessment 7, 339-348.

Brentrup, F., Küsters, J., Lammel, J. and Kuhlmann, H., 2004. Environmental impact assessment of agricultural production systems using the life cycle assessment methodology: Part I: Theoretical concept of a LCA method tailored to crop production. European Journal of Agronomy (in press).

Buckerfield, J.C., Lee, K.E., Davoren, C.W. and Hannay, J.N., 1997. Earthworms as indicators of sustainable production in dryland cropping in southern Australia. Soil Biology and Biochemistry 29, 547-554.

Burth, U., Gutsche, V., Freier, B. and Roßberg, D., 2002. Das notwendige Maß bei der Anwendung chemischer Pflanzenschutzmittel (with English abstract). Nachrichtenblatt des Deutschen Pflanzenschutzdienstes 54, 297-303.

CCE, 1991. Mapping of Critical Loads for Europe – CCE Technical Report No. 1. Coordination Center for Effects (CCE) National Institute for Public Health and the Environment, Bilthoven.

Christen, O., 1996. Nachhaltige Landwirtschaft (‘Sustainable agriculture’) (in German, with English summary). Berichte über Landwirtschaft 74, 66-86.

Christen, O., 1998. No more definitions please! Some reflections about the ideas and principles of sustainable agriculture. In: N. El Bassam, R. K. Behl and B. Prochnow (eds.), Sustainable Agriculture for Food, Energy and Industry. James & James, London, pp. 48-52.

Christen, O., 1999. Sustainable Agriculture – from the history of ideas to practical application. Institut für Landwirtschaft und Umwelt (ilu), Bonn.

Christen, O., 2000. Nachhaltige Landwirtschaft – Neue Leitbild oder altbekannte Prinzipien? (in German). In: VDLUFA-Schriftenreihe 55 (Kongreßband 2000). Darmstadt, pp. 14-23.

Conrad, J., 1999. Nachhaltige Entwicklung: einige begriffliche Präzisierungen oder der heroische Versuch, einen Pudding an die Wand zu nageln (in German, with English summary). FFU Report 00-07, Forschungsstelle für Umweltpolitik, Freie Universität Berlin.

CONSSO, 2002. Fifth International Conference on the Protection of the North Sea. Progress Report. Committee of North Sea Senior Officials (CONSSO), Bergen.

Costanza, R. and Patten, B.C., 1995. Defining and predicting sustainability. Ecological Economics 15, 193-196.

Costanza, R., 1993. Developing ecological research that is relevant for achieving sustainability. Ecological Applications 3, 579-581.

Costanza, R., Funtowicz, S.O. and Ravetz, J.R., 1992. Assessing and communicating data quality in policy-relevant research. Environmental Management 16, 121-131.

Cornforth, I.S., 1999. Selecting indicators for assessing sustainable land management. Journal of Environmental Management 56, 173-179.

Daly, H.E., 1996. Beyond Growth. The Economics of Sustainable Development. Beacon Press, Boston.

Davidson, E.A. and Kingerlee, W., 1997. A global inventory of nitric oxide emissions from soils. Nutrient Cycling in Agroecosystems 48, 37-50.

DBV, 2000. Grünbuch für eine Nachhaltige Land- und Forstwirtschaft (in German). Deutscher Bauernverband (DBV). agrar-press, Bonn.

De Kruijf, H.A.M. and Van Vuuren, D.P., 1998. Following sustainable development in relation to the North-South dialogue: ecosystem health and sustainability indicators. Ecotoxicology and Environmental Safety 40, 4-14.

Delmas, R., Serças, D. and Jambert, C., 1997. Global inventory of NO_x sources. Nutrient Cycling in Agroecosystems 48, 51-60.

Delschen, T. and Leisner-Saaber, J., 1998. Selbstversorgung mit Gemüse aus schwermetallbelasteten Gärten: Eine Gefährdungsabschätzung auf toxikologischer Basis (in German, with English abtract). Bodenschutz 1/98, 17-20.

Diepenbrock, W., Rost, D., Hülsbergen, K.-J., Abraham, J., Deimer, C., Dubsky, G., Heine, M., Heinrich, J., Meyer, D., Pasler, U. and Werner, S., 1999. Reports on the research projects "Entwicklung einer Methode zur ökologisch-ökonomischen Analyse und Bewertung landwirtschaftlicher Bewirtschaftungssysteme" and "Überführung und Weiterentwicklung der im Modell REPRO enthaltenen wissenschaftlichen Methoden und Koeffizienten zur landwirtschaftlichen Stoff- und Energiebilanzierung in eine praxisanwendbare Software" (in German). Martin-Luther University Halle-Wittenberg, Institut für Acker- und Pflanzenbau/Institut für Agrarökonomie und Agrarraumgestaltung, Halle.

Döhler, H., Eurich-Menden, B., Dämmgen, U., Osterburg, B., Lüttich, M., Bergschmidt, A., Berg, W. and Brunsch, R., 2002. BMVEL/UBA-Ammoniak-Emissionsinventar der deutschen Landwirtschaft und Minderungsszenarien bis zum Jahre 2010 (in German, with English summary). Umweltbundesamt (UBA), Berlin.

Douglass, G.K., 1984. The meanings of sustainable agriculture. In: G. K. Douglass (ed.), Agricultural Sustainability in a Changing World Order. Westview Press, Boulder, CO, pp. 1-29.

Dovers, S.R. and Handmer, J.W., 1995. Ignorance, the precautionary principle, and sustainability. Ambio 24, 92-97.

Dresselhaus, M.S. and Thomas, I.L., 2001. Alternative energy technologies. Nature 414, 332-337.

Drew, M.C., 1992. Soil aeration and plant root metabolism. Soil Science 154, 259-269.

Dumanski, J., Pettapiece, W.W. and McGregor, R.J., 1998. Relevance of scale dependent approaches for intergarting biophysical and socio-economic information of agroecological indicators. Nutrient Cycling in Agroecosystems 50, 13-22.

Dunlap, R.E., Beus, C.E., Howell, R.E. and Waud, J., 1992. What is sustainable agriculture? An empirical examination of faculty and farmer definitions. Journal of Sustainable Agriculture 3, 5-39.

DWD, 2001. Wetterdaten 1961 – 1991. Deutscher Wetterdienst (DWD), Darmstadt.

EC, 1999. Directions towards sustainable agriculture. Commission of the European Communities. COM(1999) 22 final. Brussels.

ECETOC, 1994. Ammonia Emissions to Air in Western Europe. European Centre for Ecotoxicology and Toxicology of Chemicals (ECETOC), Brussels.

Eckert, H., G. Breitschuh, and Sauerbeck, D.R., 2000. Criteria and standards for sustainable agriculture. Journal of Plant Nutrition and Soil Science 163, 337-351.

EEA, 2000. Environmental signals 2000 – regular indicator report. European Environment Agency (EEA), Copenhagen.

EEA, 2001. Joint EMEP/CORINAIR Atmospheric Emission Inventory Guidebook. European Environment Agency, Copenhagen.

EIA, 2001. International Energy Annual, Release 2001. Website of the U.S. Energy Information Administration (EIA) (http://www.eia.doe.gov/emeu/international/).

EMEP, 1999. Transboundary Acid Deposition in Europe. EMEP Summary Report 1999. EMEP/MSC-W, Oslo, Norway.

EMEP, 2002a. Emission data reported to UNECE/EMEP. MSC-W Status Report 2002. EMEP/MSC-W, Blindern.

EMEP, 2002 b. Online database (WebDaB 2002). http://webdab.emep.int. 2nd release. EMEP/MSC-W, Blindern.

Enquete-Kommission, 1994. Schutz der Grünen Erde – Klimaschutz durch umweltgerechte Landwirtschaft und Erhalt der Wälder (in German). Enquete Kommission Schutz der Erdatmosphäre des Bundestages, Ecomomica Verlag, Bonn.

Eswaran, H., Pushparajah, E., and Ofori, C., 1994. Indicators and their utilization in a Framework for Evaluation of Sustainable Land Management. In: Wood, R. C. and Dumanski, J. (eds.) Proceedings of the International Workshop on Sustainable Land Management in the 21st Century, Vol. 2: Plenary Papers. The Organizing Committee. International Workshop on Sustainable Land Management. Agricultural Institute of Canada, Ottatwa, ON. pp. 205-225.

EU, 1998. Council Directive 98/83/EC on the quality of water intended for human consumption. European Commission, Brussels.

EU, 2003. EU Maximum Residue Levels for pesticides as listed in the annexes to Council Directives 76/ 895/ EEC, 86/ 362/ EEC, 86/ 363/ EEC & 90/ 642/ EEC. (http://europa.eu.int/comm/food/fs/ph_ps/index_en.htm). European Union, Brussels.

Eurostat, 2003. Eurostat website (http://europa.eu.int/comm/eurostat/). Eurostat – Statistical Office of the European Communities, Luxembourg.

FAO, 1998. World reference base for soil resources. World Soil Resources Reports 84. Food and Agriculture Organization of the United Nations (FAO), Rome.

FAO, 2001. FAOSTAT: FAO Statistical Databases. (available at http://www.fao.org/waicent/portal/statistics_en.asp). Food and Agriculture Organisation of the United Nations (FAO), Rome.

FAO, 2003. FAO HIV/AIDS Programme. Food and Agriculture Organization of the United Nations (FAO), Rome.

FAO/WHO, 1975. Report of the 1974 Joint Meeting of the FAO Working Party of Experts on Pesticide Residues and the WHO Expert Committee on Pesticide Residues. Food and Agriculture Organization of the United Nations (FAO), Rome.

Farshad, A. and Zinck, J.A., 1993. Seeking agricultural sustainability. Agriculture, Ecosystems and Environment 47, 1-12.

