Ecosystem Services of Natural and Semi-Natural Ecosystems and Ecologically Sound Land Use

Papers and Presentations of the Workshop

„Economic Valuation of Biological Diversity – Ecosystem Services“

International Academy for Nature Conservation, Vilm, 13 – 16 May 2007

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German Federal Agency for Nature Conservation

Attached to this publication you find a CD with a pdf-file of this paper with coloured photos and figures

Bundesamt für Naturschutz

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Ecosystem Services of Natural and Semi-Natural Ecosystems and Ecologically Sound Land Use - Introduction

Burkhard Schweppe-Kraft [1]

Natural and semi-natural ecosystems as well as ecologically sound forms of land use are an indispensable basis for the conservation of biological diversity. At the same time they provide us with important ecosystem services that would be lost if these ecosystems were destroyed or converted to intensive use.

At the workshop „Economic Valuation of Biological Diversity – Ecosystem Services" methods, case studies and results were presented pointing out the economic ond social importance and the economic value of such ecosystem services.

The presentations of the workshop dealt with

- the effect of natural floodplains on the decomposition and fixation of harmful substances (Dehnhardt) and its contribution to mitigate flood damages;
- carbon dioxide sequestration in natural and sustainably used peatlands;
- organic farming as a sink for greenhouse gas due to humus accumulation and as a method to prevent soil erosion;
- the role of hedgerows, wood patches and similar biotopes in agricultural used landscapes;
- the importance of natural and semi-natural ecosystems and ecologically sound land uses for recreation;
- the role of natural and semi-natural ecosystems for recreation, recreational fishing and hunting;
- the importance of urban green for life quality and property;
- the contribution of urban green to urban climate and air quality.

There are other important services for example the protection of fresh-water, the retention of avalanches and land slides or the provision of wild fruits and berrys and medicinal plants that have to be considered additionally, if the complete ecosystem service value of biodiversity is to be evaluated.

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One aim of the workshop was to discuss suitable methods, the appropriate databases and the necessary resources to carry out countrywide assessments of the value of these ecosystem services.

The main conclusions of the presentations and discussions can be summarized as follows:

- In some cases there is already more available knowledge about ecosystem services and their values for society than even some experts would assume (e.g. carbon sequestration). In other cases the information level is rather low and additional primary studies seem to be indispensable to perform valid assessments (e.g. economic value of landscape elements for recreation).
- Even in those cases where there is already a high level of information the knowledge and the required data to perform country-wide assessments are scattered in different fields of research, departments and levels of public administration, private or public associations as well as businesses (e.g. functions of low intensive land use for water supply).
- Ecosystem services and their economic benefits very often highly depend on the specific ecological and socio-economic conditions of the surrounding areas. Therefore simple methods of benefit transfer e.g. in terms of money/ha may fail due to the heterogeneity of landscape structures, differences in population density, accessibility etc.
- Anyhow a rough pre-assessment on the basis of simple methods could help to set priorities for more detailed surveys by defining those services that may be assumed to present the highest values.
- In every detailed assessment a compromise has to be made between survey costs and the accuracy of results. Whatever precision a method promises, the outcomes will not be a single number but rather a wider range of possible figures of different likelihood.

There was broad consent that an assessment of the ecosystem services of natural and semi-natural ecosystems and ecologically sound forms of land use would provide strong evidence that the conservation of biological diversity is not only in line with ethical obligations but also has economic advantages.

The Value of Floodplains as Nutrient Sinks: Two Applications of the Replacement Cost Approach

Alexandra Dehnhardt [1] & Ingo Bräuer [2]

1 Introduction

Ecosystems such as riparian wetlands provide a wide variety of environmental goods and services which create value for humans. As the Millennium Ecosystem Assessment (MEA) recently highlighted, ecosystem services and human wellbeing are closely linked (MILLENNIUM ECOSYSTEM ASSESSMENT 2005). According to the MEA these services can be classified in four categories: provisioning services, regulating services, cultural services and supporting services, a typology that points out the functional lines in which ecosystems contribute (directly or indirectly) to human welfare (PAGIOLA ET AL. 2004). In particular, the services provided by floodplains encompass provisioning services such as the production of fish and fodder; regulating services such as the maintenance of water quality (via nutrient retention) and flood control; cultural services such as opportunities for recreational activities; and supporting services such as nutrient cycling and ecosystem resilience (that refer to the 'primary value' of ecosystems, according to GREN ET AL. 1994). Many of the provisioning services, like agricultural, fishery or forest commodities ('goods'), are often more readily apparent: the services deriving from the complex physical, biological and chemical processes and functions are less perceptible, perhaps, but their importance and high value are well recognised.

Despite the growing awareness and appreciation of wetland values over the last few centuries, wetland ecosystems are still under tremendous pressure and are amongst the most threatened and destroyed ecosystems (MILLENNIUM ECOSYSTEM ASSESSMENT 2005), inter alia because their goods and services are often neglected in environmental decision-making. Economic valuation – as an attempt to assign monetary values to ecosystem services where no market prices are available – is supposed to give environmental assets more weight in decision-making by enabling policy-makers to consider the trade-offs between well-defined alternatives in terms of costs *and* benefits in comparable units, and thus set priorities based on efficiency criteria and evaluation of

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the social welfare implications of environmental policy options. Accordingly, economic valuation can help to reduce complexity by making functions and values more obvious and, moreover, contribute to a better allocation of budget resources by ensuring that money is spent according to public preferences.

Figure 1 illustrates the interlinkages between the structure and functioning of wetland ecosystems, the benefits (i.e. goods and services) derived by humans, and their subsequent values. The challenge in the course of an economic valuation is to transform ecological functions into economic values.

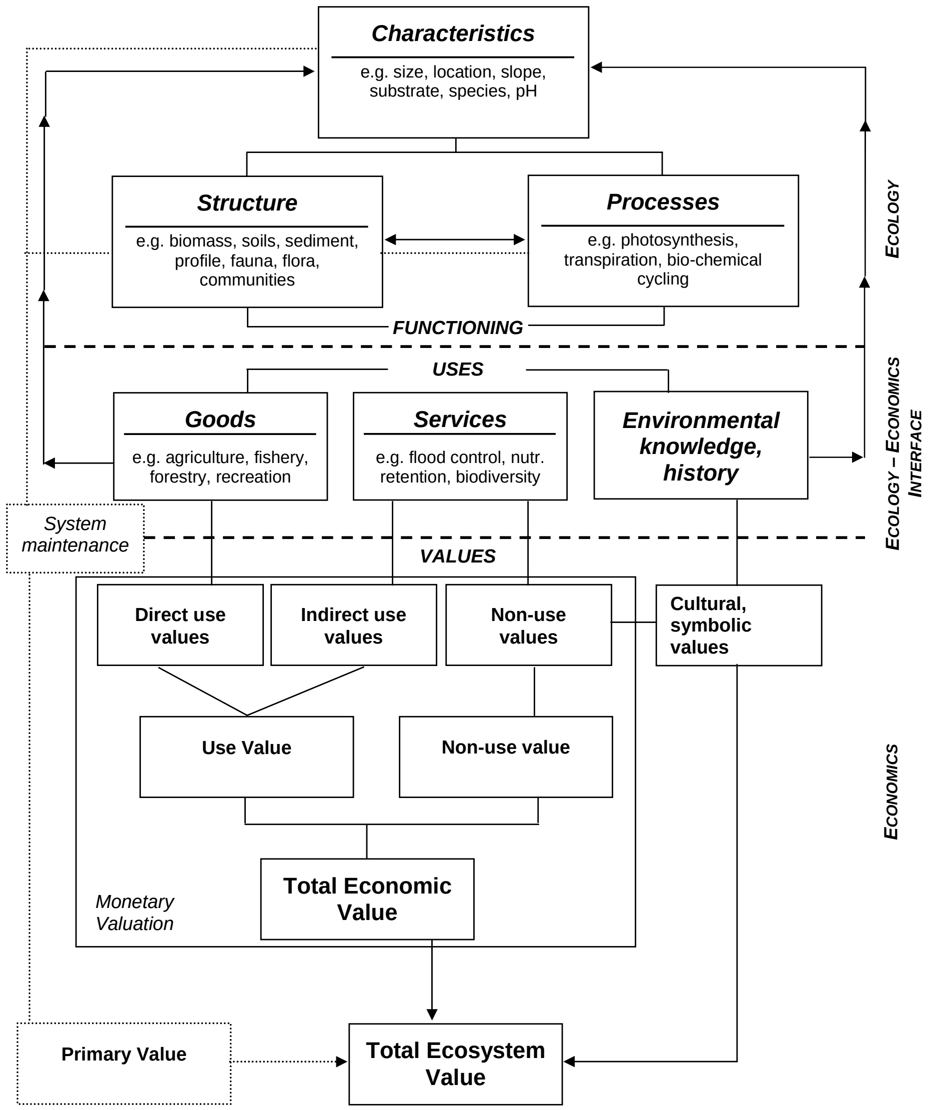

Figure 1: Economic valuation of ecosystem services (according to Turner et al. 2003)

Currently, in environmental policy and decision-making a growing interest and need to quantify and assess these values can be observed. The European Water Framework Directive (WFD) can serve as an example where (monetised) environmental benefits in river basin management are supposed to be considered by estimation of disproportionate costs according to Articles 4(4) and 4(5) (as a main decision criterion for justifying exemptions from the overarching aim of the Directive) on the basis of economic costs *and* benefits. The discussion about what kind of information is really needed, what an adequate decision criterion might be and how this information can be obtained to support decision-making is, however, still ongoing in European member states. In Germany, economic benefit estimation has not (or only insufficiently) been considered in environmental policy-making so far. One reason might be that economic valuation of environmental resources, and the values that have been elicited from contingent valuation studies in particular, suffers from low acceptance (BRÄUER & SUHR 2005). Thus, the crucial point of interest is which valuation approaches in which valuation contexts might be suitable and communicable with regard to a broader application in policy-making beyond certain case studies. The latter refer to the question of transferability of valuation results from one site to another, which is broadly discussed in the context of benefit transfer efforts.

This paper gives a short overview of different approaches to estimating the economic value of freshwater ecosystem services and presents the basic approach and the results from two case studies using the replacement cost method as an indirect valuation approach to estimate the value of the service 'improvement of water quality' provided by floodplains.

2 Measuring the economic value of floodplains ecosystem services

There is a variety of direct and indirect valuation approaches that are currently available for assessing aquatic ecosystem services. Basically, revealed preference and stated preference methods can be distinguished, which are both theoretically founded in the principles of welfare economics. While the key issue of revealed preference methods like travel cost approaches is whether ecosystem services affect people's behaviour, stated preference approaches (e.g. contingent valuation (CV), choice experiments) are survey-based methods that create hypothetical markets for people to state their preferences for the public good in question (for a good overview of different valuation approaches and their applicability see e.g. National Research Council 2005; FREEMAN 1993). In contrast, the replacement cost approach (RCA) is an indirect valuation method of estimating costs as a proxy for benefits. The fundamental idea is that if specific functions of an ecosystem can alternatively be achieved by a technical substitute, then the costs of this substitute to replace this function can be regarded as the economic value of the ecosystem's service (GREN et al. 1994; Byström 2000). Thus, since the RCA doesn't imply consumer's surplus as a preference-based welfare measure, it is critically debated and is supposed to be used with caution (NATIONAL RESEARCH COUNCIL 2005; BRANDER ET AL. 2006; PAGIOLA ET AL. 2004). This method is appropriate, however, for certain tasks (BRÄUER 2005, MEYERHOFF & DEHNHARDT 2007). This is especially the case for the investigated rivers where the WFD calls for action to reduce the nitrogen loads. Here it can be assumed that measures will be set up and hence real cost-savings can be realised.

Meanwhile, numerous valuation studies have been conducted in the context of water resource management and water quality improvement, which indicates a growing need to quantify the value of freshwater ecosystem services. The most extensive review and comprehensive summary of wetland valuation studies can be obtained from the existing meta-analyses of wetlands which each focus on different types of wetland values, including different valuation approaches. BROUWER ET AL. (1999) analysed the results of 30 CV studies of wetlands in temperate climate zones, defining wetlands in a broader sense. WOODWARD & WUI (2001) as well as BRANDER et al. (2006) include studies using a broad set of valuation methodologies and looking at a variety of wetland functions. The recently performed meta-analysis by Van HOUTVEN et al. (2007) uses only CV study results with a narrower focus, i.e. looking just at valuing water quality improvements. With respect to the different valuation methods that have been applied, the analysis from BRANDER et al. (2006) indicated that for the service 'improvement of water quality' the CV is most commonly used, but the RCA as well has largely been used to estimate wetlands' role in improving water quality (which refers to the nutrient retention function) (see also GREN 1995; BYSTRÖM 2000). Accordingly, in some cases of ecosystem functions with known functional ecological relationships RC approaches are particularly motivated and cost-savings are supposed to indicate the economic value of these services.

3 Case studies: The replacement value of the Jossa and Elbe floodplains

In both case studies presented here different valuation approaches have been applied in order to assess the economic value of the services affected by certain management actions. The overall objective of both studies was the conduction of a cost-benefit analysis to appraise certain management measures. In each case a CV study was conducted to determine the value that would arise from the ecological function 'biodiversity protection' (for details of the CV studies see MEYERHOFF 2004, BRÄUER 2002). In addition, the RCA has been applied to estimate the management effects as regards the improvement of water quality, which means the nutrient retention function. This paper focuses just on the valuation of the latter.

Table 1: Overview of the two case studies

Study area	Jossa & Sinn (Spessart / Hesse)	Elbe (German part)
	122 km² catchment area 2,3 km lengths	150,000 km² catchment area 1,100 km lengths
Management actions	Beaver reintroduction programme (implemented in 1987) 2001: about 200 individuals	Dike shifting: up to 15,000 ha new floodplains Extensification of the agricultural use (40,000 ha)
Effects	Influencing the structure and hydrology of the rivers by dam building activities	Improving hydromorphological conditions by floodplain restoration
Ecosystem services	• Enhanced 'biodiversity' • Species protection • Beaver watching • Nutrient retention	• Enhanced 'biodiversity' • Flood protection • Recreation • Nutrient retention

Table 1 summarises the main characteristics of the two case studies. While the study sites and the management programmes considered are rather different, the resulting effects are quite comparable. The measures would improve (or have improved, in the case of the Jossa) the hydromorphological status of the river systems, changes that are associated with effects on biodiversity and the nutrient retention.

3.1 Methodological Approach

The effects on nutrient retention in both case studies have been assessed by use of the RCA, which requires the following three steps in this context:

1. Identification and quantification of the retention effects (in both cases just the effects on *nitrogen* retention were considered),
2. Definition of the reference scenario (i.e. the substitute and its marginal costs) and
3. Economic valuation.

Many studies have shown that the self-purification potential of a river depends on its structure and its surrounding buffer strips (KRONVANG ET AL. 1999). Basically, owing to the restoration of floodplains and river banks, the stream velocity is affected and thus the denitrification rate increases as a result of a higher residence time of the water (BEHRENDT & OPITZ 2000). The nitrogen retention was therefore not measured directly, but parameters which are known to influence the denitrification were measured instead. Since the biochemical processes are generally comparable in both cases, the effects on the river system differ slightly in the case of a small system like the Jossa and a larger system like the Elbe.

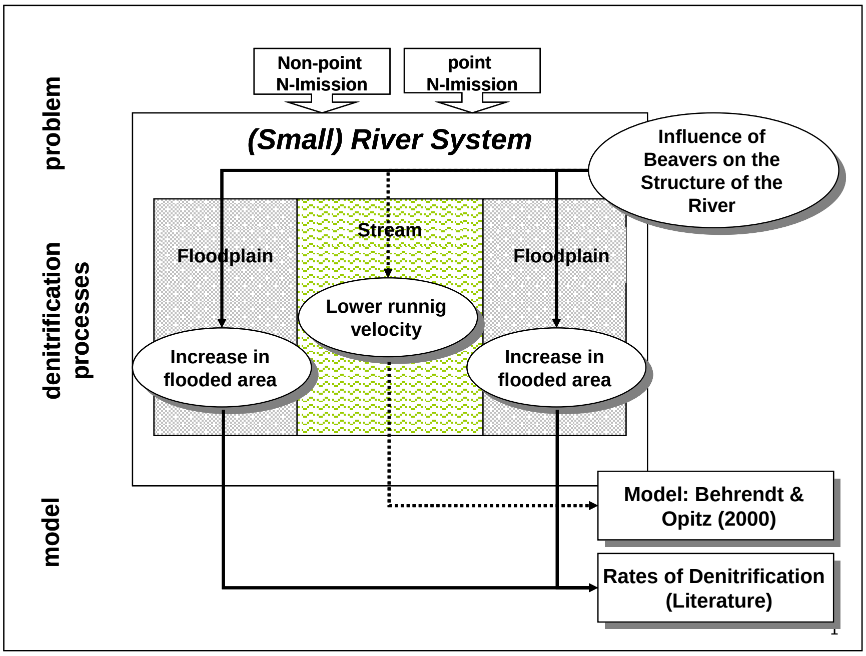

Figure 2: The hydrological model – Influence of the beaver reintroduction scheme on the river system

The influence of beavers and their dam-building activities are twofold; they affect the river by decreasing the running velocity of water as well as increasing the flooded area. Both effects have been estimated separately (see Figure 2). To quantify the changed nitrogen retention potential within the river, a statistical model from BEHRENDT & OPITZ (2000) was used. It allows calculation of the N-retention by measuring the hydraulic load (m/a) of the river which is defined as the mean discharge in relation to one square meter water surface of the river system and thus indicates the hydromorphological structure of the river: the higher the hydraulic load, the higher the running velocity (for a detailed description see DEHNHARDT & MEYERHOFF 2002, BRÄUER 2002). To estimate the denitrification in the additional flooded areas denitrification rates from the literature were used.

As in the case of the Jossa, the stream velocity is affected by the enlargement of floodplains (owing to dike relocation measures – as indicated by the red arrows in Figure 3) along the river Elbe. Owing, however, to the wide stream, the running velocity is mainly decreased in the floodplains and the edge of the river and not in the main stream itself. For estimating the effects the *effective* flooded area is decisive, and it is particularly determined by the inundation dynamic of the river (i.e. the duration and frequency of flooding) and the morphology of the floodplain surface. Thus, the effects vary substantially depending on site-specific conditions, and in an initial step the nitrogen reduction effect was quantified for two specific relocation sites (Sandau and Rogätz) where site-specific data were available. For these specific areas the same statistical model from BEHRENDT & OPITZ 2000 was used, and or the remaining areas (up to 15,000 ha) data from the literature (Figure 3).

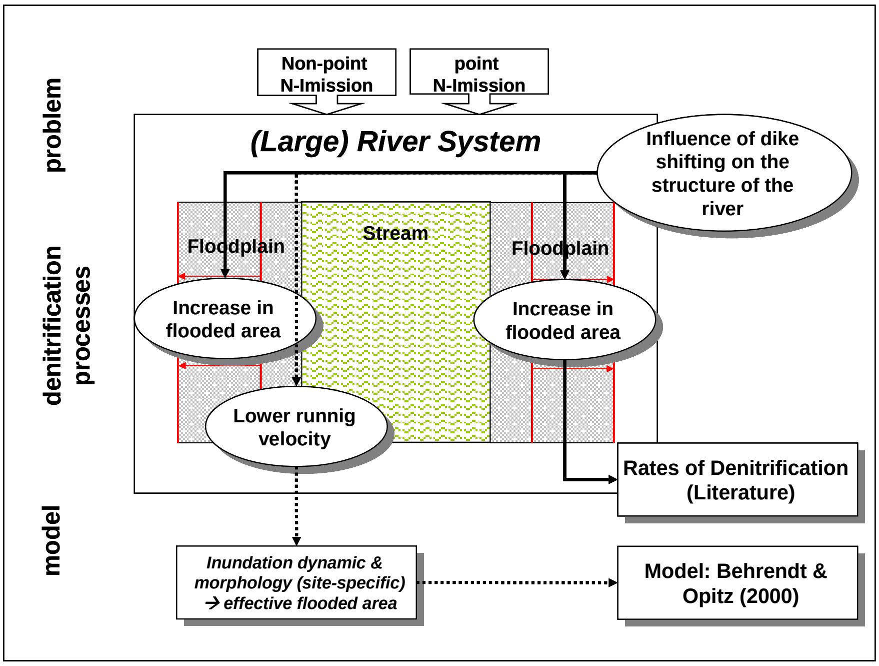

Figure 3: The hydrological model – Influence of the enlargement of floodplains by dike relocation

3.2 Results

According to the quantification of the 'ecosystem function' (the nitrogen retention) the decreased hydraulic load (-15%) and the additional flooded area (+17%) underline the fact that the beaver territories along the Jossa have a significant influence on the morphology of the river. These changes in the river structure result in a total additional retention in the Jossa and the surrounding floodplains up to 1,200 kg N/year. For the investigated area in total (16 beaver territories) the overall effect adds up to ca. 4700 kg N/year.

In the case of the Elbe, the results have shown that the *effective* retention area is comparatively small: for a mean annual discharge 88% (Sandau) respectively 20% (Rogätz) of the total area at the two relocation sites are actually inundated; the total available surface area as a result of dike relocation would only be inundated with an HQ_{100}. Accordingly, the total nitrogen retention adds up to 650 t/year for Sandau and 40 t/year for Rogätz with less appropriate site-conditions. Because of this evident high variability it is difficult to scale up these effects for the investigated area along the Elbe in total. For up-scaling the nitrogen retention effect a conservative estimate of 200 kg is used, based on our findings and further information available in the literature, which is around the lower mean of 200 kg N/ha (MITSCH & GOSSELINK 2000, KRONVANG ET AL. 1999).

Following the RCA, the valuation of the assessed ecosystem function requires as the next step the identification of the potential technical substitute and its marginal costs to provide an equivalent service. According to BOCKSTAEL ET AL. 2000 the technical substitute must provide functions that are equivalent in quality and quantity to the natural function, and the substitute must be the least cost alternative of performing this function. Even though possible substitutes for the service 'improvement of water quality' refer to different sources of nutrient load and to different functional processes in detail, they cope with the main problem. The substitute most commonly used to value the self-purification potential of rivers is sewage treatment plants, where the function process of denitrification is very similar (BYSTRÖM 2000, GREN 1995). Otherwise, political strategies to reduce the nitrogen load such as agricultural measures, which are the most cost-effective alternative, are considered. The assumed marginal costs are 7,7 €/kg N for sewage treatment plants and 2,5 €/kg N for agricultural measures (see BRÄUER 2005, MEYERHOFF & DEHNHARDT 2007).

Finally, the monetary value was estimated by assessing the total cost of the substitute to provide the same service. Table 2 summarises the results for the two case studies. For the Spessart case the estimated value obtained for the Jossa was scaled up. For these replacement values to be included within the cost-benefit analysis they have to be considered for the whole project duration to make them comparable to the costs of implementing the management actions (the time period was fixed at 20 years and the discount rate at 3%). The value arising from the nutrient retention is in both cases rather small compared with the results from the CV study, which encompasses the maintenance of 'biodiversity'. In comparison with the costs, however, the economic benefits of the ecosystem service 'improvement of water quality' alone make up 12% (Spessart) respectively 11% (Elbe) of the total investment costs (see Table 2).

Table 2: Value of the nitrogen retention at the Jossa and the Elbe

Study area	Spessart	Elbe
		Sandau: 1,700,000 €/year Rogätz: 99,000 €/year Other sites: 6,900,000 €/year
Value	approx. 12,000 €/year	approx. 8,700,000 €/year
Value for the project duration	approx. 250,000 €	approx. 54 mill. €
Proportion of total costs	12 %	11 %

4 Discussion

The case studies presented and their results can be discussed from several points of view. The results of both studies show that the non-market benefits of riparian wetlands can prove substantial; and furthermore that one ecosystem function alone constitutes a significant part of the total management costs. Together with the results of the CV studies the benefits of restoring the floodplains would easily pass a cost-benefit test in both cases and the results are stable, as the sensitivity analyses show.

Apart from these findings the suitability of the RCA and the applicability to guide decision-making must be the objects of critical discussion. The method is mainly criticised from a welfare economist perspective. Since a 'value' arises from human preferences, i.e. what individuals perceive an ecosystem service contributes to his / her own wellbeing (Bockstael et al. 2000), it is still debated whether the application of the RCA results in valid estimates of economic values (Sundberg 2004). Bockstael et al. 2000 emphasise a correct application of the concept of economic value when estimating monetary values in the course of cost-benefit analysis and policy-making.

In spite of this (qualified) criticism and the weaknesses of the RCA, however, the applicability of this method in political-economic processes is to consider. If the valuation approaches that are in compliance with economic theory are not accepted, and therefore not included in decision-making, alternative approaches may come into question. Given this, one advantage of the RCA is its transparency; furthermore, it is a user-friendly approach using comparable, easily-available data. It is particularly applicable to ecosystem functions with which the general public is not familiar and that are (owing to the complexity) difficult to capture through surveys – where a comprehensive description of the service in question is required (Nunes & van den Bergh 2001).

Estimating the monetary value of the nutrient retention function of riparian wetlands using the RCA has, however, revealed some flaws that have to be regarded when we interpret the results and might restrict transferability for policy-making purposes. The results from the case studies show a high variability of the effects; determining the cause-and-effect relations has proved to be difficult, is characterised by uncertainty and has a tremendous influence on the results. But, if the relationships between the ecological and the economic system are well known, the ecosystem function can be clearly defined and an appropriate substitute identified, then the cost-saving can indicate the economic value of the service regarded.

All in all, the method can serve as a proxy valuation, if some conditions are met. Besides the two requirements mentioned above (perfect substitute and cost-effectiveness), there should also be substantial evidence that the service would be demanded by society (BOCKSTAEL ET AL. 2000, FREEMAN 1993, NATIONAL RESEARCH COUNCIL 2005). As BIROL ET AL. 2006: 111) stated, "[t]his method is particularly applicable where there is a standard that must be met, such as a certain level of water quality." With regard to water resource management, and the implementation of the WFD that requires achievement of a good ecological status, a societal demand for water quality improvements might be derived and, the RCA might be an appropriate approach. Apart from all the criticism regarding compliance with the economic theory the values are a very good first proxy that at least could be used to illustrate the value of ecosystem services in an understandable manner. Their further use in CBA has to be evaluated critically on a case by case basis.