Ferm, M., 1998. Atmospheric ammonia and ammonium transport in Europe and critical loads – A review. Nutrient Cycling in Agroecosystems 51, 5-17.

Fields, C.B., 2001. Sharing the Garden. Science 294, 2490-2491.

Finnish Environment Institute, 2002. Evaluation of the implementation of the 1988 Ministerial Declaration regarding nutrient load reductions in the Baltic Sea catchment area. Helsinki.

Fixdal, J., 1997. Consensus conferences as 'extended peer groups'. Science and Public Policy 24, 366-376.

Freibauer, A. and Kaltschmitt, M., 2002. Controls and models for estimating direct nitrous oxide emissions from temperate and sub-boreal agricultural mineral soils in Europe. Biogeochemistry (in press).

Freibauer, A., 2002. Regionalised inventory of biogenic greenhouse gas emissions from European agriculture. European Journal of Agronomy 19, 135-160.

Friend, A.M., 1996. Sustainable development indicators: exploring the objective function. Chemosphere 33, 1865-1887.

Fukuyama, F., 1995. Trust: the social virtues and the creation of prosperity. The Free Press, New York.

Funtowicz, S.O., and Ravetz, J.R., 1990. Uncertainty and quality in science for policy. Kluwer Academic Press, Dordrecht.

Funtowicz, S.O. and Ravetz, J.R., 1993. Science for the post-normal age. Futures 25, 739-755.

Funtowicz, S.O., and Ravetz, J.R., 1994. The worth of a songbird: ecological economics as a post-normal science. Ecological Economics 10, 197-207.

Gaillard, G., Crettaz, P. and Huasheer, J., 1998. Umweltinventar der landwirtschaftlichen Inputs im Pflanzenbau (in German). FAT Schriftenreihe Nr. 56. Eidgenössische Forschungsanstalt für Agrarwissenschaft und Landtechnik (FAT), Tänikon.

Gäth, S. and Wohlrab, B., 1994. Strategiekonzept der Deutschen Bodenkundlichen Gesellschaft zur Reduzierung standort- und nutzungsbedingter Belastungen des Grundwassers mit Nitrat (in German). Wirtschafts- und Verlagsgesellschaft Gas und Wasser, Bonn.

Geier, U. and Köpke, U., 2000. Analyse und Optimierung des betrieblichen Umweltbewertungsverfahrens "Kriterien umweltverträglicher Landbewirtschaftung" (KUL) (in German). Berichte über Landwirtschaft 3/2000, 70-91.

Gell-Mann, M., 1994. The quark and the jaguar – Adventures in the simple and the complex. Freeman, New York.

GenStat Committee, 2003. GenStat® Release 7.1: Reference Manual. VSN International, Oxford.

Gierich, J., 1997. Gegenüberstellung der Bewertungskonzepte für Agrar-Umweltindicatoren und bei Ökobilanzen. (in German). In: W. Diepenbrock (ed.). Umweltverträgliche Pflanzenproduktion: Indikatoren, Bilanzierungsansätze und ihre Einbindung in Ökobilanzen. Deutsche Bundesstiftung Umwelt, Osnabrück. pp. 291-312.

Gilbert, A., 1996. Criteria for sustainability in the development of indicators for sustainable development. Chemosphere 33, 1739-1748.

Gutsche, V. and Enzian, S., 2002. Quantifizierung der Ausstattung einer Landschaft mit naturbetonten terrestrischen Biotopen auf Basis digitaler topographischer Daten (German, with English abstract). Nachrichtenblatt des Deutschen Pflanzenschutzdienstes 54, 92-101.

Haag, D. and Kaupenjohann, M., 2000. Landscape fate of nitrate fluxes and emissions in Central Europe – A critical review of concepts, data, and models for transport and retention. Agriculture, Ecosystems and Environment 86, 1-21.

Hagedorn, K., 1997. Agriculture in Germany: Some considerations on economic, political and environmental sustainability. Ambio 26, 456-461.

Halberg, N., 1999. Indicators of resource use and environmental impact for use in a decision aid for Danish livestock farmers. Agriculture, Ecosystems and Environment 76, 17-30.

Hansen, J.W. 1996. Is agricultural sustainability a useful concept? Agricultural Systems 5, 117-143.

Härdtlein, M. and Vandré, R., 2000. Allocation of Regional Environmental Effects to the Local Production and Utilisation of Energy Crops. In: E. van Ierland, A. O. Lansink and E. Schieman (eds.), Sustainable Energy: New Challenges for Agriculture and Implications for Land Use. Wageningen.

Harger, J.R.E., and Meyer, F.-M., 1996. Definition of indicators for environmentally sustainable development. Chemosphere 33, 1749-1775.

Haynes, R.J. and Tregurtha, R., 1999. Effects of increasing periods under intensive arable vegetable production on biological, chemical and physical indices of soil quality. Biology and Fertility of Soils 28, 259-266.

HELCOM, 2003. Executive Summary of the Fourth Baltic Sea Pollution Load Compilation. Helsinki Commission (HELCOM)/Baltic Marine Environment Protection Commission, Helsinki, Finland.

Helling, C.S., Chesters, G. and Corey, R.B., 1964. Contribution of organic matter and clay to soil cation-exchange capacity as affected by the pH of the saturating solution. Soil Science Society of America Proceedings 28, 517-520.

Hennings, V. (co-ordinator), 1994. Methodendokumentation Bodenkunde: Auswertungsmethoden zur Beurteilung der Empfindlichkeit und Belastbarkeit von Böden (in German). E. Schweizerbart'sche Verlagsbuchhandlung, Hannover.

Hobhouse, H., 1985. Seeds of change: five plants that transformed mankind. Sidgwick & Jackson, London.

Horlacher, D. and Marschner, H., 1990. Schätzrahmen zur Beurteilung von Ammoniakverlusten nach Ausbringung von Rinderflüssigmist (in German, with English abstract). Journal of Plant Nutrition and Soil Science 153, 107-115.

Horn, R., Paul, R., Simota, C. and Fleige, H., 1996. Schutz vor mechanischer Belastung (in German). In: H.-P. Blume (ed), Handbuch der Bodenkunde, Kap 7.4 (14. Erg. Lfg. 12/2002). ecomed, Landsberg (Lech), pp. 1-12.

Hoyningen-Huene, P., 1999. The nature of science. Nature and Resources 35, 4-8.

Huijbregts, M.A.J. and Seppälä, J., 2001. Life cycle impact assessment of pollutants causing aquatic eutrophication. International Journal of Life Cycle Assessment 6, 339-343.

Huijbregts, M.A.J., Schöpp, W., Verkuijlen, R., Heijungs, R. and Reijnders, L., 2000. Spatially explicit characterisation of acidifying and eutrophying air pollution in life-cycle assessment. Journal of Industrial Ecology 4, 125-142.

IFA/FAO, 2001. Global estimates of gaseous emissions of HN_3, NO and N_2O from agricultural land. International Fertilizer Industry Association and Food and Agricultural Organization of the United Nations, Rome.

IFAP/Via Campesina, 2000. CSD Eighth Session, 24 April - 5 May 2000: Secretary-General's Note for the Multi-Stakeholder Dialogue on Sustainable Agriculture, Addendum 2: Discussion paper prepared by the International Federation of Agricultural Producers (IFAP) and Via Campesina. UN Commission on Sustainable Development (CSD), New York.

International Agri-Food Network, 2000. CSD Eighth Session, 24 April - 5 May 2000: Secretary-General's Note for the Multi-Stakeholder Dialogue on Sustainable Agriculture, Addendum 1: Discussion paper contributed by the International Agri-Food Network. UN Commission on Sustainable Development (CSD), New York.

IPCC, 1996. Revised 1996 IPCC Guidelines for National Greenhouse Gas Inventories – Reference Manual. Intergovernmental Panel on Climatic Change, Geneva.

IPCC, 2000. Good Practice Guidance and Uncertainty Management in National Greenhouse Gas Inventories. Intergovernmental Panel on Climatic Change (IPCC), Geneva.

IPCC, 2001. IPCC Third Assessment Report – Climate Change 2001. Intergovernmental Panel on Climatic Change(IPCC), Geneva.

Isermann, K. and Isermann, R., 1998. Food production and consumption in Germany: N flows and N emissions. Nutrient Cycling in Agroecosystems 52, 289-301.

ISO, 2003. ISO management systems website (http://www.iso.ch/iso/en/iso9000-14000/index.html). International Organization for Standardization (ISO), Geneva.

Iyer-Raniga, U. and Treloar, G., 2000. A context for participation in sustainable development. Environmental Management 26, 349-361.

Izac, A.-M.N. and Swift, M.J., 1994. On agricultural sustainability and its measurement in small-scale farming in sub-Saharan Africa. Ecological Economics 11, 105-125.

Jamieson, D., 1998. Sustainability and beyond. Ecological Economics 24, 183-192.

Jenkinson, D.S., 2001. The impact of humans on the nitrogen cycle, with focus on temperate arable agriculture. Plant and Soil 228, 3-15.

Johnson, D., 1997. The triangular distribution as a proxy for the beta distribution in risk analysis. The Statistician 46, 387-398.

Johnson, N.L. and Kotz, S., 1999. Non-smooth sailing or triangular distributions revisited after some 50 years. The Statistician 48, part 2, 179-187.

Jörß, W. and Handke, V., 2003. Appendix 1 to: Air Quality Control 2010. National Programme of the Federal Republic of Germany pursuant to Article 6 of Directive 2001/81/EC of 23 October 2001 on national emission ceilings for certain atmospheric pollutants (NEC Directive). Institut für Zukunftsstudien und Technologiebewertung (ITZ), Berlin.