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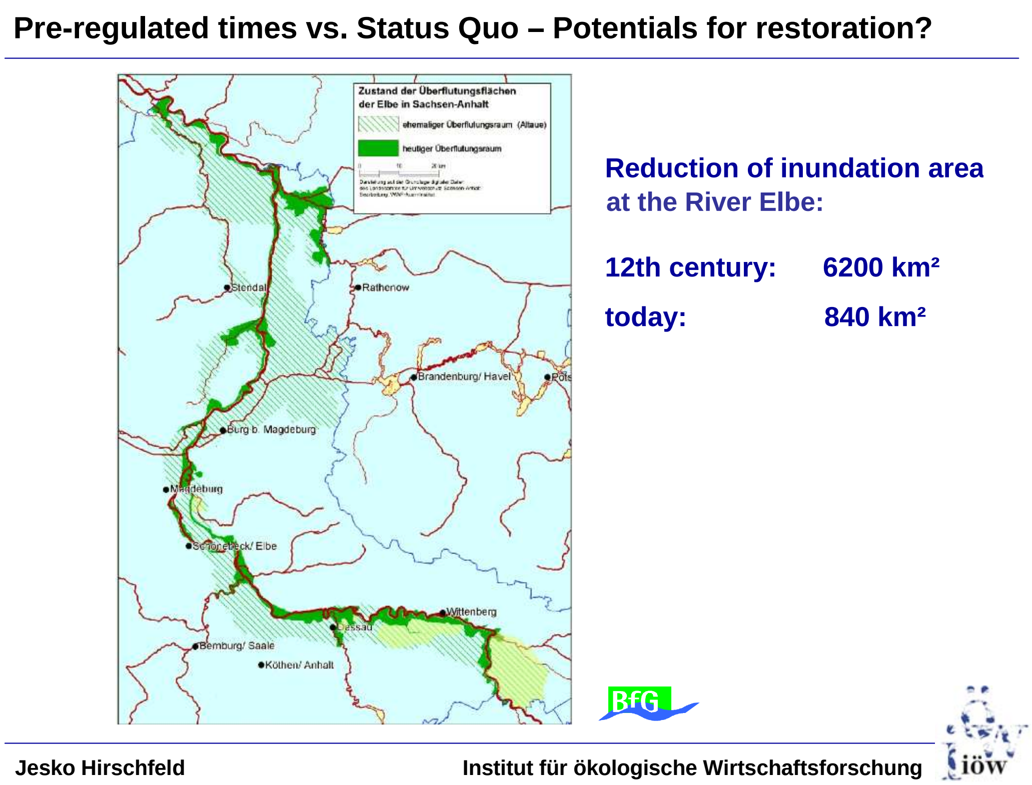
Ecosystem Services of Alluvial Floodplains in an Ecologically Extended Cost-Benefit Analysis: Flood Protection

Jesko Hirschfeld [1], Alexandra Dehnhardt [2] & Daniel Drünkler [3]

Presentation
selected slides



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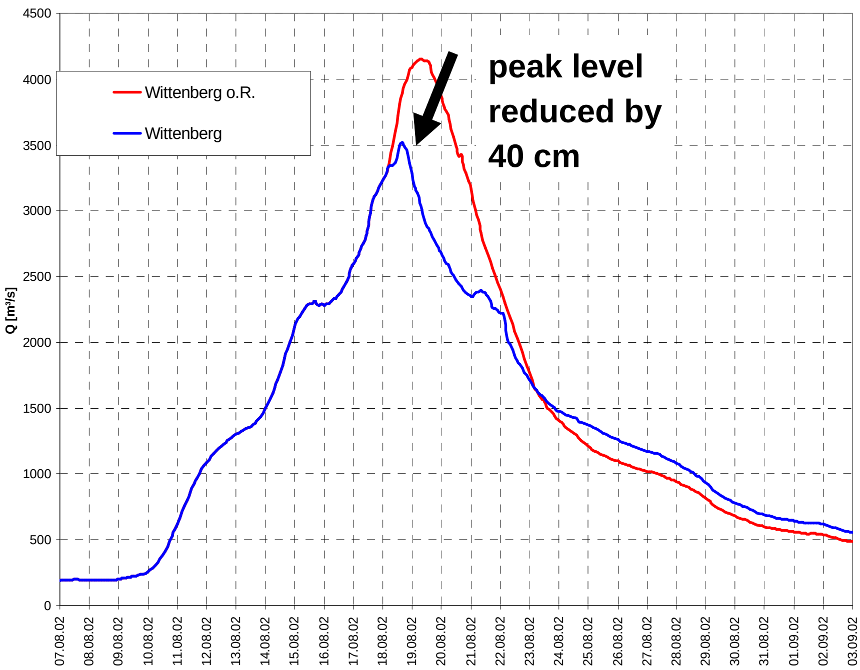
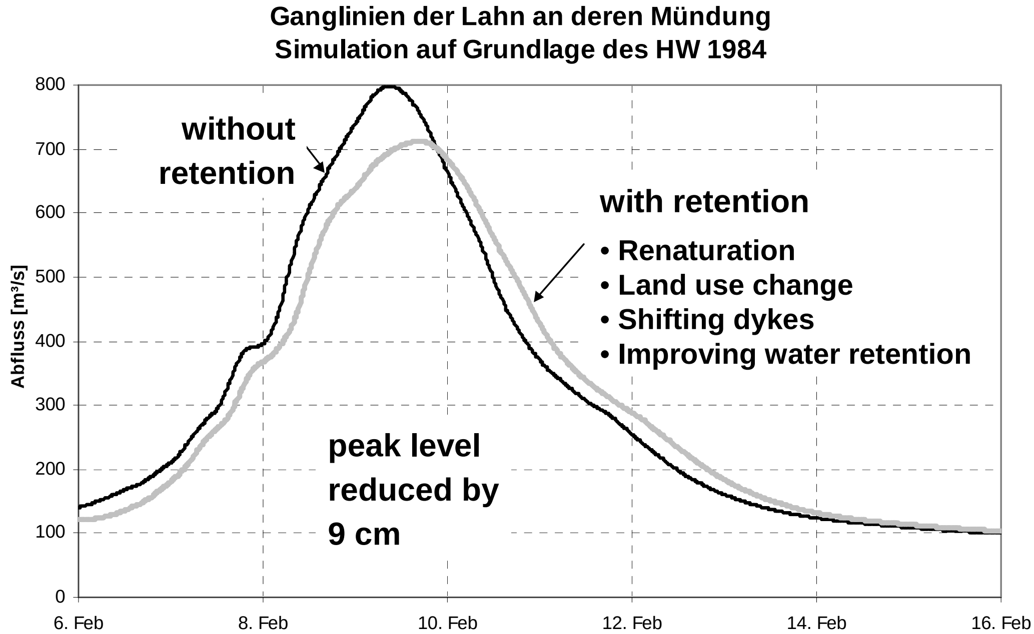
Floodplain restoration effects: Ecosystem Services

Flood peak level reduction	**Damage reduction**
Slowing down the wave	**Damage reduction**
Flood period duration increased	**Mixed effect on damages**
Biodiversity effects	**Existence value**
Intensive landuses reduced	**Opportunity costs**
Nutrient retention	**Replacement value**
Recreational value	**Use value**
	ECOSYSTEM VALUES

Floodplain services: The case of dyke breaches (Elbe flood 2002)



Floodplain services: Restoring the natural rentention function



Effects of flood risk management measures on peak and duration

Dependend on spatial scale level, character and dimension of flood event

Spatial scale level	Regional effects				Inter-regional effects			
Dimension of flood event	small		large		small		Large	
	duration	peak	duration	peak	duration	peak	duration	peak
Afforestration, succession								
Ecological farming								
Removal of sealing								
Restoration of floodplains								
Shifting dikes backwards								
Technical retention systems								
Building additional dikes								
Enlargement of profiles								
Reducing damage potential								

Positive effect	
Uncritical negative effect	
Critical negative effect	

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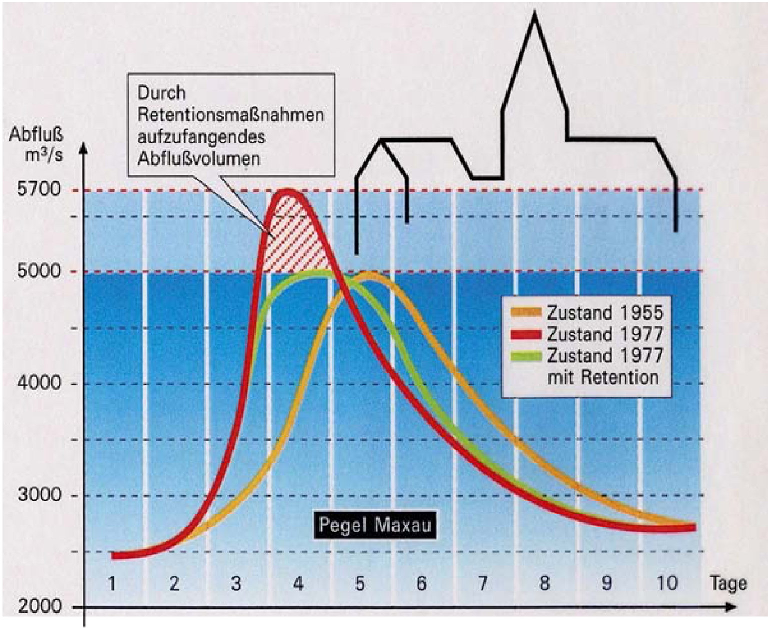
Costs of flood risk management options

Direct costs	All costs that are directly associated with the flood control measure (e.g.): • construction costs and land purchase • planning costs • costs for monitoring and forecasting systems • costs for information programmes to enhance risk awareness • operating and maintenance costs
Indirect costs	Costs that occur as a result of flood control measures (e.g.): • opportunity costs (foregone use options: agriculture and forestry, urbanisation and industry, inland navigation) • damages as a result a lower groundwater level • loss of land value (flood plain areas) • transaction costs (interregional co-operation, negotiations,...)

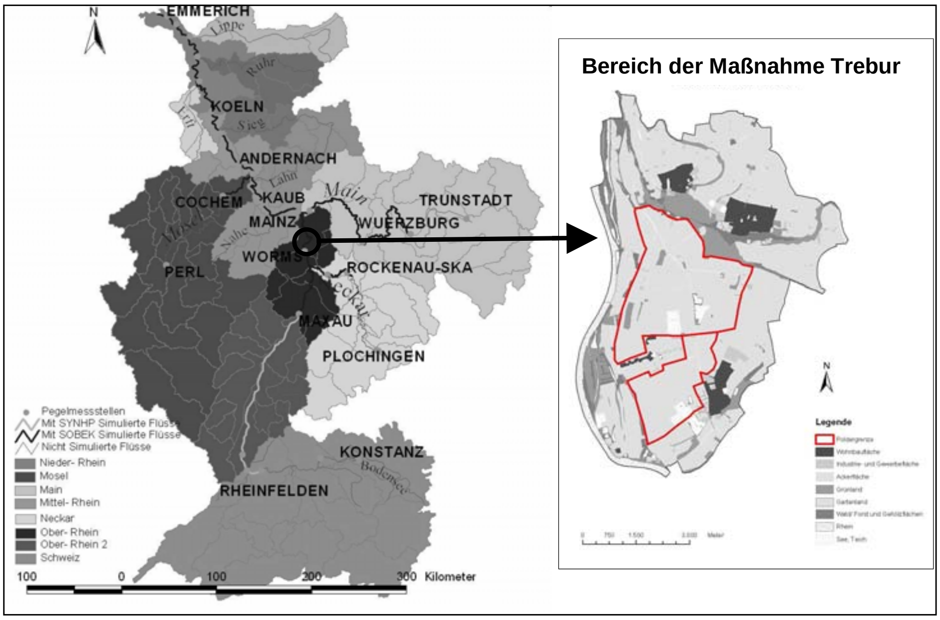
Benefits of flood risk management options

Direct benefits	• damages avoided (buildings, agricultural land and infrastructure) • avoided losses of added value avoided intangible damages (health, environment) • avoided downstream damages (as a result of upstream measures)
Indirect benefits	External benefits as a result of flood control measures (e.g.): • increase / conservation of biodiversity • improvement of recreation opportunities • reduced pressure for downstream users

Regaining retentention volumes at the river Rhine



Extended Cost-Benefit Analysis: Polder Trebur, Rhine



Extended Cost-Benefit Analysis: Polder Trebur, Rhine

	Maximum Peak Level Redution [cm]			
	Region Trebur		Region Cologne	
Modelled-Floods	HW88	HW95	HW88	HW95
Köln HQ100 (without/with Polder Trebur)	15	4	15	3
Köln HQ200 (without/with Polder Trebur)	15	9	15	5
Worms HQ100 (without/with Polder Trebur)	13	10	14	7
Worms HQ200 (without/with Polder Trebur)	18	19	18	13

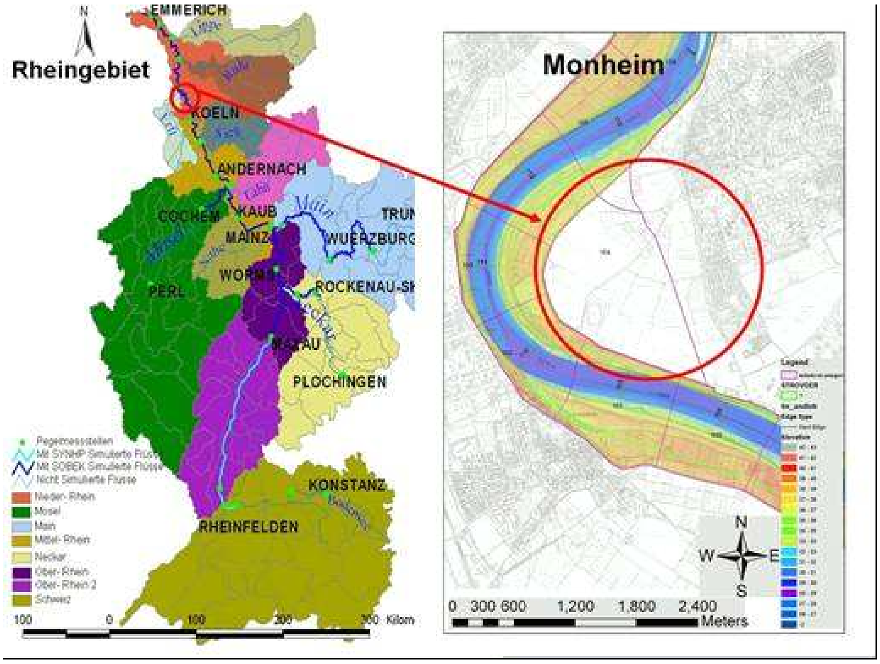
Costs (regional) **116 Mio. €**

Present value of damage reduction (regional) **20 Mio. €**

Present value of damage reduction (downstream) **146 Mio. €**

Benefit-cost ratios: regional 0.2 : 1 vs. interregional: 1.3 : 1

Extended Cost-Benefit Analysis: Dyke shifting (Monheim, Rhine)



Extended Cost-Benefit Analysis: Dyke shifting (Monheim, Rhine)

Monetarised effects	Enforcement of original dike €	Shift of dike €
Avoided damages Monheim (present value)	2.000.000	2.000.000
Avoided losses of added value Monheim (present value)	350.000	350.000
Avoided damages Cologne (present value)	0	3.700.000
Avoided losses of added value Cologne (present value)	0	673.000
Recreational value	0	18.900.000
Biodiversity	0	50.300.000
Nutrient retention	0	0
Present value of project benefits	**2.350.000**	**75.923.000**
Present value of project costs	**10.000.000**	**38.800.000**
Benefit-Cost Ratio	**0,24**	**1,96**

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Water Retention in the Rhine Basin (IKSR action plan)

(1) Water retention in the Rhine catchment area

- Renaturation (11.000 km)
- Restoration of floodplains (1.000 km²)
- Extensification of agriculture (3.900 km²)
- Afforestation, succession (3.500 km²)
- Removal of surface sealing (2.500 km²)
- technical retention systems (73 Mio. m³)

Reduction of flood peak levels until 2020

~ 10 cm

Costs: ca. 8.400 Mio.€ respectively **840 Mio €/cm reduction**

(2) Water retention along the Rhine river

- Restoration of floodplains (160 km²)
- technical retention systems (364 Mio. m³)

Reduction

~ 60 cm

Costs: ca. 2.400 Mio.€ respectively **40 Mio €/cm reduction**

BfG

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Valuation of Ecosystem Services: Extended Cost-Benefit Analysis

Floodplain restoration for flood protection purposes?