Julius, C., Møller, C., Osterburg, B. and Sieber, S., 2003. Indikatoren einer nachhaltigen Landwirtschaft im Regionalisierten Agrar- und Umweltinformationssystem für die Bundesrepublik Deutschland (RAUMIS) (in German, with English abstract). Agrarwirtschaft 52, 185-194.

Kaiser, E.-A. and Ruser, R., 2000. Nitrous oxide emissions from arable soils in Germany – An evaluation of six long-term field experiments. Journal of Plant Nutrition and Soil Science 163, 249-260.

Kauffman, S.A., 1993. Origins of Order – Self-Organization and Selection in Evolution. Oxford University Press, New York.

Köhler, W., Schachtel, G. and Voleske, P., 1996. Biostatistik (in German). Springer, Berlin/Heidelberg.

Kreis Borken, 2002. Region, Zahlen, Fakten. Kreis Borken.

KTBL, 2002. Taschenbuch Landwirtschaft 2002/03 (in German). Kuratorium für Technik und Bauwesen in der Landwirtschaft (KTBL), Darmstadt.

Kürten, W. v. (ed.), 1977. Die naturräumlichen Einheiten auf Blatt 95/96 Kleve/Wesel (in German). Naturräumliche Gliederung Deutschlands, Geographische Landesaufnahme 1:200 000. Bundesforschungsanstalt für Landeskunde und Raumordnung, Bonn.

Kutsch, W.L., Steinborn, W., Herbst, M., Baumann, R., Barkmann, J. and Kappen, L., 2001. Environmental indication: A field test of an ecosystem approach to quantify biological self-organization. Ecosystems 4, 49-66.

LABO-AG, 2000. Cadmiumanreicherung in Böden/Einheitliche Bewertung von Düngemitteln (in German). Bericht für die 26. ACK der Umweltministerkonferenz am 11./12.10.2000. Bund/Länder-Arbeitsgemeinschaft Bodenschutz (LABO), Hamburg.

Lammers, M.A., 1999. Ergebnisse der Bodenuntersuchungen in Westfalen-Lippe (in German). Landwirtschaftskammer Westfalen-Lippe, Untersuchungszentrum Münster – LUFA, Münster.

Landau, S., Mitchell, R.A.C., Barnett, V., Colls, J.J., Craigon, J., Moore, K.L. and Payne, R.W., 1998. Testing winter wheat simulation models' prediction against observed UK grain yields. Agricultural and Forest Meteorology 89, 85-99.

Lange, H., 1999. Are ecosystems dynamical systems? International Journal of Computing Anticipatory Systems 3, 169-186.

LDS, 2001. Statistisches Jahrbuch Nordrhein-Westfalen 2001 (in German). Landesamt für Datenverarbeitung und Statistik Nordrhein-Westfalen (LDS), Düsseldorf.

Lefroy, R.D.B., Bechstedt, H.-D. and Rais, M., 2000. Indicators for sustainable land management based on farmer surveys in Vietnam, Indonesia, and Thailand. Agriculture, Ecosystems and Environment 81, 137-146.

Leithold, G., Hülsbergen, K.-J., Michel, D. and Schönmeier, H., 1997. Humusbilanzierung – Methoden und Anwendungen als Agrar-Umweltindikator. In: W. Diepenbrock, M. Kaltschmitt, H. Nieberg and G. A. Reinhardt (eds.), Umweltverträgliche Pflanzenproduktion – Indikatoren, Bilanzierungsansätze und ihre Einbindung in Ökobilanzen – Fachtagung am 11. und 12. Juli 1996 in Wittenberg. Schriftliche Fassung der Beiträge. Zeller, Osnabrück, pp. 43-54.

Lélé, S., and Norgaard, R.B., 1996. Sustainability and the scientist's burden. Conservation Biology 10, 354-365.

Lerdau, M.T., Munger, J.W. and Jacon, D.J., 2000. The NO_2 flux conundrum. Science 289, 2291-2293.

Lewandowski, I., Härdtlein, M. and Kaltschmitt, M., 1999. Sustainable crop production: definition and methodological approach for assessing and implementing sustainability. Crop Science 39, 184-193.

Lewis, K.A., Newbold, M.J. and Tzilivakis, J., 1999. Developing an emission inventory from farm data. Journal of Environmental Management 55, 183-197.

LK WL, 2002. Zahlen in Westfalen-Lippe (2000-2002) (in German). Landwirtschaftskammer Westfalen-Lippe (LK WL), Münster.

Lockeretz, W., 1988. Open questions in Sustainable Agriculture. American Journal of Alternative Agriculture 3, 174-181.

Loske, R. (ed.), 1996. Zukunftsfähiges Deutschland: ein Beitrag zu einer global nachhaltigen Entwicklung (in German). Birkhäuser, Basel.

Ludwig, D., Hilborn, R. and Walters, C., 1993. Uncertainty, resource exploitation, and conservation: Lessons learned from history. Science 260, 17, 36.

Lutz, W., Sanderson, W. and Schernov, S., 2001. The end of world population growth. Nature 412, 543-545.

Malthus, T., 1798. An Essay on the Principle of Population. J. Johnson, London.

Matson, P.A., Parton, W.J., Power, A.G. and Swift, M.J., 1997. Agricultural intensification and ecosystem properties. Science 277, 504-509.

Mayrhofer, P., 1997. Das Ökopunktemodell Niederösterreich – Aufbau und Umsetzung in der Agrarumweltpolitik (in German). In: W. Diepenbrock (ed.). Umweltverträgliche Pflanzenproduktion: Indikatoren, Bilanzierungsansätze und ihre Einbindung in Ökobilanzen. Deutsche Bundesstiftung Umwelt, Osnabrück. pp. 197-208.

Mayring, P., 1993. Qualitative Inhaltsanalyse. Deutscher Studienverlag, Weinheim.

Meadows, D.L., Meadows, D.H., Randers, J. and Behrens, W.W.I., 1972. The Limits of Growths. Universe Books, New York.

Merkle, A., and Kaupenjohann, M., 2000. Derivation of ecosystemic effect indicators – Method. Ecological Modelling 130, 39-46.

Miller, F.P., and Wali, M.K., 1995. Soils, land use and sustainable agriculture: A review. Canadian Journal of Soil Science 75, 413-422.

Mitchell, G., May, A. and McDonald, A., 1995. PICABUE: a methodological framework for the development of indicators of sustainable development. International Journal of Sustainable Development and World Ecology 2, 104-123.

Mosier, A.R., 2001. Exchange of gaseous nitrogen compounds between agricultural systems and the atmosphere. Plant and Soil 228, 17-27.

Mosier, A.R., Kroeze, C., Nevison, C., Oenema, O., Seitzinger, S. and Cleemput, O.v., 1998. Closing the global N_2O budget: nitrous oxide emissions through the agricultural nitrogen cycle. Nutrient Cycling in Agroecosystems 52, 225-248.

Mosier, A.R., Duxbury, J.M., Freney, J.R., Heinemeyer, O. and Minami, K., 1996. Nitrous oxide emissions from agricultural fields – assessment, measurement and mitigation. Plant and Soil 181, 95-108.

Müller, F., 1997. State-of-the-art in ecosystem theory. Ecological Modelling 100, 135-161.

Müller-Wenk, R., 1996. Political and scientific targets in distance-to-target valuation methods. In: A. Braunschweig, R. Förster, P. Hofstetter and R. Müller-Wenk (eds.), Developments in LCA Valuation. IWÖ-Diskussionsbeitrag 32. Institut für Wirtschaft und Ökologie, St. Gallen.

Münchhausen, H. v. and Nieberg, H., 1997. Agrar-Umweltindikatoren: Grundlage, Verwendungsmöglichkeiten und Ergebnisse einer Expertenbefragung (in German). In: W. Diepenbrock (ed.). Umweltverträgliche Pflanzenproduktion: Indikatoren, Bilanzierungsansätze und ihre Einbindung in Ökobilanzen. Deutsche Bundesstiftung Umwelt, Osnabrück. pp. 13-29.

Newman, M.E.J. and Palmer, R.G., 1999. Models of extinction – A review. SFI Working Papers 99-08-061. Santa Fe Institute, Santa Fe, MN.

NGOs at CSD8, 2000. CSD Eighth Session, 24 April - 5 May 2000: Secretary-General's Note for the Multi-Stakeholder Dialogue on Sustainable Agriculture, Addendum 4: Discussion paper contributed by the Non-Governmental Organizations. UN Commission on Sustainable Development (CSD), New York.

Nieder, R. and Richter, J., 2000. C and N accumulation in arable soils of West Germany and its influence on the environment – Developments 1970 to 1998. Journal of Plant Nutrition and Soil Science 163, 65-72.

Niu, W.-Y., Lu, J.J. and Khan, A.A., 1993. Spatial systems approach to sustainable development: A conceptual framework. Environmental Management 17, 179-186.

Norgaard, R.B., 1988. Sustainable development: a co-evolutionary view. Futures 20, 606-620.

Nowotny, H., Scott, P. and Gibbons, M., 2001. Re-thinking Science: Knowledge and the Public in the Age of Uncertainty. Polity Press, Cambridge.

OECD, 1995. Sustainable Agriculture. Concepts, Issues and Policies in OECD Countries. Organisation for Economic Co-operation and Development (OECD), Paris.

OECD, 1998a. Environmental Indicators. Towards Sustainable Development. Organisation for Economic Co-operation and Development (OECD), Paris.

OECD, 1998b. OECD Work on Sustainable Development: A Discussion Paper on Work to be Undertaken over the Period 1998-2001. Organisation for Economic Cooperation and Development (OECD), Paris.

OECD, 2000. Environmental Indicators for Agriculture. Methods and Results. Organisation for Economic Co-operation and Development (OECD), Paris.

O'Hara, S.U., 1996. Discursive ethics in ecosystems valuation and environmental policy. Ecological Economics 16, 95-107.

Öko-Institut, 1998. TEMIS (Total Emission Model for Integrated Systems), Version 2.18. Freiburg.

Öko-Institut, 2002. Updated results from GEMIS 4.14 (September 2002). GEMIS website (http://www.oeko.de/service/gemis/en/index.htm). Öko-Institut (Institute for Applied Ecology e.V.), Darmstadt.

Olson, J.S., Watts, J.A. and Allision, L.J., 1983. Carbon in Live Vegetation of Major World Ecosystems. Environmental Science Division, Publication No. 1997. Oak Ridge National Laboratory (ORNL), Oak Ridge, TN.

O'Neill, R.V., Johnson, A.R. and King, A.W., 1989. A hierarchical framework for the analysis of scale. Landscape Ecology 3, 193-205.

Oreskes, N., Schrader-Frechette, K. and Belitz, K., 1994. Verification, validation, and confirmation of numerical models in earth science. Science 263, 641-646.

Ortega y Gasset, J., 1944. Mission of the university. University Press, Princeton.

OSPAR, 2000. Quality Status Report 2000. OSPAR Commission, London.

Pain, B.F., Werden, T.V.D., Jarvis, S.C., Chambers, B.J., Smith, K.A., Demmers, T.G.M. and Phillips, V.R., 1997. Ammonia Emission Inventory for the UK. Institute of Grassland and Environmental Research (IGER), Okehampton.

PAN/EEB, 2002. Suggested text for a Directive of the European Parliament and of the Council on measures for reduction of use and of impacts to health and environment from pesticides. Pesticide Action Network (PAN) Europe and The European Environmental Bureau (EEB), London.

Patyk, A. and Reinhardt, G.A., 1997. Düngemittel – Energie- und Stoffstrombilanzen (in German). Vieweg, Braunschweig.

Paul, R. 1999. Zur Verdichtungsgefährdung im Rahmen des Bodengefügeschutzes auf großen Flächen (in German). Kolloquium "Einfluß der Großflächen-Landwirtschaft auf den Boden". Thüringer Ministerium für Landwirtschaft, Naturschutz und Umwelt. Jena, 6 May 1999.

Pearce, D.W., and Atkinson, D.D. 1993. Capital theory and the measurement of sustainable development: An indicator of weak sustainability. Ecological Economics 8, 103-108.

Pearce, D.W., and Turner, K. 1990. Economics of Natural Resources and the Environment. Johns Hopkins Press, Baltimore.

Peters, D., 2001. Presentation held at the 1st Iglo Sustainable Agriculture Workshop in Reken, Germany, Nov. 5-6, 2001.

Popp, J.H., Hyatt, E.D. and Hoag, D., 2000. Modeling environmental conditions with indices: a case study of sustainability and soil resources. Ecological Modelling 130, 131-143.

Popper, K., 1963. Conjectures and Refutations: The Growth of Scientific Knowledge. Routledge, London.

Posch, M., de Smet, P.A.M., Hettelingh, J.-P. and Downing, R.J. (eds.), 1995. Calculation and mapping of critical thresholds in Europe. Status Report 1995. Coordination Center for Effects (CCE), RIVM, Bilthoven.

Pretty, J., 1998. Supportive policies and practice for scaling up sustainable agriculture. In: N.G. Röling and M.A.E. Wagemakers (eds.), Facilitating sustainable agriculture: participatory, learning and adaptive management in times of environmental uncertainty. Cambridge Universtity Press, Cambridge, pp. 23-46.

Putnam, R., 2000. Bowling alone: the collapse and revival of American community. Simon and Schuster, New York.

Radermacher, W., Zieschank, R., Hoffmann-Kroll, R., van Nouhuys, J., Schäffer, D. and Seibel, S., 1998. Entwicklung eines Indikatorensystems für den Zustand der Umwelt in der Bundesrepublik Deutschland mit Praxistest für ausgewählte Indikatoren und Bezugsräume (in German). Metzler-Poeschel, Stuttgart.

Redfield, A.C., 1958. The biological control of chemical factors in the environment. American Scientist 46, 205-221.

Reganold, J.P., Glover, J.D., Andrews, P.K. and Hinman, H.R., 2001. Sustainability of three apple production systems. Nature 410, 926-929.

Renard, K.G., Foster, G.R., Weesies, G.A., McCool, D.K. and Yoder, D.C. (eds.), 1997. Predicting Soil Erosion by Water: A Guide to Conservation Planning with the Revised Universal Soil Loss Equation (RUSLE). USDA-ARS, Washington, DC.

Reynolds, C.S. and Davies, P.S., 2001. Sources and bioavailability of phosphorus fractions in freshwaters: a British perspective. Biological Reviews 76, 27-64.

Ricker, W.E., 1975. Computation and interpretation of biological statistics of fish populations. Bulletin of the Fisheries Research Board of Canada, 191, 2-6.

Rigby, D., Woodhouse, P., Young, T. and Burton, M., 2001. Constructing a farm level indicator of sustainable agricultural practice. Ecological Economics 39, 463-478.

Rinne, H. 1994. Bevölkerungs- und Wirtschaftsstatistik: Erläuterungen – Erhebungen – Ergebnisse (in German). Oldenbourg, München.

Rheinbaben, W. v., 1990. Nitrogen losses from agricultural soils through denitrification – a critical evaluation. Journal of Plant Nutrition and Soil Science 153, 157-166.

RNE, 2002. Goals for Sustainable Development in Germany – Main Points of Emphasis. Dialogue Paper of the German Council for Sustainable Development (RNE) (http://www.nachhaltigkeits rat.de/n_strategy/dialogue_paper/index.html).

Robertson, G.P., Paul, E.A. and Harwood, R.R., 2000. Greenhouse gases in intensive agriculture: Contributions of individual gases to radiative forcing of the atmosphere. Science 289, 1922-1925.

Rochette, P., Bochove, E.v., Prévost, D., Angers, D.A., Côté, D. and Bertrand, N., 2000. Soil carbon and nitrogen dynamics following application of pig slurry for the 19th consecutive year: II. Nitrous oxide fluxes and mineral nitrogen. Soil Science Society of America Journal 64, 1396-1403.

Rode, M., 1995. Quantifizierung der Phosphorbelastung von Fließgewässern durch landwirtschaftliche Flächennutzung (in German). Dissertation, University of Gießen.

Rogler, H. and Schwertmann, U., 1881. Erosivität der Niederschläge und Isoerodentenkarte Bayerns (in German). Zeitschrift für Kulturtechnik und Flurbereinigung 22, 99-112.

Röling, N.G., and Wagemakers, M.A.E., (eds.), 1998. Facilitating Sustainable Agriculture: Participatory, Learning and Adaptive Management in Times of Environmental Uncertainty. Cambridge University Press, Cambridge.

Rosenberg, A.A., Fogarty, M.J., Sissenwine, M.P., Beddington, J.R. and Shepherd, J.G., 1993. Achieving sustainable use of renewable resources. Science 262, 828-829.

Roßberg, D., Gutsche, V., Enzian, S. and Wick, M., 2002. NEPTUN 2000 – Erhebung von Daten zum tatsächlichen Einsatz chemischer Pflanzenschutzmittel im Ackerbau Deutschlands (in German, with English abstract). Berichte aus der BBA, Heft 98. Biologische Bundesanstalt für Land- und Forstwirtschaft (BBA), Braunschweig.

Ruelle, D., 1997. Chaos, predictability and idealization in physics. Complexity 3, 26-28.

Ruttan, V., 1994. Constrains on the design of sustainable systems of agricultural production. Ecological Economics 10, 209-219.

Sala, O.E., Chapin, F.S.I., Armestos, J.J., Berlow, E., Bloomfield, J., Dirzo, R., Huber-Sanwald, E., Huenneke, L.F., Jackson, R.B., Kinzig, A., Leemans, R., Lodge, D.M., Mooney, H.A., Oesterheld, M., Poff, N.L., Sykes, M.T., Walker, B.H., Walker, M. and Wall, D.H., 2000. Global biodiversity scenarios for the year 2100. Science 287, 1770-1774.

Sands, G.R. and Podmore, T.H., 2000. A generalized environmental sustainability index for agricultural systems. Agriculture, Ecosystems and Environment 79, 29-41.

Schachtschabel, P., Blume, H.-P., Hartge, K.H., Schwertmann, U. and Renger, M., 1984. Scheffer/ Schachtschabel: Lehrbuch der Bodenkunde (in German). Enke, Stuttgart.

Schäfer, D., Schoer, K., Seibel, S., Zieschank, R., Barkmann, J., Baumann, R., Meyer, U., Müller, F., Lehniger, K., Steiner, M. and Wiggering, H., 2002. Makroindikatoren des Umweltzustandes (in German). Beiträge zu den Umweltökonomischen Gesamtrechnungen, Band 10. Statistisches Bundesamt. Metzler-Poeschel, Wiesbaden.

Schüller, H., 1969: Die CAL-Methode, eine neue Methode zur Bestimmung des pflanzenverfügbaren Phosphats in Böden (in German). Zeitschrift für Pflanzenernährung und Bodenkunde 123, 48-63.

Schwertmann, U., Vogl, W. and Kainz, M., 1987. Bodenerosion durch Wasser – Vorhersage des Abtrags und Bewertung von Gegenmaßnahmen. Ulmer, Stuttgart.

Seitzinger, S. and Kroeze, C., 1998. Global distribution of nitrous oxide production and N inputs in freshwater and costal marine ecosystems. Global Biogeochemical Cycles 12, 93-113.

Skiba, U., Fowler, D. and Smith, K.A., 1997. Nitric oxide emissions from agricultural soils in temperate and tropical climates: Sources, controls and mitigation options. Nutrient Cycling in Agroecosystems 48, 75-90.