Ratio of direct costs and direct benefits often not really convincing

„Conventional“ cost-benefit analysis usually favours technical solutions

Only if additional ecosystem services are taken into account,

(in the course of an ecologically extended cost-benefit analysis)

floodplain restoration becomes favourable

Most efficient strategy:

Minimise damage potentials by conserving natural floodplains!

Acknowledgements / Sources

The presentation draws on data generated for the project:

Cost-Benefit-Analysis of Flood Risk Management Measures
Research Project PKZ 204 21 212 financed by
Umweltbundesamt
(German Federal Environment Agency)

Institute for Ecological Economy Research (IÖW)
(project leader)
www.ioew.de

German Federal Institute of Hydrology in Koblenz (BfG)
(cooperation partner)
www.bafg.de

Eco-system Valuation for Real Habitat Creation through Managed Realignment of the English Coastline

Ino Kremezi [1]

1 Introduction

Traditional flood risk management strategies in England have consisted of 'hard' sea walls and the drainage of wetlands for agricultural use. Many of these old sea walls are now collapsing, reaching the end of their design lives and maintaining and upgrading them can be very costly. In the light of rising sea levels and increases in extreme weather conditions due to climate change the cost-effectiveness of traditional flood risk management is questioned. It is now thought that a multi-functional approach to flood risk management, such as managed realignment may be more sustainable from an environmental, and an economic point of view.

At the moment, the allocation of capital for flood risk management projects in England and Wales is decided on the outcome of a cost-benefit analysis that compares different options. The current policy guidelines in England state that the environmental benefits of flood and coastal erosion management schemes should be included in the economic appraisal of these schemes. This is rarely done in practice however, because of the difficulties in quantifying environmental benefits from habitat creation. This article explores potential significance of the economic valuation of benefits from habitat creation and their effect on option choice.

Responsibility for flood and coastal management in England and Wales is currently shared by a number of organisations and agencies, including DEFRA[2], the Environment Agency, Local and Regional Flood Defence Committees, coastal district councils and individual land owners. The Environment Agency invests significant resources in the maintenance and construction of coastal defences and other forms of coastal erosion risk management. The aggregate budget for such projects is approximately £600 million per year[3]. The Shoreline Management Plan guidance[4], defines four strategic policies for coastal erosion and flood risk management. These are: **1. Hold the line** -

[1] Senior Economist, Environment Agency, ino.kremezi@environment-agency.gov.uk

[2] Department for the Environment, Food and Rural Affairs

[3] Following recent flooding events (summer 2007) the government has agreed to increase the budget for capital investment in flood and coastal erosion management projects by £200m to around £800m per year

[4] Published by DEFRA in 2006

Maintaining or changing the standard of protection, **2. Advance the line** - Building new defences on the seaward side of the original, **3. Managed realignment** - Allowing the shoreline to move backwards or forwards, with management to control or limit movement and **4. No active intervention** - No investment in coastal defences or operations.

So far, a form of cost-benefit analysis (CBA) applied on a project by project basis has justified the expenditure of public funds on flood risk management schemes. It is becoming increasingly apparent that there is a strong case for switching to a mixed, "multi-functional" approach, with protection focused on strategic and high value areas and the rest of the coastline left to adapt to change more naturally. This need for change in policy is expressed in DEFRA's new approach to flood defences in Making Space for Water in March 2006, where the need for integrated appraisal in line with the principles of sustainable development is highlighted.

2 Economic Valuation of Environmental Impacts from FCERM[5] schemes

Despite the growth in formal requirements to value environmental impacts in monetary terms, in most cases cost-benefit calculations do not include the economic value of environmental benefits and costs. Consequently, schemes encouraging the creation of environmental improvements have tended to be passed over in favour of more traditional options, and where decision-making has favoured the environmental option, the rationale has been to conserve the status quo as required by statutory protection, rather than the creation of new habitats. Moreover, where abandonment of a flood and/or coastal defence is clearly the least cost solution it has been difficult to make the case for a managed process, which re-creates desirable habitats with a degree of certainty.

The most commonly used arguments for the exclusion of environmental benefits from the economic appraisal are (1) that it rarely makes a difference to the actual decision and option choice and (2) that the cost of acquiring the data is greater than the loss from a sub-optimal choice. The main reason for this is the lack of reliable information quantifying and valuing the environmental effects of different schemes. Collecting this information on a project by project basis (through individual valuation studies) is too expensive and time consuming. As a result, any additional benefits from the environmental valuation are outweighed by the additional costs incurred during the appraisal process. This could be overcome if there was a commonly agreed method of taking re-

[5] FCERM: Flood and Coastal and Erosion Management

sults from valuation undertaken in one circumstance and applying it to other circumstances, a method known as "benefits transfer".

3 Economic & Environmental Impacts of Managed Realignment Schemes

The main economic ***cost*** of managed realignment schemes is the opportunity cost of the land that was previously protected by the existing defences. If this is agricultural land the cost can be measured in terms of future agricultural production that will be lost. Other costs include the cost of the works needed depending on whether there is a need for new defences to be set up inland or not, costs of breaching and maintenance costs. Costs of habitat creation may also be included.

The main ***benefit*** from managed realignment schemes is mainly flood defence with the newly created inter-tidal zone acting as a natural sea defence. However, the re-creation of inter-tidal habitat has also a number of other benefits both economic and environmental. The extent of these benefits will depend on the amount and the type of the habitat that will be created and the speed of habitat recreation. These benefits include: commercial and recreational fisheries, carbon sequestration, nutrient and contaminant recycling, heavy metal storage, benefits to biodiversity through improved water quality and vegetation, as well as recreational and tourism benefits.

4 Economic Valuation

There are a number of techniques available to environmental economists to estimate economic value of non-marketed goods and services in monetary terms. These methods can be divided in three main categories: (i) market-based methods (ii) revealed preference techniques and (iii) stated preference techniques. It is not always easy to initiate a new study in a project area to determine people's preferences regarding some environmental change. However, if a similar project has previously been undertaken elsewhere, estimates of its economic consequences might be usable as an indicator of the impacts of the new project. This approach is called 'benefits transfer'.

Benefits transfer is a technique that applies the results of previous environmental valuation studies to new policy or decision-making contexts. There are several approaches to benefits transfer, which differ in the degree of complexity, the data requirements and the reliability of the results, but in principle they all use either average WTP values or WTP functions. The most important reason for using previous research results in new policy contexts is that it saves a lot of time and money. Applying previous research findings to similar decision situations is a very attractive alternative to expensive and time consuming original research to inform decision-making.

There are some concerns about the benefits transfer method. The first concern is whether the replication of findings is reliable. For example, there are questions concerning the extent to which the method is able to produce the same outcomes at different sites across different groups of people at different points in time. The other main concern is of course whether any inaccuracies resulting from the use of inappropriate analogues are greater than the ones resulting from the non-valuation of eco-system value.

5 The issues

The monetary valuation of environmental benefits from managed realignment schemes is possible and can have a significant influence on the choice of option. However, there are a number of issues that have to be taken into account to enable a proper appraisal of environmental gains/losses. The most important of those issues are:

5.1 The lack of original data

One of the main problems encountered when trying to value the environmental benefits of managed realignment projects is the lack of original research. Over the past years, a number of studies have looked at the economic values associated with the environmental benefits provided by wetlands and riverine floodplains. The findings and implications of these studies have been drawn together in a number of meta-analysis studies and have been used in economic appraisals of managed realignment projects through benefits transfer. However, most of these studies refer to non-UK sites, most of them originate from the US, others from other European countries. This can have important implications, since there are issues of different geographical and climatic conditions, as well as different mentalities expressed in the people's preferences. The ComCoast project (see Section 6.2) will provide original data for the UK set in Essex, but there is a need for much more applied research if the environmental benefits are to be included in the appraisal processes.

5.2 Valuation on an individual site level versus estuary level

When benefit-cost analysis is undertaken for a flood compartment ("cell") or length of coast, the estimated benefits might not necessarily reflect the overall economic benefits felt elsewhere. A broad scale, holistic economic appraisal is therefore better. Estuarine processes are complex and any changes at one location will result in changes elsewhere the estuary. For instance, implementing a managed realignment scheme to turn a flood cell into wash land might not be economically justifiable if it is only the benefits arising from the flood cell itself that are taken into consideration. However, the scheme could contribute to the reduction of hydrodynamic stresses elsewhere in the estuary by reducing extreme tide levels and result in habitat creation to compensate the estuary-wide habitat loss due to coastal squeeze. Although it is recognised that broad scale economic appraisal represents a better way of evaluating the economic benefits of flood and coastal defence projects, there are constraints in the broad scale economic appraisal process. Broad scale benefit-cost analysis can be complex since the study areas are often large and a good understanding of the interaction of the complex estuary or coastal processes is required.

5.3 Environmental quality variable

One very important function of the inter-tidal and saltmarsh zones created through managed realignment is their potential for sediment storage and carbon and nutrient cycling (ANDREWS ET AL., 2006). The understanding of geochemical/geological functions of inter-tidal habitats is of global relevance. In coastal seas in general, and more inshore waters in particular, wetland nutrient cycling has been estimated to represent their most valuable environmental service (COSTANZA ET AL., 1997). Similarly, wetlands represent the largest component of the global terrestrial organic carbon inventory, with tidal saline wetlands storing in excess to 45 Tg C a^{-1} (CHMURA ET AL., 2003) carbon burial in saline wetlands is thus potentially an important sink for atmospheric CO2. The evaluation of managed realignment at the estuary level requires an understanding of the sedimentological, hydrological, ecological and biogeochemical systems together with the socio-economic goods and services provided. ANDREWS ET AL. (2006) present a plausible, albeit extreme, managed realignment scenario for the UK Humber estuary, to demonstrate the maximum possible biogeochemical effects and economic outcomes of estuarine management decisions. Their ‘Extended Deep Green Scenario’ for the Humber identifies the maximum possible area that could be returned to the inter-tidal zone over the next few decades to be 7,494 ha and is compared to a ‘Hold the Line’ scenario where no habitat is created. Cost-Benefit Analysis was used to determine the economic efficiency of managed realignment. The results of the CBA showed that although the managed realignment schemes are not cost-effective over a 25-year period, over a 50-year and a 100-year period it is more cost-effective than the Hold the Line option. The CBA does not include the valuation of heavy metals stored in the habitat, but there is a potentially significant economic benefit from heavy metal storage that should be included in the valuation. This paper highlights the potential significant economic effects that the inclusion of environmental benefits in option appraisal can have and the possible effect on option choice.

5.4 Environmental quality as perceived by the local community

Economic valuation of environmental assets is usually achieved by stated preference methods. Stated preference methods require survey respondents to make judgements on the environmental good they are asked to value (Willingness to Pay). In order for this method to work, individuals have to make informed, rational decisions. However, it has been shown that many times people do not have sufficient knowledge of the good they are asked to value. This can lead to misleading results. In the case of benefits provided by habitat recreation this issue is even more relevant. Inter-tidal habitats, as mentioned before, provide a wide range of services including flood protection, water quality regulation etc. Many of these services are complex and it is unlikely that people will be aware or have a good understanding of them. Research has shown that people tend to value assets in their local area higher than assets elsewhere and the regional variations can be significant (CHRISTIE ET AL., 2004). This is because people value the fact that they can personally enjoy (see) a specific species or landscape locally and also because they are most likely to be aware of them and to be affected by a change.

6 Progress

6.1 Economic Valuation of Environmental Effects (Handbook)

The Environment Agency has recently commissioned a study by EFTEC to establish a framework for estimating environmental costs and benefits of FCERM[6] measures in monetary terms. The principle output of this study is a short, concise Handbook 'Introducing the Environment into Flood Risk Decision Making'. The handbook guidance is aimed at appraisal practitioners in the Environment Agency. It augments current appraisal guidance – from the HM Treasury Green Book and FCDPAG series – and further develops the approach to assessing environmental costs and benefits outlined in the recent Flood Hazard Research Centre 'Multi-coloured Manual'.

This Handbook describes a process for estimating the value of environmental gains and losses associated with FCERM schemes. In order to aid practical applications, the Handbook develops a transparent approach to collate evidence of environmental im-

[6] Flood and Coastal Erosion Management

pacts in qualitative, quantitative and monetary terms, and to report on associated uncertainties and limitations. The Handbook is accompanied by case studies that demonstrate the application of the methodology, a review of economic value evidence, a spreadsheet template and a technical report for the interested reader.

The decision-making context, legal requirements, scheme characteristics, location, habitats affected, uses of the environment and so on will determine the 'accuracy' that is needed from economic valuation evidence. In order to cater for both the simpler and the more complex ends of the scale of accuracy requirements, this guidance presents two levels of analysis:

I. *The **"first cut – a quick look at the economic value evidence"***: this provides a series of default values for use in option development. Here the intention is to provide an indication of the magnitude of economic value evidence related to typical environmental effects associated with FCERM scheme options. This is particularly appropriate for preliminary assessments of an initial 'long list' of FCERM options ensuring especially that an explicit account is made of the environmental costs and benefits. Depending on the requirements of the decision-making context, the evidence generated by the first cut may or may not be sufficient.

II. *The **"second cut – benefits transfer":*** is a full scale benefits transfer process in the specific context of FCERM schemes with the express intention of inputting to CBA. This level of analysis requires more information and practitioner effort than the first. The level of effort should of course be appropriate to the needs of the overall decision-making context.

It is important to stress that this guidance is the start of a continuing process. It is our intention that this work will be periodically revised and updated as new information becomes available from initiatives such as ComCoast (see below) and other original valuation work

6.2 The ComCoast project

ComCoast (COMbined function in COASTal defence zones) is an international project that aims to promote and implement an integrated approach to improving coastal defence systems. It is jointly funded by the Interreg IIIB Community Initiative Programme (North Sea Region) and the project partners, which are: Holland (lead partner), the United Kingdom, Germany, Belgium and Denmark. There are several themes within the ComCoast project, which are divided into 6 work packages. Work Package 2 - Socio-Economic Evaluation is led by the UK and comprises three doctoral assignments that aim to demonstrate the value of new managed realignment schemes in terms of the wider coastal or estuary benefits e.g. nutrient storage, local fisheries, flood protection and carbon tax: 1. A bio-chemistry assignment that aims to quantify the bio-chemical processes that occur in newly created mudflats and salt marsh as a result of managed realignment schemes at various locations along the Essex coast, 2. A fisheries assignment that has the main objective to assess the related benefits of managed re-alignments and the resulting commercial and recreational fisheries benefits. It explores methods for quantifying fish utilisation of recreated inter-tidal areas using secondary data relating to the hydraulic, climatic and chemical nature of the Blackwater Estuary and Salcott Channel and 3. An economics assignment that examines the economic case for a more integrated approach to estuary management for the Blackwater, incorporating the other assignment material. The management actions and strategies for the Blackwater estuary will be appraised and evaluated via the economic cost-benefit approach and if felt necessary via multi-criteria analysis (using monetary and non-monetary decision criteria). The aim will be to investigate the social costs and benefits associated with the management strategy. In particular, managed realignment, as a component of a coastal protection or sea defence policy will be appraised.

The ComCoast projects ends in the end of 2007 and the results from this inter-disciplinary process are already available. They are the first original data of this type for the UK[7].

7 Conclusion

Flood and coastal defence is a significant benefit provided by habitats recreated through managed realignment, but there are other environmental and economic benefits, the most significant of which are benefits to fisheries, carbon storage and nutrient and contaminant recycling. Case studies demonstrate the significance of environmental valuation for use in the economic appraisal of flood risk management schemes. The inclusion of economic benefits of habitat recreation in project appraisal could have a significant effect on option choice and could mean that more managed realignment schemes would be justified on economic grounds. In conclusion, the significance of valuation is that:

[7] It is anticipated that the knowledge will be disseminated as practical guidance and information for regulatory bodies and those organisations that are involved in the design, implementation and management of managed realignment schemes. The guidance will assist in improving the environmental performance of schemes and in particular enhance the value of the scheme in terms of fish utilisation and nutrient capture.

- it can help transform a process of abandonment into one of a positive re-creation of habitat through realignment and
- it provides an incentive within the generality of FRM schemes to recreate/create habitat wherever possible to do so.

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Ecosystem Services of Peatlands

Achim Schäfer [1]

Presentation
selected slides

(Main Source: Barthelmes, A. et al. (2005): Erlenaufforstung auf wiedervernässten Niedermooren. Alnus Leitfaden. Greifswald: Institut für Dauerhaft Umweltgerechte Entwicklung von Naturräumen der Erde (DUENE) e.V.)

Definition and facts

Peatlands are

- areas with peat
- mires are peatlands that accumulate peat
- fens are percolation, inundation, and terrestrialision mires

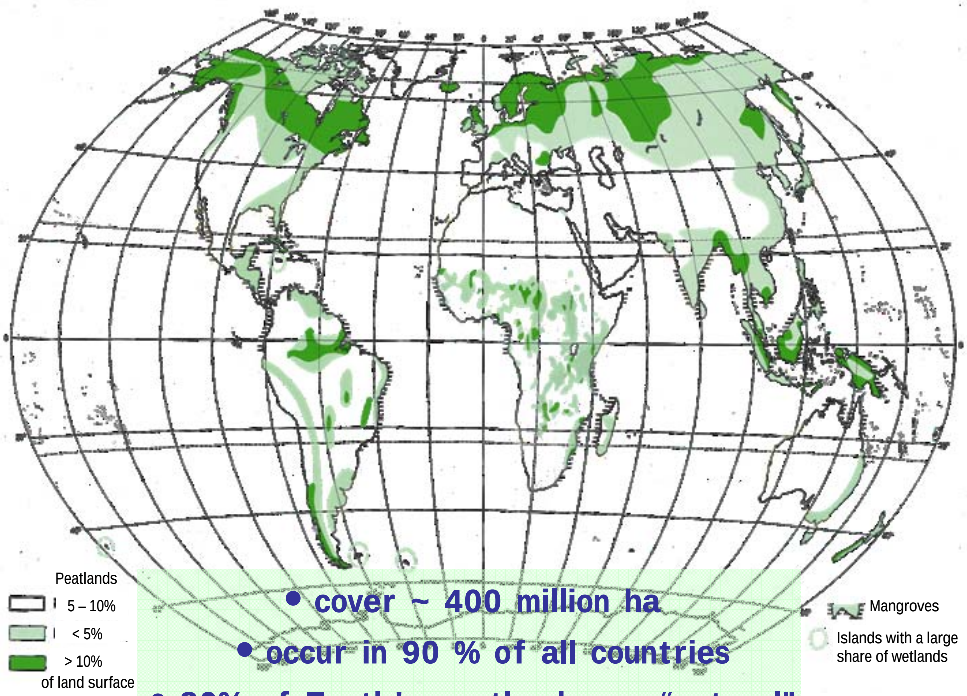
Peatlands cover

- "only" 3 % of the land area
- but contain 30% of the world's soil carbon
- an equivalent of 60% of all atmospheric carbon
- as much carbon as all terrestrial biomass

➢ Peatlands are important for the global carbon cyle

[1] Institute for Sustainable Development of Landscapes of the Earth (DUENE e.V) c/o Institute of Botany and Landscape Ecology, University of Greifswald, schaefea@uni-greifswald.de

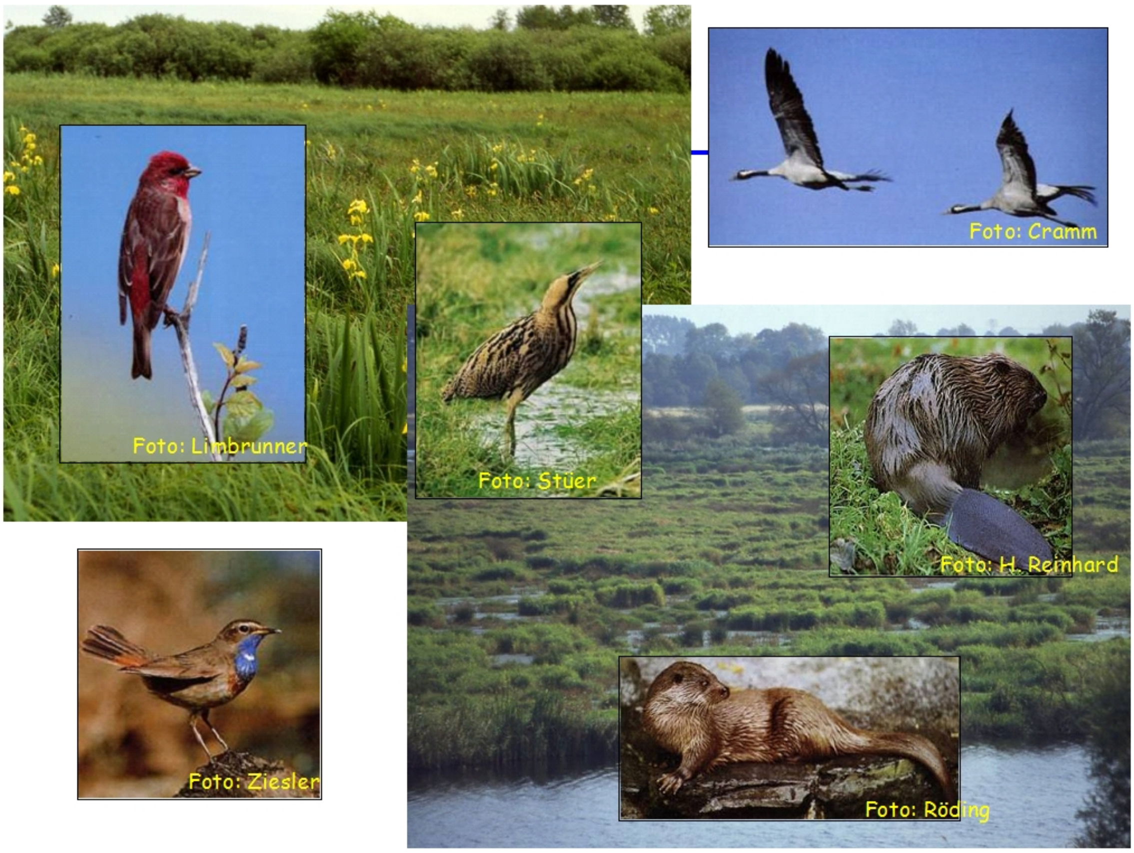
Where peatlands are



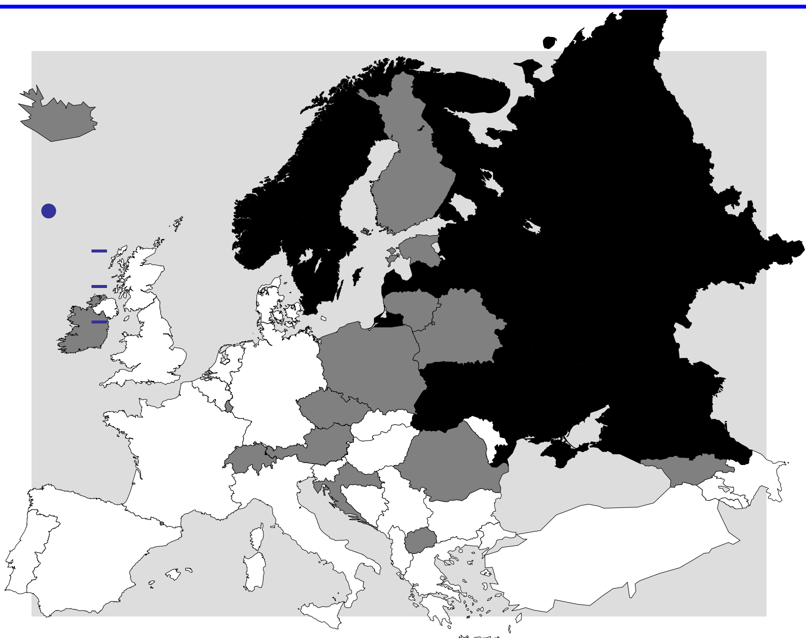
- cover ~ 400 million ha
- occur in 90 % of all countries
- 80% of Earth's peatlands are “natural”
- 60% still accumulate peat

Source: Lappalainen 1996

Peatlands and biodiversity



Peatlands are endangered



Mires left

- < 10%
- 10-50%
- > 50%

of original area

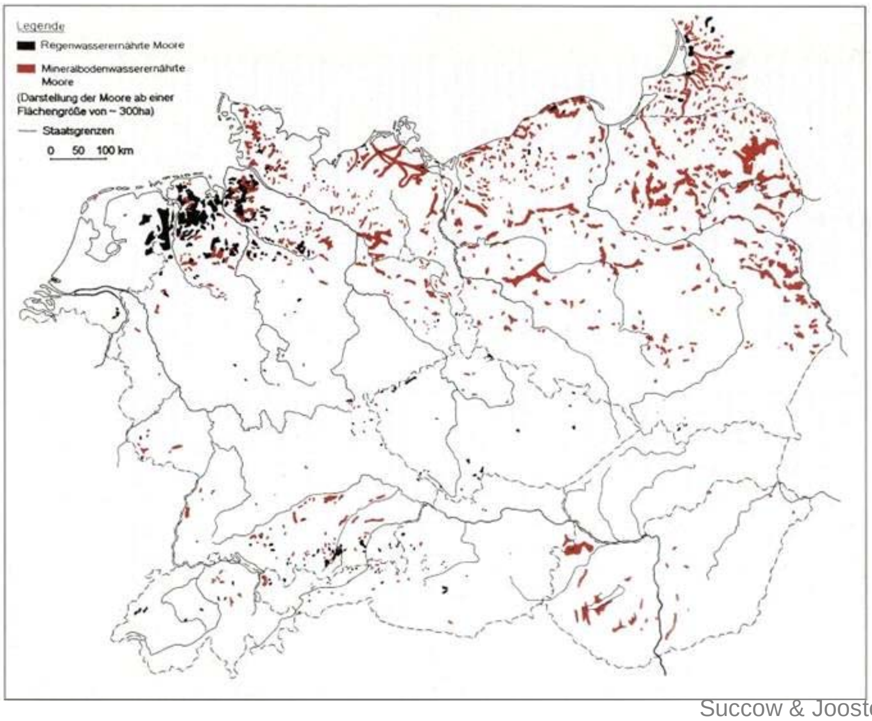
Total left: 48 %

(33 % excl. Russia)

Humans have destroyed > 50 million ha of mires:
50 % by agriculture
30 % by forestry
10 % by peat excavation

Source: Joosten 2003

Peatlands in Central Europe



Succow & Joosten 2001

Losses of peatlands in Germany

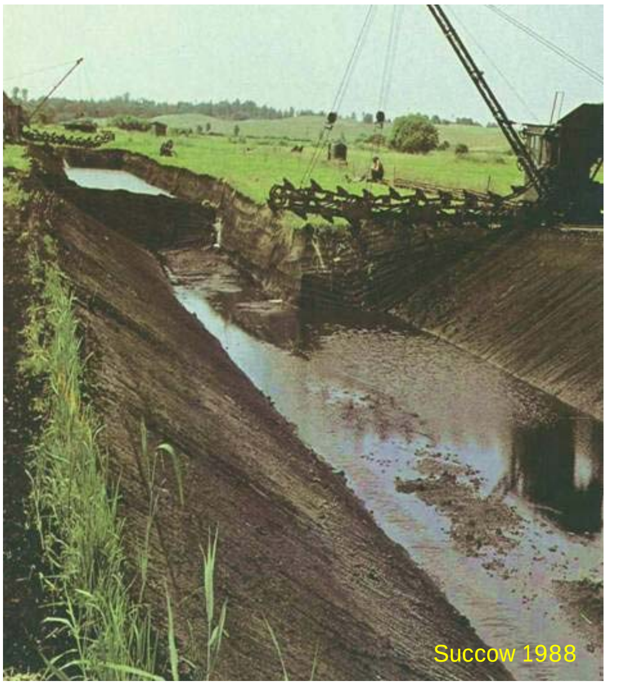
	Losses		Period
	ha	%	
Schleswig-Holstein	30.000	17	1954 - 98
Niedersachsen	50.000	24	1980 - 97
Mecklenburg- Vorpommern	28.800	13	1965 - 95
Brandenburg	60.000	28	1965 - 00
Baden-Württemberg		12	1960 - 95
Bayern	80.000	40	1914 - 92
Sum	**248.000**		