Smakhtin, V., Revenga, C. and Döll, P., 2004. A pilot global assessment of environmental water requirements and scarcity. Environmental Conservation (submitted).

Smith, C.S., and McDonald, G.T., 1998. Assessing the sustainability of agriculture at the planning stage. Journal of Environmental Management 52, 12-37.

Smyth, A.J. and Dumanski, J., 1995. A framework for evaluating sustainable land management. Canadian Journal of Soil Science 75, 401-406.

Sommer, S.G. and Hutchings, N.J., 2001. Ammonia emissions from field applied manure and its reduction – invited paper. European Journal of Agronomy 15, 1-15.

SRU, 1985. Sondergutachten 1985: Umweltprobleme der Landwirtschaft (in German). Der Rat von Sachverständigen für Umweltfragen (SRU). W. Kohlhammer, Stuttgart.

SRU, 1994. Umweltgutachten 1994 des Rates von Sachverständigen für Umweltfragen: Für eine dauerhaft-umweltgerechte Entwicklung (in German, English excerpts published as SRU: Envrionmental report 1994: In pursuit of sustainable environmentally sound development. available via http://www.umweltrat.de). Metzler-Poeschel, Stuttgart.

Stanners, D. and Bourdeau, P. (eds.), 1995. Europe's Environment – The Dobris Assessment. European Environment Agency, Copenhagen.

Steinborn, W. and Svirezhev, Y., 2000. Entropy as an indicator of sustainability in agro-ecosystems: North Germany case study. Ecological Modelling 133, 247-257.

Stevenson, M., and Lee H., 2001. Indicators of sustainability as a tool in agricultural development: Partitioning scientific and participatory processes. International Journal of Sustainable Development and World Ecology 8, 57-65.

Stobbelaar, D.J., and van Mansvelt, J.D., 2000. The process of landscape evaluation: Introduction to the 2nd special AGEE issue of the concerted action 'The landscape and natural production capacity of organic/sustainable types of agriculture'. Agriculture, Ecosystems and Environment 77, 1-15.

Stockle, C.O., Papendick, R.I., Saxton, K.E., Campbell, G.S., and van Evert, F.K., 1994. A framework for evaluating the sustainability of agricultural production systems. American Journal of Alternative Agriculture 9, 45-50.

Sum, M.C. and Hills, P., 1998. Interpreting sustainable development. Journal of Environmental Science 10, 129-143.

Sutton, M.A., Place, C.J., Eager, M., Fowler, D. and Smith, R.I, 1995. Assessment of the magnitude of ammonia emissions in the United Kingdom. Atmospheric Environment 29, 1393-1411.

Syers, J.K. and Springett, J.A., 1984. Earthworms and soil fertility. Plant and Soil 76, 93-104.

Syers, J.K., Hamblin, A. and Pushparajah, E., 1995. Indicators and thresholds for the evaluation of sustainable land management. Canadian Journal of Soil Science 75, 423-428.

Tacconi, L., 1998. Scientific methodology for ecological economics. Ecological Economics 27, 91-105.

Taylor, D.C., Mohamed, Z.A., Shamsudin, M.N., Mohayidin, M.G. and Chiew, E.F.C., 1993. Creating a farmer sustainability index: A Malaysian case study. American Journal of Alternative Agriculture 8, 175-184.

Tellarini, V., and Caporali, F., 2000. An input/output methodology to evaluate farms as sustainable agroecosystems: an application to farms in central Italy. Agriculture, Ecosystems and Environment 77, 111-123.

TFZ, 2004. Website of the TZF (in German): http://www.landwirtschaft.bayern.de/alle/cgi-bin/go.pl?region=home&page=http://www.stmlf.bayern.de/tfz/tec/oel/oel_char_eigenschaften.htm. Technologie- und Förderzentrum (TFZ) im Kompetenzzentrum für Nachwachsende Rohstoffe Bayern.

Thies, C. and Tscharntke, T., 1999. Landscape structure and biological control in agroecosystems. Science 285, 893-895.

Tilman, D., Reich, B.P., Knops, J., Wedin, D., Mielke, T. and Lehman, C., 2001. Diversity and productivity in a long-term grassland experiment. Science 294, 843-845.

Troeh, F.R., Hobbs, J.A. and Donahue, R.L., 1998. Soil and Water Conservation. Prentice Hall, Upper Saddle River, N.Y.

UBA (ed.), 1997. Nachhaltiges Deutschland – Wege zu einer dauerhaft-umweltgerechten Entwicklung (in German). Umweltbundesamt (UBA). Erich Schmidt Verlag, Berlin.

UBA, 2001a. Grundsätze und Maßnahmen für eine vorsorgeorientierte Begrenzung von Schadstoffeinträgen in landbaulich genutzte Böden. UBA-Texte 59/01 (in German). Umweltbundesamt (UBA), Berlin.

UBA, 2001b. Ziele für die Umweltqualität – Eine Bestandsaufnahme (in German). Erich Schmidt Verlag, Berlin.

UBA, 2002. Environmental Data Germany 2002. Umweltbundesamt (UBA), Berlin.

UMK-AMK-LABO-AG, 2000. Cadmiumanreicherung in Böden/Einheitliche Bewertung von Düngemitteln (in German). Bericht für die 26. ACK der Umweltministerkonferenz am 11./12.10.2000, Bund/Länder-Arbeitsgemeinschaft Bodenschutz (LABO), Hamburg.

UN, 2003. United Nations Division for Sustainable Development (UN-DSD). Website: http://www.un.org/esa/sustdev/.

UN-CED, 1992. Rio Declaration on Environment and Development. Report of the UNCED, Annex I. Rio de Janeiro, 3-14 June 1992. United Nations Conference on Environment and Development (UN-CED).

UNEP, 2003. GLOBIO: Global Methodology for Mapping Human Impacts on the Biosphere. Website: http://www.globio.info/. UNEP's Division for Early Warning and Assessment (UNEP-DEWA), UNEP/GRID-Arendal, World Conservation Monitoring Centre (UNEP-WCMC) and the National Institute of Public Health and the Environment in the Netherlands (RIVM).

UN-FCCC, 2003. Greenhouse Gas Inventory Database. Website: http://ghg.unfccc.int/. The UN Framework Convention on Climatic Chance (UN-FCCC).

Unilever, 2002. Growing for the Future II – Unilever and sustainable agriculture (also available at www.growingforthefuture.com). Unilever, Rotterdam.

Unilever, 2004. Unilever Environment and Society. Website: www.unilever.com/environmentsociety /sustainability/.

USGS, 2001. Mineral Commodity Summaries 2001. U.S. Geological Survey (USGS), Washington, DC.

Van Breemen, N., Driscoll, C.T. and Mulder, J., 1984. Acidic deposition and internal proton sources in acidification of soils and waters. Nature 307, 599-604.

van Mansvelt, J.D., 1997. An interdisciplinary approach to integrate a range of agro-landscape values as proposed by representatives of various disciplines. Agriculture, Ecosystems and Environment 63, 233-250.

van Mansvelt, J.D., and van der Lubbe, M.J., 1999. Checklist for Sustainable Landscape Management Elsevier, Amsterdam.

VDLUFA, 1991. VDLUFA-Methodenbuch. Band I: Die Untersuchung von Böden (in German). VDLUFA-Verlag, Darmstadt.

Veldkamp, E. and Keller, M., 1997. Fertiliser-induced nitric oxide emissions from agricultural soils. Nutrient Cycling in Agroecosystems 48, 69-77.

Vester, F., 1999. Die Kunst vernetzt zu denken. Bericht an den Club of Rome (in German). DVA, Stuttgart.

Vestreng, V., 2003. EMEP/MSC-W Technical report. Review and Revision. Emission data reported to CLRTAP. MSC-W Status Report 2003. EMEP/MSC-W Note 1/2003.

Vitousek, P.M., Mooney, H.A., Lubchenco, J. and Melillo, J.M., 1997. Human domination of earth's ecosystems. Science 277, 494-499.

Wackernagel, M. and Yount, J.D., 1998. The ecological footprint: an indicator of progress towards regional sustainability. Environmental Monitoring and Assessment 51, 511-529.

Wagner-Riddle, C. and Thurtell, G.W., 1998. Nitrous oxide emissions from agricultural fields during winter and spring thaw as affected by management practices. Nutrient Cycling in Agroecosystems 52, 151-163.

Walters, C., 1986. Adaptive management on renewable resources. Macmillan, New York.

Walther, G.-R., Post, E., Convey, P., Menzel, A., Parmesan, C., Beebee, T.J.C., Fromentin, J.-M., Hoegh-Guldberg, O. and Bairlein, F., 2002. Ecological response to recent climate change. Nature 416, 389-395.

WCED, 1987. Our common future. The World Commission on Environment and Development (WCED), Oxford University Press, Oxford.

Weigel, A., Russow, R. and Körschens, M., 2000. Quantification of airborne N-input in long-term field experiments and its validation through measurements using ^{15}N isotope dilution. Journal of Plant Nutrition and Soil Science 163, 261-265.

Weil, R.R., 1990. Defining and using the concept of sustainable agriculture. Journal of Agronomic Education 19, 126-130.

Wenz, P.S., 1999. Pragmatism in practice: the efficiency of sustainable agriculture. Environmental Ethics 21, 391-410.

Werner, D. and Paul, R., 1999. Kennzeichnung der Verdichtungsgefährdung landwirtschaftlich genutzter Böden (in German, with English summary). Wasser und Boden 51, 10-14.

Werner, W., 1997. Implementation and efficiency of counter-measures against diffuse nitrogen an phosphorus input into groundwater and surface waters from agriculture. In: E. Romstad, J. Simonsen and A. Vatn (eds.): Controlling Mineral Emissions in European Agriculture. CAB International, Wallingford, pp. 73-88.