Drained fens – a strong source of greenhouse gases

> 95% of north-East German fens drained

Carbon dynamics of peat is strongly accelerated after drainage

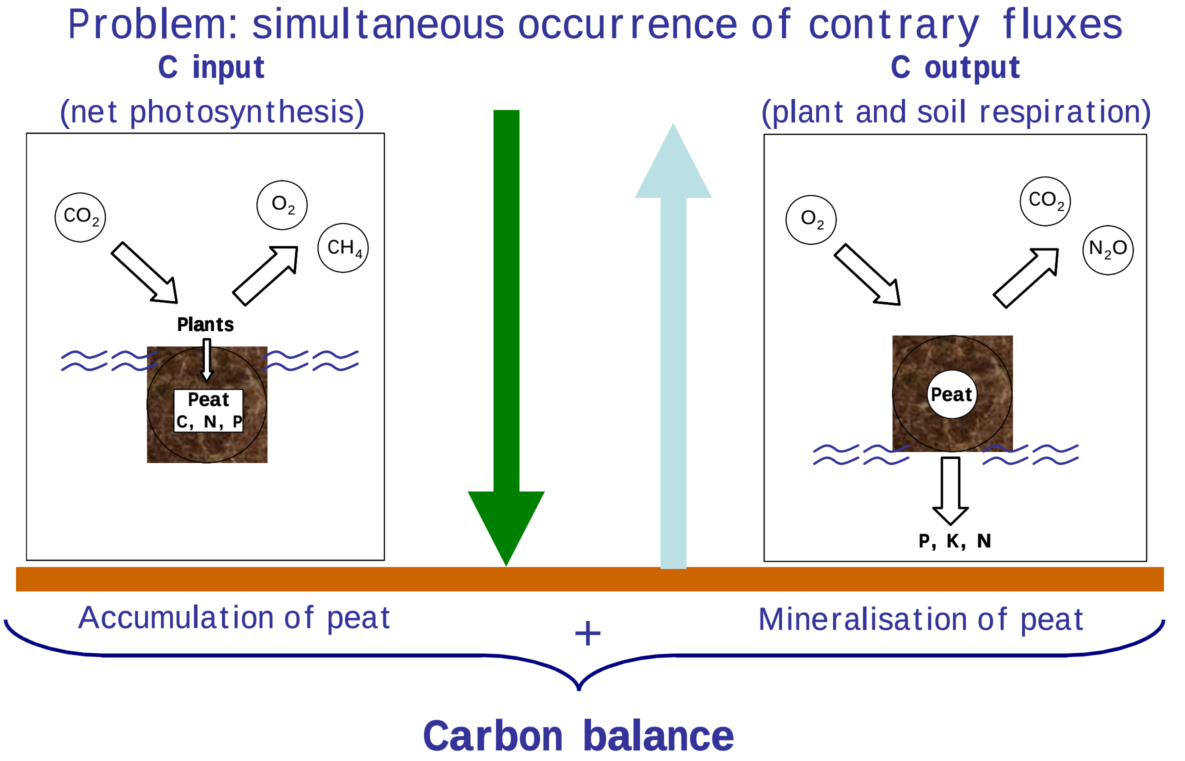
Strong N_2O source
Very weak CH_4 source
Weak - middle CO_2 source?



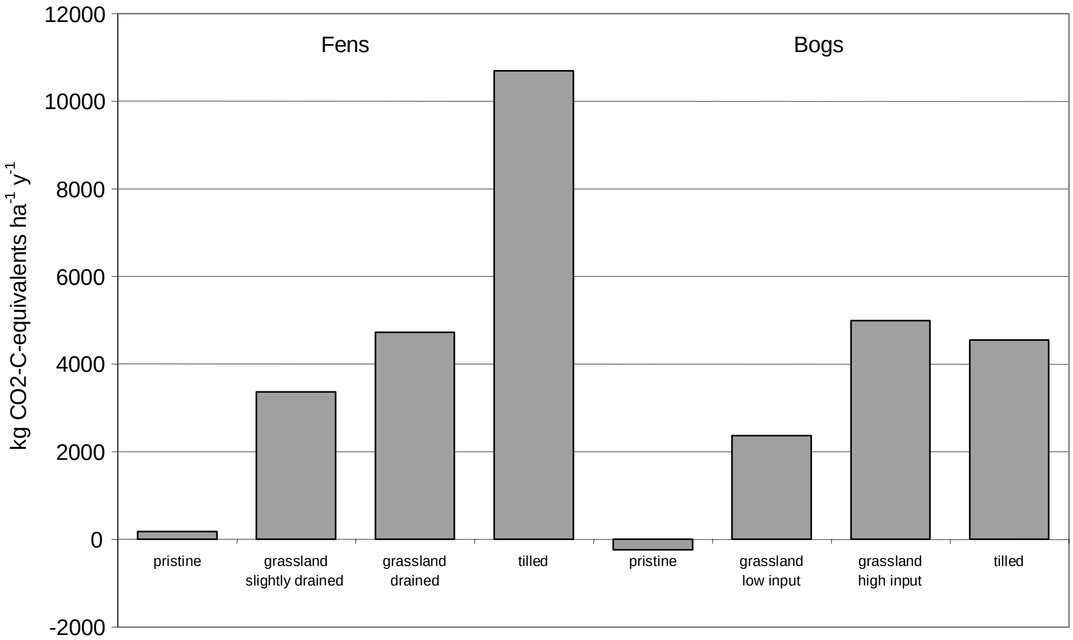
Loss and shrinkage of peat

Peenetalmoor at Lake Malchin

Determination of net CO_2 fluxes



Effect of use of peatlands on radiative forcing



Source: after Höper 2000

Consequences of an unsustainable use of fen peatlands

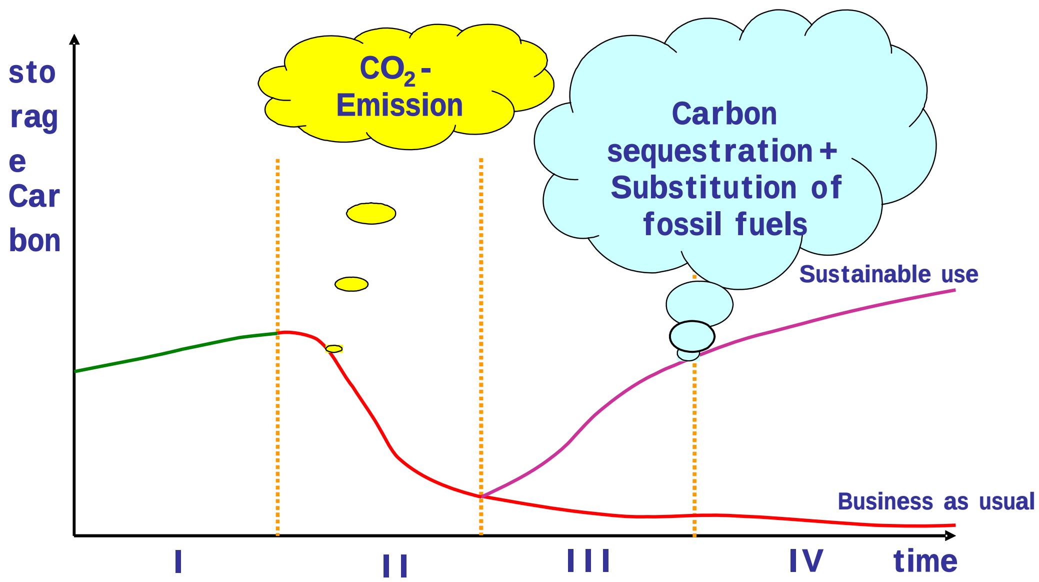
- Conventional agricultural use of peatlands have negative effects on
  - aquatic ecosystems
  - atmosphere (radiative forcing)

	kg ha^{-1} a^{-1}	Norddeutschland
Phosphorus [1)]	1 -16	830 – 13.300 t a^{-1}
Nitrogen [1)]	75 – 470	62 – 390 kt a^{-1}
Carbon [2)]	6.600 - 16.500	5 – 14 Mt a^{-1}
Carbon dioxide	24.100 - 61.700	20 – 50 Mt a^{-1}

1) Gelbrecht et al. 2001, 2) Höper 2002

Compared to:
CO_2-avoidance from the „Renewable Energy Sources Act Incentive Programme“: 0,5 Mt a^{-1}

Business as usual or sustainable use?



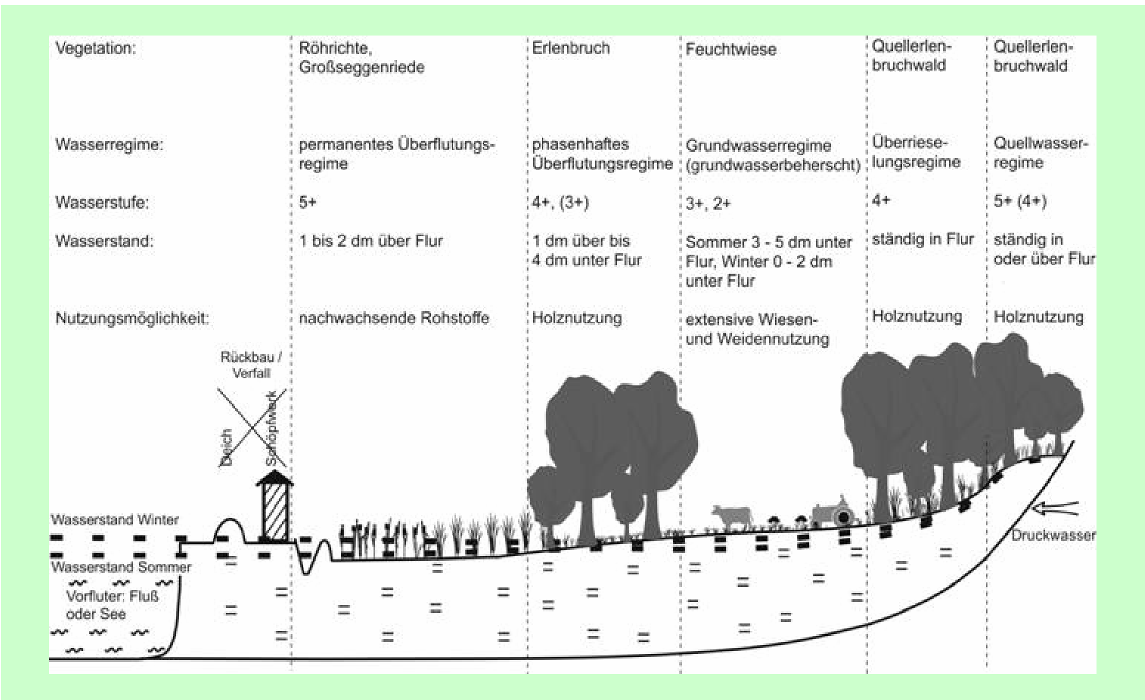
Sustainable use of peatlands

- Land use alternatives
  - Nature protection and landscape protection
  - Production of biomass for energetic use or as raw material
  - Production of wood (red alder) and reed on re-wetted sites

- Prerequisites
  - Natural hydrogenic conditions
  - Deconstruction of dikes, pumps, drainage ditches
  - No water management

- Possible problems
  - Not all sites are optimal for rewetting (catchment area, water availabillity)
  - Technical problems (cultivation and harvest)

Sustainable use of peatlands



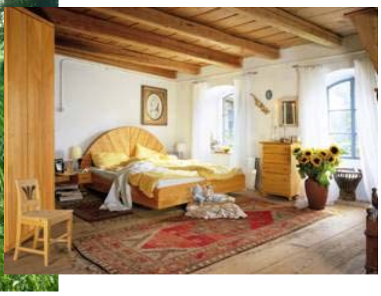
Source: after Succow & Joosten 2001

Functions of rewetted peatlands

Function	Description
Sink	Deposition and recycling of nutrients
Disposal	Carbon sequestration
Regulation	Keeping cultural landscapes open; site- and culture-specific biodiversity; ground-water retention
Conservation	Regional responsibility for plant communities; key species
Production	Fodder, food, biomass, raw materials
Information	Landscape beauty, recreation, aesthetics and cognition, research

Sink- and production function

High quality timber



Furniture from solid wood

Reed as a raw material for thatching

Economic Valuation

- To whom?
- For what?
  - Financing of investions to get revenues
  - Tradeable services on markets
  - Criteria for decision making
- Which criteria?
  - Efficiency is the „strongest“ economic argument

Example: Using rewetted peatlands for wood production - profitablity

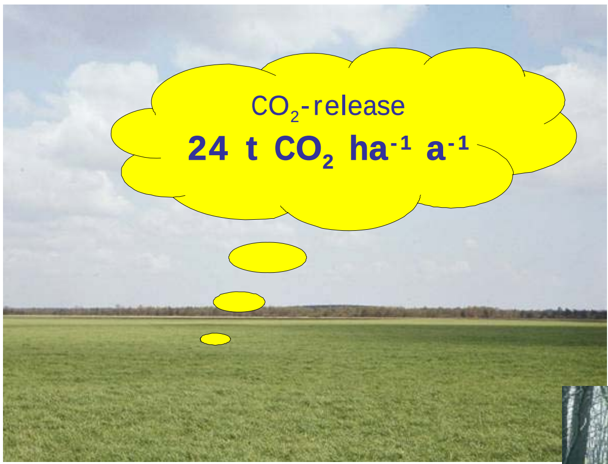
Present Value (one rotation period)	3 % p.a.	2 % p.a.
	EUR ha^{-1}	
Variant 1	-3.777	-2.582
Variant 2	-2.278	245

Variant 1: Common provenance of plants, average yield class, standard forest management

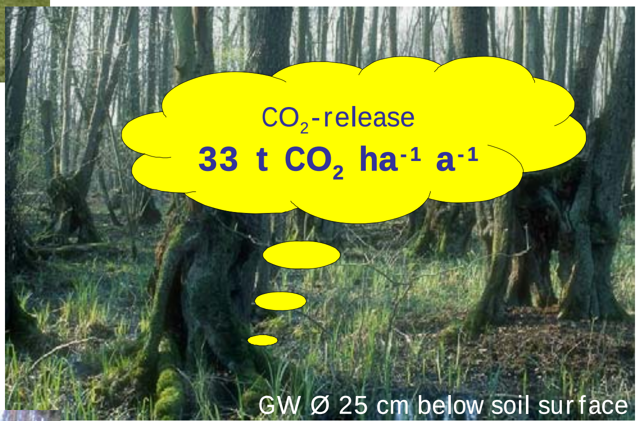
Variant 2: Best provenance of plants, best yield class, best forest management