Werner, W., Olfs, H.W., Auerswald, K. and Isermann, K., 1991. Stickstoff- und Phosphoreinträge in Oberflächengewässer über “diffuse Quellen” (in German). In: A. Hamm (ed.), Studie über Wirkungen und Qualitätsziele von Nährstoffen in Fließgewässern. Academia Verlag, Sankt Augustin, pp. 665-764.

WHO, 1997. Guidelines for Predicting Dietary Intake of Pesticide Residues (revised). Programme of Food Safety and Food Aid, World Health Organisation (WHO), Geneva.

WHO, 1999. Principles for the assessment of risks to human health from exposure to chemicals. Environmental Health Criteria No. 210. World Health Organization (WHO), Geneva.

Wiens, J.A., 1987. Spatial scaling in ecology. Functional Ecology 3, 385-397.

Wiersum, K.F., 1995. 200 years of sustainability in forestry. Lessons from history. Environmental Management 19, 321-329.

Wignarajah, P., 2001. Social/Human Capital & Local Economy Indicators. A report to the Birds Eye Wall's Sustainable Agriculture Project 06/2001. Unilever R&D Colworth.

Wirén-Lehr, S. v., 2001. Sustainability in agriculture – an evaluation of principal goal-oriented concepts to close the gap between theory and practice. Agriculture, Ecosystems and Environment 84, 115-129.

Wischmeier, W. H. and Smith, D.D., 1978: Predicting rainfall erosion losses – a guide to conservation planning. USDA, Agricultural Handbook No. 537. United States Department of Agriculture, Washington, DC.

WMO, 2002. Scientific Assessment of Ozone Depletion: 2002. Global Ozone Research and Monitoring Project – Report No. 47. World Meteorological Organization (WMO).

WRI, 2003a. Carbon Emissions from Energy Use and Cement Manufacturing, 1850 to 2000. Available on-line through the Climate Analysis Indicators Tool (CAIT) at: http://cait.wri.org. World Resources Institute (WRI), Washington, DC.

WRI, 2003b. Watersheds of the World_CD. World Conservation Union (IUCN), International Water Management Institute (IWMI), Ramsar Convention Bureau, and World Resources Institute (WRI), Washington, DC. Available online at: http://www.iucn.org/themes/wani/eatlas/.

Wynne, B., 1992. Misunderstood misunderstanding: social identities and public uptake of science. Public Understanding of Science 1, 281-304.

Yienger, J.J. and Levy, H. II, 1995: Empirical model of global soil biogenic NO_x emissions. Journal of Geophysical Research 100: 11447-11464.

Young, J.W.S, 1997. A framework for the ultimate environmental index – Putting atmospheric change into context with sustainability. Environmental Monitoring and Assessment 46, 135-149.

Youngs, G.A., Jr., Goreham, G.A., and Watt, D.L., 1991. Classifying conventional and sustainable farmers: Does it matter how you measure? Journal of Sustainable Agriculture 2, 91-115.

Yuan, T.L., Gammon, N.J. and Leighty, R.G., 1966. Relative contribution of organic and clay fractions to cation-exchange capacity of sandy soils from several soil groups. Soil Science 104, 123-128.

Zinck, J.A., and Farshad, A., 1995. Issues of sustainability and land management. Canadian Journal of Soil Science 75, 407-412.

ZMP, 2003. Marktberichte Gemüse 2003 (in German). Zentrale Markt- und Preisberichtsstelle für Erzeugnisse der Land-, Forst und – Ernährungswirtschaft (ZMP), Bonn.

Zwiers, F.W., 2002. Climate change: The 20-year forecast. Nature 416, 690-691.

Annex I

Soil Input/Output Ratio (mass balance)

$$\text{Soil Input/Output Index } (-) = \frac{\text{soil erosion rate (t ha}^{-1}\text{yr}^{-1})}{\text{soil formation rate (t ha}^{-1}\text{yr}^{-1})}$$

For data used refer to Table A.1.

Soil Compaction Index (potential compaction, modified after Werner & Paul, 1999)

$$\text{Soil Compaction Index } (-) = \frac{\sum P_e}{\sum R_e}$$

where

P_e (kPa yr^{-1}) = ground pressures of capacity exceeding passes per year

R_e (kPa yr^{-1}) = soil resistances of capacity exceeding passes per year

'capacity exceeding passes' = field passes, for which the ground pressure in 5 cm depth ≥ soil resistance in 5 cm depth.

For data used refer to Table A.2.

Proton Input/Output Ratio (mass balance, modified after Van Breemen et al., 1984)

$$\text{Proton Input/Output Ratio } (-) = \frac{\text{Sum of proton inputs (kg H}^{+}-\text{eq ha}^{-1}\text{ yr}^{-1})}{\text{Sum of proton outputs (kg H}^{+}-\text{eq ha}^{-1}\text{ yr}^{-1})}$$

For data used refer to Table A.3.

Heavy Metal Accumulation Index (mass balance)

$$\text{Heavy Metal Accumulation Index } (-) = \frac{\sum (I_m / T_m)}{\sum (O_m / T_m)}$$

where

I_m (g ha^{-1} yr^{-1}) = inputs of accumulating metal *m* per year

O_m (g ha^{-1} yr^{-1}) = outputs of accumulating metal *m* per year

T_m (mg kg^{-1}) = Soil Precautionary Value (*Vorsorgewert*) for metal *m*

'accumulating metals' = heavy metals, for which inputs ≥ outputs.

For data used refer to Table A.4.

Soil Organic Matter (SOM) Output/Input Ratio (mass balance, modified after Leithold et al., 1997)

$$\text{SOM Output/Input Ratio } (-) = \frac{\text{Sum of SOM inputs (HU ha}^{-1}\text{ yr}^{-1})}{\text{Sum of SOM outputs (HU ha}^{-1}\text{ yr}^{-1})}$$

where HU = ‘Humus Units’.

For data used refer to Table A.5.

Nutrient Output/Input Ratios (mass balance)

$$\text{Nutrient Input/Output Ratio } (-) = \frac{\text{Sum of nutrient inputs (kg ha}^{-1}\text{ yr}^{-1})}{\text{Sum of nutrient outputs (kg ha}^{-1}\text{ yr}^{-1})}$$

For data used refer to Table A.6.

Naturalness Degradation Potential (modified after Brentrup et al., 2002)

$$\text{Naturalness Degradation Potential } (-) = \frac{\sum (A_l \times f_l)}{\sum A_l}$$

where

A_l (ha yr^{-1}) = area occupied by land use type *l*

f_l (–) = hemeropby factor for land use type *l*.

For data used refer to Table A.7.

Global Warming Potential

$$\text{Global Warming Potential (kg CO}_2\text{-eq ha}^{-1}\text{ yr}^{-1}) = \sum (GHG_i \times f_i)$$

where

GHG_i (kg ha^{-1} yr^{-1}) = emission of greenhouse gas *i*

f_i (CO_2-eq) = global warming potential of greenhouse gas *i* (in kg CO_2-eq over 100 years).

Values for f_i from IPCC (2001): CO_2 = 1; CH_4 = 23; N_2O = 296.

Acidification Potential (after Brentrup et al., 2003)

$$\text{Acidification Potential (kg SO}_2\text{-eq ha}^{-1}\text{ yr}^{-1}) = \sum (E_i \times f_i)$$

where

E_i (kg ha^{-1} yr^{-1}) = emission of acidifying substance *i*

f_i (–) = acidification potential of acidifying substance *i* (in kg SO_2-eq).

f_i for emission from Germany (after Huijbregts et al., 2000):
NH_3 = 1.50; NO_X = 0.53; SO_2 = 1.30.

Terrestrial Eutrophication Potential (after Brentrup et al., 2003)

Terrestrial Eutrophication Potential (kg NO_X-eq ha^{-1} yr^{-1}) $= \sum (E_i \times f_i)$

where

E_i (kg ha^{-1} yr^{-1}) = emission of eutrophying substance *i*

f_i (–) = eutrophication potential of substance *i* (in kg NO_X-eq).

f_i for emission from Germany (after Huijbregts et al., 2000):

NH_3 = 4.6; NO_X = 1.5.

Marine Eutrophication Potential (after Brentrup et al., 2003)

Marine Eutrophication Potential (kg PO_4-eq ha^{-1} yr^{-1}) $= \sum (E_i \times f_i)$

where

E_i (kg ha^{-1} yr^{-1}) = emission of eutrophying substance *i*

f_i (–) = eutrophication potential of substance *i* (in kg PO_4-eq).

f_l for emission from Germany (after Huijbregts and Sepällä, 2001):

NH_3 = 0.05; NO_X = 0.02; P = 3.06, NO_3 = 0.42.

Pesticide Use Intensity Index (*Behandlungsindex*, modified after Gutsche & Enzian, 2002)

$$\text{Pesticide Use Intensity Index } (–) = \sum_i \frac{R_{a,i}}{R_{m,i}}$$

where

$R_{a,i}$ (kg ha^{-1} yr^{-1}) = actual application rate of pesticide *i*

$R_{m,i}$ (kg ha^{-1} yr^{-1}) = maximum registered application rate of pesticide *i*.

For data used refer to Table A.8.

Annex II

Table A.1 Data used for calculating soil balances. Soil erosion rates (t ha^{-1} yr^{-1}) and factors for the Revised Universal Soil Loss Equation (RUSLE; Renard et al., 1997) as adapted to German conditions (Hennings, 2000). The RUSLE factors are dimensionless. A soil formation rate of 1 t ha^{-1} yr^{-1} was assumed (Troeh et al., 1998).

		Erosivity factor	Soil factor	Cover factor	Topography fctrs.	Protection fctr.	Erosion rate
		R[a]	K[b]	C[c]	LS[d]	P	E
Spinach	(S)	95	0.19	0.47	0.13	1.0	1.1
Herbs (parsley)	(H)	95	0.19	0.25	0.13	1.0	0.6
Maize[e]	(M)	95	0.19	0.55	0.13	1.0	1.3
Cereals	(C)	95	0.19	0.07	0.13	1.0	0.2
Sugar beets[e] or potatoes	(B/P)	95	0.19	0.19	0.13	1.0	0,4
Rotation 1	S-C-M-M	95	0.19	0.41	0.13	1.0	1.0
Rotation 2	S-H-B/P-M	95	0.19	0.37	0.13	1.0	0.9
Rotation 3	S-H-M-M	95	0.19	0.46	0.13	1.0	1.1
Standard rotation[f]		95	0.19	0.41	0.13	1.0	1.0

[a]R = –53.23+0.365 * [summer precipitation], where summer precipitation (mm) is rainfall from May to October (Hennings, 2000, p. 157, Equation c).

[b]Area weighted mean of K factors for different soil types with 62% sand (K=0.16), 24% loamy sand (K=0.25), 8% sandy loam (K=0.27), 5% silty loam and clay (K=0.19) and 1% others. Soil type distribution the County of Borken from Lammers (1999), K factors from Hennings (2000)

[c]Mean of different sowing dates and different soil cover situations, calculated after Auerswald and Kainz (1998) and Auerswald and Schwab (1999).

[d]Assuming an erosive slope length of 310 m at 1.05% inclination. Inclination is area weighted mean of topography in the municipality of Velen, which was assumed to be characteristic the County of Borken (Olaf Nölle, 2003; pers. communication).

[e]Conventional tillage.

[f]Weighted mean of 80% Rotation 1, 10% Rotation 2 and 10% Rotation 3.

Table A.2 Data used for calculating the Soil Compaction Index.

Operation	Machinery used	Weight	Tyre contact area [a]	Ground pressure [b] at depth of 0 cm	Ground pressure [b] at depth of 15 cm [c]	Soil resistance[d] at depth of 15 cm [c]
		kg	cm²	kPa		
Mulch maize stubbles	Tractor, 88 kW	5,500	4,580	109	100	110
	Mulcher	850				
Plough	Tractor, 88 kW	5,500	4,580	111	102	110
	Plough	950				
Cultivate	Tractor, 62 kW	4,250	1,510	265	191	110
	Cultivator	850				
Sow	Tractor, 88 kW	5,500	4,580	131	120	110
	Drill	2,118				
Spread synthetic fertiliser (solid)	Tractor, 62 kW	4,250	1,510	335	241	110
	Fertiliser spreader	1,750				
Spread synthetic fertiliser (liquid)	Tractor, 62 kW	4,250	1,510	307	221	110
	Sprayer	1,325				
Spread lime	Tractor, 88 kW	5,500	4,580	126	116	110
	Lime spreader	7,400	4,752	122	113	110
Spread liquid manure	Tractor, 88 kW	5,500	4,580	142	130	110
	Slurry spreader	11,100	5,284	165	154	110
Spray pesticide	Tractor, 62 kW	4,250	1,510	307	221	110
	Sprayer	1,325				
Spread solid manure or compost	Tractor, 88 kW	5,500	4,580	109	101	110
	Spreader	3,500	4,752	58	53	110
Mechanical weeding	Tractor, 37 kW	3,000	1,510	169	122	110
	Weeding hoe	250				
Harvest spinach or fine herbs	Spinach harvester	9,700	3,927	194	175	110
Transp. spinach or herbs on field	Tractor, 154 kW	8,400	6,637	136	132	110
	Chassis and container	19,000	17,671	92		110
Harvest cereal or maize	Combine harvester	10,700	7,918	106	102	160[d]
Harvest sugar beets	Sugar beet harvester	26,500	31,416	83	82	110
Harvest potatoes	Tractor, 88 kW	5,500	4,580	148	136	110

[a]Contact area approximated by circle area with diameter of tyre width, multiplied by the number of tyres:
Contact area = [0.25*π*(tyre width)²]*number of tyres (Paul, 1999).
[b]Quotient of gravity force (Weight*9.81) and contact area. Assumed weight distribution: Tractors: 80% on rear 20% on front axle. Mounted implements and equipment: 100% on rear axle of tractor. Trailed equipment: 20% up to a max of 2.5 t on rear axle of tractor, remaining weight on own axle(s). Contact area pressure of front tyres assumed to always stay below 110 kPa.
[c]Pressure propagation into soil after Newmark (Equation A.1, next page).
[d]Soil mechanical resistance after Horn et al. (1996) and Werner & Paul (1999) at soil moisture 70% of field capacity (cereal harvest assumed at 50% of field capacity).
Data source: Information on machinery and tyres used locally from own data and by Dr Norbert Uppenkamp (2002, 2003; personal communication). Machinery weights from manufacturers' catalogues.

Newmark equation for pressure propagation into soil (as cited by Paul, 1999):

$$\sigma_Z = \sigma_0 \left[1 - \frac{1}{\sqrt{\left[\left(\frac{R}{Z} \right)^2 + 1 \right]^{vk}}} \right] \qquad \text{(A.1)}$$

where

σ_Z	Pressure in depth Z (kPa)
σ_0	Soil surface contact pressure (kPa)
R	Radius of tyre (cm)
vk	Concentration factor, here assumed to be 3.5 (dimensionless).

Table A.3 Data used for calculating the Soil Proton Input/Output Ratio.

Proton source [reference unit]	kmol H^+-eq. kg^{-1}	Atmos. depos.	Spinach	Cereals	Maize	Herbs	S. beets, potatoes[a]
			kg ha^{-1} yr^{-1}				
N Deposition [NO_X-N, NH_Y-N]	0.0714	13					
Sulphur deposition [SO_2]	0.0313	7					
Cal. amm. nitr. (20% CaO)[CAN-N]	0.0582		94	65	25	135	45
Urea ammonium nitrate [UAN-N]	0.0714		108				
Di-ammonium phosphate [DAP-N]	0.1429			12			
Slurry (70% TAN) [N]	0.1000			120	144		90
N volatilisation [NH_3-N]	–0.1429		11	15	17	4	11
Denitrification [N_2-N]	–0.0714		13	12	11	9	9
N-uptake [N][b]	–0.0357		130	145	130	62	80
Liming [CaO]	–0.0179		350	200	200	200	200

[a]Average of the two crops.
[b]For balance calculation it was assumed that: (1) all NH_X is fully nitrified (after subtraction of NH_3 losses), (2) all plant uptake is NO_3, (3) 50% of NO_3 taken up by plant lead to proton consumption (cf. Van Breemen et al., 1984).
Data source: Atmospheric deposition data from EMEP (2002b), fertilisation and crop data based on official statistics (LK WL, 2002) and expert judgement (Ferdinand Pollert, 2003; pers. communication). Gaseous N losses estimated as described in Chapter 4.

Table A.4 Data used for calculating the Heavy Metal Accumulation Index.

	Cd	Cr	Cu	Hg	Ni	Pb	Zn
Inputs and outputs	g ha^{-1} yr^{-1}						
Atmospheric deposition [a]	0.20	2.1	13.0	0.1	9.9	5.3	229
Leaching loss	–0.40	–13.1	–11.4	–0.4	–25.4	–0.8	–54
Plant off-take							
Spinach (42 t, 6% DM)	–1.90	–10.8	–41.8	n.a.	–2.0	–1.0	–158
Cereals (8 t FM, 86% DM)	–0.25	–0.8	–37.8	n.a.	–5.4	–3.8	–295
Grain Maize (9.7 t FM, 65% DM)	–0.13	–5.9	–11.5	n.a.	–10.6	–13.1	–180
Silage Maize (46 t FM, 33% DM)	–0.31	–14.1	–27.8	n.a.	–25.5	31.6	–434
Herbs	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.
Sugar beets (53 t, 20% DM)	–1.34	–13.4	–52.4	n.a.	–26.8	–3.4	–171
Potatoes (40 t FM, 20% DM)	–2.64	13.1	–59.5	n.a.	–16.1	–15.5	–64
N fertilisers	g per t N						
Urea ammonium nitrate (UAN)	0.1	4	18	n.a.	1	0.6	6
Calcium ammonium nitrate (CAN)	0.9	32	15	0.01	14	79.3	142
NP fertiliser	0.4	4	1	<0.01	1	0.2	7
P fertilisers	g per t P						
Single super phosphate	137.5	1,451	219	n.a.	367	235.5	3,004
Triple super phosphate	136.4	1,466	139	0.20	185	61.1	2,490
NP fertiliser	91.1	910	214	0.20	179	54.8	1,504
K fertilisers	g per t K						
KCl (40% K_2O)	0.4	11	9	0.06	5	1.5	11
KCl (*Patentkali*)	0.4	21	14	n.a.	16	10.0	70
Lime	g per t CaO						
Limestone calcium carbonate	0.6	14	15	0.07	11	10.9	76
Lime from slags (*Konverterkalk*)	0.9	3,124	17	0.04	20	42.2	21
Organic fertilisers	g per m^3 FM (7% DM content)						
Cattle slurry	0.3	7	45	0.06	6	7.7	270
Pig slurry	0.4	9	309	0.02	10	6.2	585
Mixed slurry	0.3	8	177	0.04	8	7.0	264
	g per t (DM)						
Compost (*Grünschnittkompost*)	0.4	22	29	0.08	7.8	34.1	145
Soil precautionary value	mg per kg soil (*Aqua regia* extraction)						
(*Vorsorgewert*)	0.4	40	30	20	0.1	15	60

[a]Deposition of Cd, Hg and Pb from EMEP 50 km x 50 km grid, other values national estimates.
[b]Toxicological precautionary values after German Soil Protection Directive (BBodSchV, 1999).