Land expectation value	3 % p.a.	2 % p.a.
	EUR ha^{-1} a^{-1}	
Variant 1	-114	-52
Variant 2	-69	5

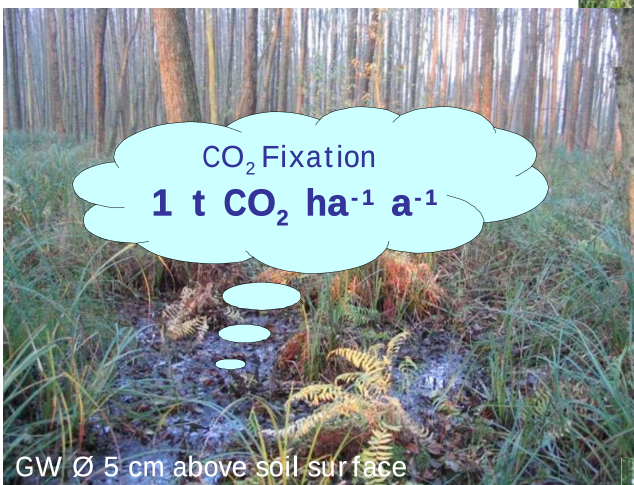
CO_2 balance grassland on peat soil



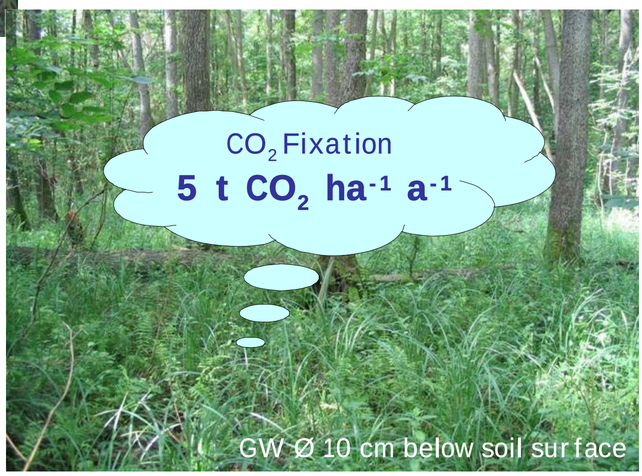
CO_2 balance drained alder stand



CO_2 balance permanently overflowed alder stand



CO_2 balance managed alder stand



CO_2 fixation and wood production

	CO_2-Fixation kg ha^{-1} yr^{-1}
a) Eternal CO_2-Sink (Continuous peat production)	244 - 3.700
b) Temporal CO_2-Sink (Wood production in 70 years)	7.428

c) CO_2-Emission avoidance through substitution of fossil fuels	**CO_2-Substitution** kg ha^{-1} yr^{-1}
- energetic use of wood from thinning	2.054
- complete energetic use of the wood	6.000

Monetary value of a sustainable wood production on rewetted peatlands

- Services are the beneficial outcomes that result from ecosystem functions.
- One service which society will get by restoring peatlands are reduced emissions of greenhouse gases.
- A negative Land Expectation Value is the price which society have to pay for the supply of an ecosystem service.
- What are the alternatives?
- Comparing the costs with other mitigation measures and find the cheapest mitigation strategy.

Monetary value of a sustainable wood production on rewetted peatlands

	EUR per t CO_2
Insulation of buildings	350 – 750
Renewable Energy Sources Act Incentive Programme	200
Windmill energy	70
Petrol taxes	60
Hydropower	22
Alder forestry on rewetted fens	1 – 2

Investing in peatland conservation is more effective than other CO_2 mitigation.

Substitution of fossil fuels

- Further emission avoidance can be reached by using the biomass on the rewetted fens for substituting fossil fuels and raw materials.
- Such "paludicultures" are in conformity with the European policy that envisages the replacement of fossil fuels with biomass.
- While "paludicultures" as such may not be fully profitable, they become so by taking CO_2 avoidance into consideration.

Some "soft" arguments

- Additional benefits of rewetting
  - revitalisation of traditional land use combined with new ways of processing
  - new land-use concepts with minimal damage
  - raw materials for energetic and industrial uses
  - increase in energy political autarchy
  - improved perspectives for (eco)tourism
  - restored habitats for rare species/communities
  - improved landscape hydrology
  - reduced nutrient run-off into the Baltic Sea

Effects of Organic Farming on Humus, Carbon and its Contribution to Climate Changes

Uta Hoyer [1]

According to Article 3.4 of the Kyoto Protocol (available under www.unfccc.int), agriculture is explicitly regarded as a possibility for reducing CO_2-emissions by corresponding management. In this connection, essential importance falls to the humus content in the soil as source and sink of carbon from the atmosphere. The question we are facing is whether and to which extent organic farming can make a contribution to C-sequestration and how it may be evaluated. First of all, the paper describes basic functions of humus in cultivated soils and how they may be influenced. With comparisons between organic and conventional farming differences in humus content and carbon fluxes are elucidated.

1 Humus and its functions in cultivated soils

In cultivated soils, humus fulfils a large number of important tasks, among them storage and mobilisation of nutrients, filter and buffer functions, water retention ability, improvement of the soil structure (aggregate stability, pore volume, air and water regime, crumb structure), improvement of the soil-microbial activity as well as CO_2 sink and store.

Thus, a sufficient content of humus is essential for a long-term achievement of high and stable yields (DICK & GREGORICH, 2004; SAUERBECK, 1992). Especially in organic farming, where the application of mineral nitrogen fertilizer is prohibited, humus management requires major attention, in order to guarantee an optimal nutrition of the plants. This involves the hypothesis that organic farm management would lead to higher humus levels, thus raising the sequestration of atmospheric CO_2 in the soil organic matter (SOM) (ARDEN-CLARKE & HODGES, 1988; SMITH, 2004).

Humus can be subdivided into several pools of differing stability and turnover rate (KÖRSCHENS et al., 1998). At least two major pools can be distinguished: an inert and a decomposable pool. The inert fraction of the SOM includes the humus or C- and N-content of the soil that never falls below a minimum level under natural conditions with long-term abstention from any fertilizer input and cultivation of humus depleting

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crops or clean fallow. It is largely independent of mineralization processes and site conditions (KÖRSCHENS, 1980). A regular supply of major quantities of crop and root residues as well as organic fertilizers favours the accumulation of organic intermediate products remaining in the soil. They form the decomposable pool of SOM (DSOM), which is subjected to gradual mineralization. The proportion of DSOM amounts to approx. 30 % of the total SOM and is decisively influenced by the management system. Changes in the total humus content of the soil are mainly the result of changes in DSOM. The strongest effect on the humus content is exerted by the site conditions as well as by climate and soil texture (HASSINK et al., 1997). Corresponding management methods allow to influence DSOM considerably. Highest humus levels resulting from the type of landuse are recorded in natural soils and in forest locations (CARTER et al., 1998). Changes in landuse, particularly the clearing of woodland and its transformation to arable areas, entail a loss of 30 to 50 % of the soil-borne carbon. This becomes evident when we take into consideration that about 54 Gt C were released to the air in the last 150 years by land conversion (SAUERBECK, 2001) and contributed largely to the global rise of CO_2-contents in the atmosphere. Owing to lower C-inputs, arable soils usually have lower C-stocks than grassland or forests (SMITH, 2005). However, their historically high C-losses explain the fact that they have the highest C-sequestration potential (PAUSTIAN et al., 2000).

Fig. 1 shows the possibilities of carbon reduction or accumulation in the soil by modifying landuse. Interferences like transformation of woodland into arable land destroy the original equilibrium; and thus LP/D < 1; humus decomposition (D) becomes higher than the production of organic residues (litter production = LP). This again leads to the phenomenon that SOM decreases fast in the beginning, but later this process will be slowing down. When the new management type remains constant, a new steady state will develop which will be lower than the state before the shift (LP/D = 1). A change of the system to a situation, where LP/D > 1, induces accumulation of SOM until a new steady state will have established. This may involve a low, moderate or strong rise of the humus level, depending on the management regime (JOHNSON et al., 1995). Also this rise will be fast in the beginning and slow down until the new steady state is reached. The new state will last as long as the new management system is supported (PAUSTIAN et al., 2000). Every soil has its specific C-carrying capacity which represents a function of the given site conditions and soil formation. This means that the C-storage capacity of the soil is limited (VERNON COLE et al., 1993).

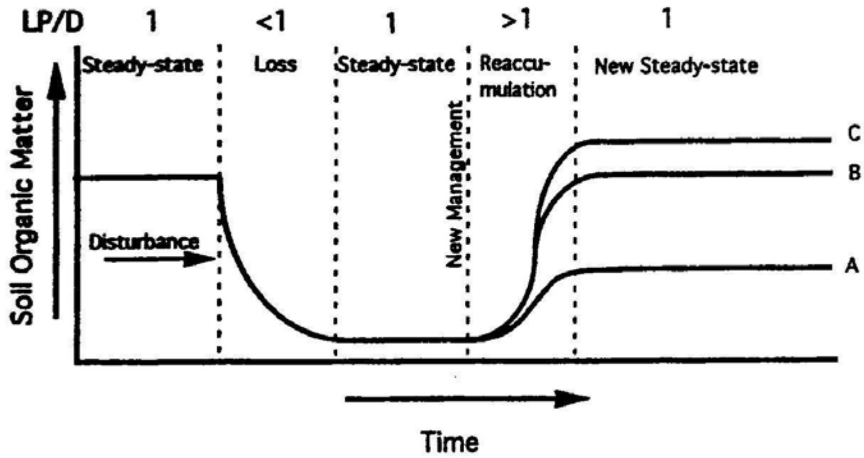

Fig. 1: Model concept of humus decomposition and accumulation after management changes (JOHNSON et al., 1995)

2 Comparison of humus contents in organic and conventional systems

In order to study the differences between organic and conventional farm management with regard to their humus levels, soil samples were taken in different regions of Germany in spring 2005 and 2006 on organically and conventionally cultivated adjacent fields of equal soil texture with subsequent analyses of different SOM-relevant measuring parameters.

Fig. 2 demonstrates the relationship between the percentage of fine fraction (clay [%] + fine silt [%]) and the C_{org}-content as well as the microbial biomass. The higher the content of fine fraction in the soil, the higher are also the levels of C_{org}. There were only

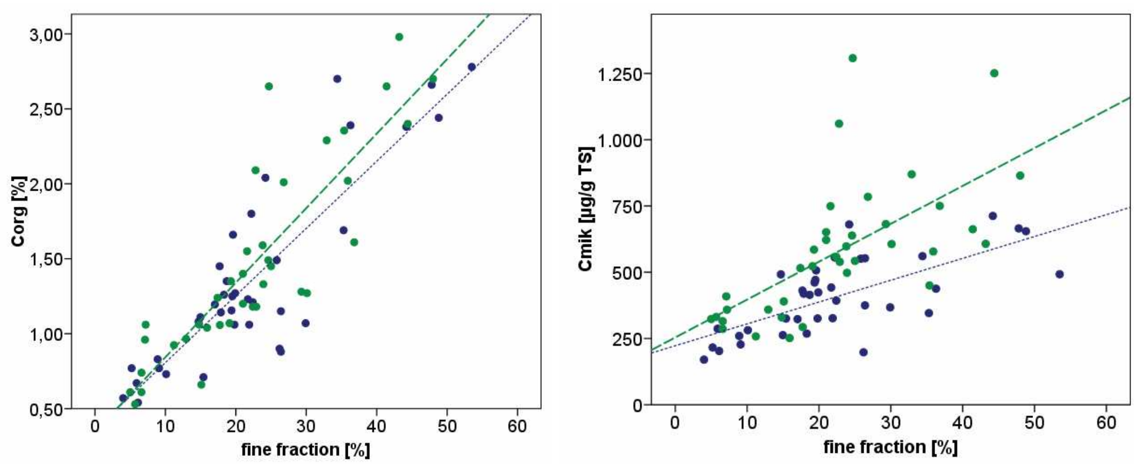

Fig. 2: Relationship between the percentage of fine faction in the soil and the contents of Corg (left) and microbial biomass-C (Cmic) (right); green: organic system, blue:

negligible, insignificant differences between organic and conventional management. The microbial biomass, however, reached distinctly higher values under organic management (Hoyer et al., 2007a).

It is obvious that organic management does not necessarily lead to higher humus contents compared to conventional farming and that the site conditions are the main influence factor. An explanation might be the fact that in organic farming an increasing number of simple-structured farming systems with low humus replacement performance is found (for example cash crop farms with a high proportion of cereals on low-productivity sites). But also the comparative conventional farms may differ considerably in their structure. Organically managed soils, on the other hand, have a clearly higher microbial activity, in order to mineralize sufficient nutrients for the plants (HOYER et al., 2007b). Presumably, the sampled ecological areas had already reached their maximum C-uptake capacity, and the additionally supplied organic matter was mineralized mainly for nutrient provision. Regarding the humus levels, the effect of site conditions and variability within a system is obviously stronger than the differences between organic and conventional management. Simply distinguishing between organic and conventional farming is therefore not sufficient for the assessment of humus situation. Moreover, profound attention has to be given to the farm-specific management measures.

3 Comparison of C-cycles on the example of an organic and a conventional farming system

For the description of farming systems with regard to their CO_2-emissions, it is not sufficient to consider only the humus stock of the soil. Rather the carbon fluxes of the total farm have to be recorded and evaluated. This includes the interrelated farm-internal C-fluxes between soil, plants, animals and the atmosphere. The farm management program REPRO (HÜLSBERGEN, 2003) represents a tool for estimating these cycles. Figures 3 and 4 show the C-cycles of an organic and a conventional farming system.

The C-cycles of the two systems are basically different. In the organic system, mass fluxes are largely closed. The crop rotation in the organic system consists of clover-grass – potatoes with mustard as catch crop – wheat – sunflowers with clover-grass undersown – wheat – rye with clover-grass undersown. The stock density is 1.4 LSU ha^{-1} (Livestock Units cattle), whose excreta are returned to the fields in form of farmyard manure. Tillage is performed with a plough to 25 cm in depth.