Data source: Deposition data for Cd, Hg and Pb from EMEP (2002b), plant off-take for spinach after own data, Bergmann (1992) and Delschen & Leisner-Saaber (1998), compost data from Hansjörg Komnik (2004; personal communication). All other data from UBA (2001).

Table A.5 Data used for calculating the Soil Organic Matter (SOM) Input/Output Ratio.

		SOM loss (HU [a] ha^{-1} yr^{-1})	Crop yield (t ha^{-1} yr^{-1} FM)	Harvest residues (t ha^{-1} yr^{-1} DM)	Replacement coefficient (—)	SOM replacement (HU [a] ha^{-1} yr^{-1})
Main crops						
Spinach	(S)	1.40	42.0	2.5	0.13	0.3
Cereals	(C)	0.70	8.0	7.0	0.14	1.0
Grain maize	(Mg)	0.70	9.7	4.0	0.13	0.5
Silage maize	(Ms)	1.35	46.0	1.0	0.13	0.1
Herbs (parsley)	(H)	0.70	12.5	1.0	0.13	0.1
Sugar beets	(B)	2.30	53.0	7.5	0.13	1.0
Potatoes	(P)	1.80	40.0	2.5	0.13	0.3
Winter cover crops [b]						
Rye		0.30	0.0	2.0	0.13	0.3
Fodder radish, mustard, turnip		–0.15 [c]	0.0	3.5	0.13	0.5
Rotation 1	S-C-Mg-Ms	1.04				0.80
Rotation 2	S-H-B/P-Ms	1.38				0.70
Rotation 3	S-H-Mg-Ms	1.04				0.75
Standard rotation [d]		1.07				0.77
Organic fertilisers						
Slurry (mixed, 7% DM)					0.014	
Plant biomass (DM)					0.13	
Solid manure (cattle)					1.0	
Compost					1.1	

[a] HU: humus unit, equivalent to 1 t solid manure with 580 kg C.
[b] Winter cover crops not harvested, biomass incorporated into field. During a four-year rotation, one cover crop of rye, two of radish or mustard and one year without a cover crop are assumed.
[c] Negative SOM loss denotes SOM gain.
[d] Weighted mean of 80% Rotation 1, 10% Rotation 2 and 10% Rotation 3.

Data source: Default values for SOM balance after Leithold et al. (1997). Crop yields and harvest residues from own data, official statistics (LK WL, 2002) and Ferdinand Pollert (2003; personal communication).

Table A.6 Crop cultivation data and nutrient offtake with harvested product.

Crop	Fertilisation kg ha^{-1} yr^{-1}		Cultivation[e]		Yield t ha^{-1} yr^{-1}		Nutrient offtake kg ha^{-1} yr^{-1}		
					FM	DM	N	P	K
Spinach	350 CaO[a] 108 N (UAN) 94 N (CAN)	245 K (KCl)	2 Plo 3 Cult 2.75 Sow	1 Lime 6 Fert[c] 6 Pest	42	2.5	130	15	186
Herbs (Parsley)	200 CaO (LC)[b] 135 N (CAN)	133 K (KCl)	1 Plo 2 Cult 1.75 Sow	0.33 Lime[b] 2 Fert 2 Pest	12.5	1.6	62	7	83
Maize[f]	200 CaO (LC)[b] 144 N (slurry) 25 N (min NP)[d]	31 P (slurry) 11 P (min NP)[d] 108 K (slurry) 33 K (KCl)	1 Plo 1.75 Cult 1.75 Sow 1 Mulching	0.33 Lime[b] 1 Slur 1 Fert 1.2 Pest	39	13	145	35	140
Winter wheat	200 CaO (LC)[b] 120 N (slurry) 12 N (DAP) 80 N (CAN)	26 P (slurry) 13 P (DAP) 90 K (slurry)	1 Plo 1.75 Cult 1.75 Sow	0.33 Lime[b] 1 Slur 2 Fert 3.5 Pest	8	6.9	130	28	40
Winter barley	200 CaO (LC)[b] 120 N (slurry) 12 N (DAP) 50 N (CAN)	26 P (slurry) 13 P (DAP) 90 K (slurry)	1 Plo 1.75 Cult 1.75 Sow	0.33 Lime[b] 1 Slur 2 Fert 2.5 Pest	8	6.9	130	28	40
Sugar beets	200 CaO (LC)[b] 120 N (slurry) 50 N (CAN)	26 P (slurry) 90 K (slurry) 90 K (KCl)	1 Plo, 2 Cult 1. 75 Sow 1 Hoe	0.33 Lime[b] 1 Slur, 1.5 Fert 3 Pest	53	10.6	80	23	110
Starch potatoes	200 CaO (LC)[b] 60 N (slurry) 40 N (CAN)	13 P (slurry) 45 K (slurry) 100 K (K_2SO_4)	1 Plo, Cult 1. 75 Sow 1 Ridging	0.33 Lime[b] 1 Slur, 1.5 Fert 8 Pest	40	8.0	80	24	200

[a] Both limestone calcium carbonate and smelter lime (*Konverterhüttenkalk*) are used. A ratio of 70% to 30% was assumed for calculation.
[b] LC = Limestone calcium carbonate. Applied once in three years.
[c] Four applications of solid, two of liquid fertiliser.
[d] min NP = mineral NP fertiliser.
[e] Fraction numbers reflect planting winter cover crops in three of four years and lime application once in three years, respectively Plo = ploughing; Cult = cultivator use; Sow = drilling; Lime = lime application; Fert = fertiliser application; Slur = slurry application; Pest = pesticide application.
[f] Both silage and grain maize are grown. A ratio of 80% to 20% was assumed and all figures shown are weighted averages of this ratio.

Data source: Standard cultivation methods from Ferdinand Pollert (2003; personal communication), official statistics (LK WL, 2002) and Iglo data base (for spinach and herbs). Yield and nutrient offtake data from LUFA Münster (personal communications) and Iglo data base (for spinach and herbs).

Table A.7 Land cover of the County of Borken and hemeroby factors of different land uses used to calculate the Naturalness Degradation Potential (after Brentrup et al., 2002; modified).

Land cover type	Fraction of County area (ha ha^{-1})	Hemeroby factor [a]	ETC/LC code [b]	Faction of area x Hem. factor (—)
Agricultural land	*0.69*			
arable	0.45	0.80	2.1.1	0.36
pasture	0.14	0.55	2.3.1	0.08
forest	0.09	0.35	3.1.3	0.03
farm yard and buildings	0.02	0.70	1.4.1	0.01
Semi-natural area [c]	0.17	0.25	3.1.1, 3.2.2	0.04
Built-up and urban area	0.09	0.85	1.1.2	0.07
Traffic infrastructure	0.05	0.90	1.2.2	0.05
Total	1.00			0.64

[a] Factor indicating the degree of 'Naturalness Degradation' of a particular land cover, 0.0 being entirely natural, 1.0 being entirely unnatural. After Brentrup et al., 2002.

[b] ETC/LC (European Topic Cente for Land Cover), as cited in Brentrup et al., 2002.

[c] Woodland, waterbodies, moor and heathland.

Data source: Land cover data from Kreis Borken, 2002, Hemeroby classes from Brentrup et al., 2002.

Table A.8 Land use data and the Pesticide Use Intensity Index in the County of Borken.

	Fraction of arable land (ha ha^{-1})	Pesticide Use Intensity Index (*Behandlungsindex*) Fungi-cides	Herbi-cides	Insecti-cides	Growth regulators	Sum	Fraction of area x Sum
Winter wheat	0.06	1.78	1.45	0.65	0.72	4.60	0.26
Winter barley	0.10	1.31	0.88	0.24	0.58	3.01	0.29
Winter rye	0.03	1.39	1.00		0.99	3.38	0.09
Triticale	0.07	0.85	0.97	0.20	1.21	3.23	0.23
Summer barley	0.08	0.80	0.40		1.00	2.20	0.18
Oat	0.01	0.33	0.78	1.06	0.62	2.79	0.04
Oil seed rape	0.003	0.69	1.00	0.98	0.15	2.82	0.01
Sugar beets	0.01		2.55			2.55	0.03
Potatoes	0.03	11.48	1.83	0.84		14.15	0.44
Maize	0.59		1.25			1.25	0.74
Field vegetables[a]	0.02					3.88 [a]	0.08
Total	1.00						2.37

[a] Assumed to be 70% spinach (Index = 2.5; own calculations) and 30% other field vegetables (Index = 7.1; Gutsche & Enzian, 2002). The figure given as Sum for vegetables is the weighted average of these numbers.

Data source: Pesticide Use Intensity Indices for the crops grown in the County of Borken from Gutsche & Enzian, 2002, land use data (reference year 1999) from LK WK, 2002.

Table A.9 Endowment of the municipalities within the spinach growing area with small (semi-) natural landscape elements.

Municipality	Arable land [a] (ha)	Percentage of small (semi-) natural landscape elements: Actual [a] (%)	Target [a] (%)	Fraction of municipality in growing area [b] (ha ha^{-1})	Municipality's share in total growing area (%)	Target x Share (%)	Actual x Share (%)
Borken	8581	18.3	8.6	1.0	27	2.3	5.0
Gescher	4959	21.6	8.0	0.5	8	0.6	1.7
Heiden	2793	18.7	9.5	1.0	9	0.8	1.7
Raesfeld	3762	17.0	7.3	1.0	12	0.9	2.0
Reken	3798	21.8	8.3	1.0	12	1.0	2.6
Rhede	4465	15.6	8.0	1.0	14	1.1	2.2
Südlohn	2879	16.8	8.0	0.5	5	0.4	0.8
Velen	4137	19.3	8.3	1.0	13	1.1	2.5
Total					100	8.3	18.5

[a] Data from BBA, 2002

[b] Assumed values.

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