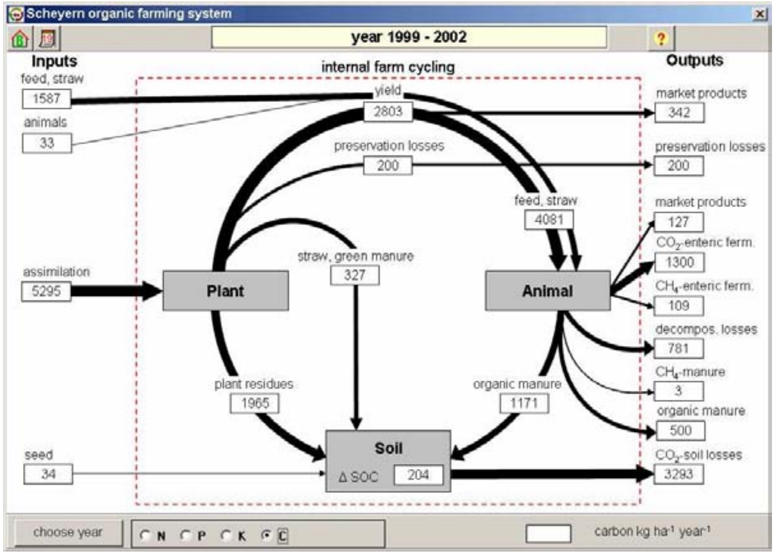

Fig. 3: On-farm C-cycle of an organic system (Küstermann et al., 2007)

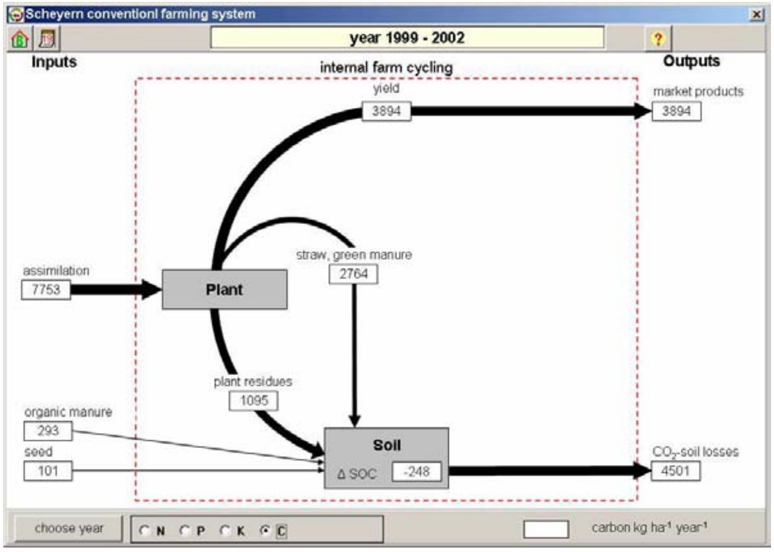

Fig. 4: On-farm C-cycle of a conventional farming system (Küstermann et al., 2007)

The conventional cash crop system includes a four-field crop rotation with potatoes – wheat – silage maize – wheat with mustard as catch crop. Organic fertilizer is applied in form of liquid manure (imported) at the adequate level of 0.6 LSU/ha. Tillage is executed without ploughing not deeper than 10 cm. The two farming systems have different yield levels: the organic farm produces 4 t ha^{-1} wheat and 26 t ha^{-1} potatoes on average, the conventional farm reaches 7 t ha^{-1} wheat and 37 t ha^{-1} potatoes. Crop rotation and fertilization lead to different C-inputs in both systems (Fig. 5).

In the organic system, the C input with crop and root residues is 80 % higher compared with the conventional variant. In the conventional system, on the other hand, large C-quantities are supplied with straw and green manuring and raise the total C-input in the integrated variant beyond the level of the organic variant (RUSER et al., 2007). Nevertheless, depending on the different humus replacement potential of farmyard manure and liquid manure compared to straw and due to the humus accumulation by clover-grass, the program computed an increase in SOM in the organic system and a decline in the conventional counterpart. CO_2-respiration by micro-organisms as calculated by the model software reached 3293 kg ha^{-1} a^{-1} in the organic farm and 4501 kg ha^{-1} a^{-1} in the integrated system. Caused by different yield levels and deviating product use, C-output is also different. In cash crops, 342 kg C ha^{-1} a^{-1} are bound in the organic and 3894 kg C ha^{-1} a^{-1} in the integrated system. The C-amounts in animal products amount to 127 kg C ha^{-1} a^{-1} in the organic farm. Here, other than in the integrated system, C-loss processes take place, i. e. 109 kg C ha^{-1} a^{-1} metabolic CH_4, 1300 kg C ha^{-1} a^{-1} CO_2-emissions from the animals as well as losses from manure storage were determined (KÜSTERMANN et al., 2007).

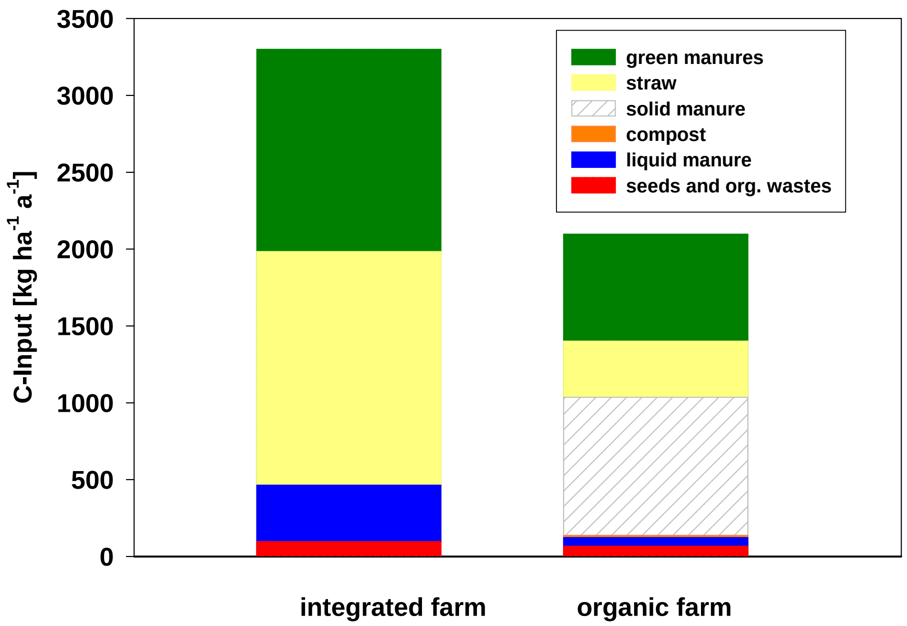

Fig. 5: Input of organic C on arable areas of the conventional and of the organic managed farm (Ruser et al., 2007)

4 Conclusions

In arable farming, soils have the potential to sequestrate a certain quantity of C. With an increasing percentage of fine particles in the soil the potential of C-sequestration rises. Comparisons between farm pairs with organic and conventional systems revealed a trend to higher C_{org}-levels on the organic areas, which allows to conclude that more C can be stored in the soil owing to different crop rotations and frequently higher inputs of organic fertilizers. Simultaneously, however, an intensification of microbial decomposition processes in the soil is observed. Distinguishing exclusively between organic and conventional management is therefore not sufficient for evaluations of the humus budget. Moreover, the regarded farms have to be profoundly analyzed for their specific management system. Evaluations of the climatic relevance of farming systems require also to depict and characterize all C-fluxes within a farm. On the basis of two example systems it has been demonstrated how carbon inputs and outputs can be quantified in organic and integrated systems as well as fluxes between single compartments. Although the conventional variant has a higher total C-input, it is an intensive system with large CO_2 release. The organic system, on the other hand, has largely closed mass cycles, and a major proportion of carbon is stored on SOM.

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Impact of Organic Farming Systems on Soil Erosion

Norman Siebrecht & Kurt-Jürgen Hülsbergen [1]

Organic farming is often understood as a form of landuse especially sound for the environment. Nevertheless, also under these management conditions adverse effects, for example soil erosion, may be recorded. Reducing such negative concomitants requires suitable tools to estimate the extent of erosion. However, the available models pay only insufficient attention to the specific conditions of organic farming, and thus may indicate excessive soil loss. It is shown that organic farm management may inhibit the occurrence of erosion.

1 Introduction and outline of the problem

The guiding principle of organic farming is environment-friendly land management implying the promotion of natural life processes and closed mass cycles. The used resources are to be safeguarded and maintained by cautious treatment. With regard to the soil, management is targeted at conserving the fertility of soils, enhancing their quality, protecting soil microorganisms and soil structure (BIOLAND 2006, NATURLAND 2006). In view of these objectives, it is often suggested that organic farming involves less adverse effects for the environment. Yet, the mentioned approach may cause also negative developments. To estimate such effects, it is necessary to analyse a farm from the angle of its specific management measures and the resulting impact on the environment. In the field of agriculture, numerous systems for farm and environment management have been developed, which can be used for analysing and evaluating the influence on the environment and for optimising farm processes. In Germany, mainly the computer programs KUL (Eckert & Breitschuh 1994) and REPRO should be mentioned.

The present paper describes a method for the analysis of soil erosion which allows assessing the influence of farm operations on the loss of soil. Due to the great importance of water erosion, other types of erosion (e.g. wind or technological causes) were neglected in this study. In addition to the methodical description, results obtained in an example farm are presented, which characterise the effects of organic farming on soil erosion.

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2 The Environment and Farm Management System REPRO

REPRO is a module-based software for the sector of agriculture (Fig.1) and allows to analyse and evaluate farm enterprises in manifold fields (Hülsbergen 2003, Küstermann et al. 2007). For this purpose, information on site conditions (soil and climate), farm structure (fields, crop rotation, ..), production technologies as well as yields and performance are entered. These data are processed in Module 1 "Management Conditions" and make it possible to describe a farm in its entity. Emanating from this, analyses (Menu 2) can be computed on the basis of indicators for the different subject areas. In the past, the attention focused on the abiotic environment (nutrient, energy, harmful soil compaction, ..); analyses of the biotic environment are currently supplemented (biodiversity). Under Menu 3 "Evaluation", the indicators are linked with target values and evaluated; farm comparisons are made or checkups of the compliance with default values.

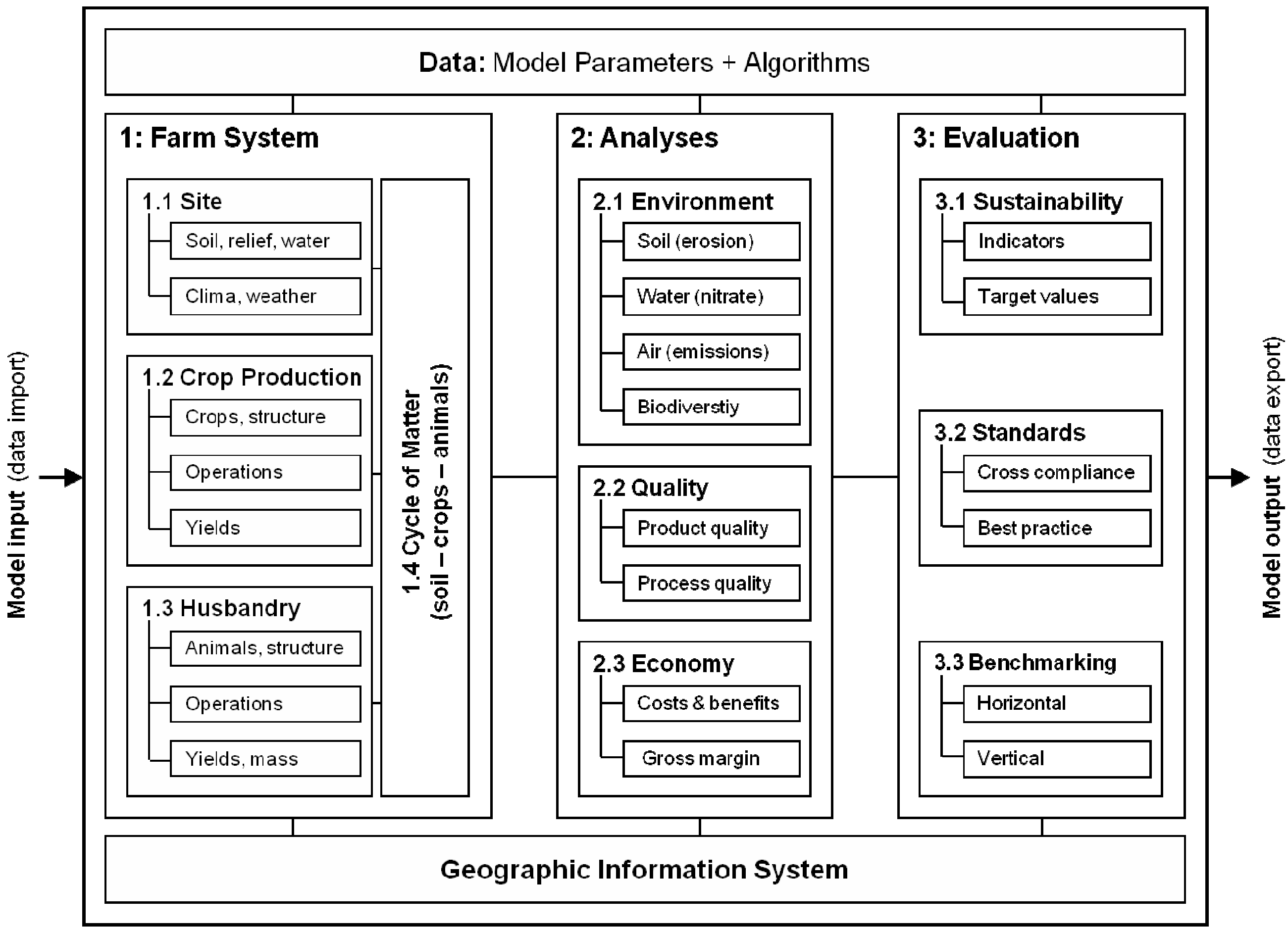

Fig. 1: Structure of the Environment and Farm Management System REPRO

3 Estimation of soil erosion by water in REPRO

For the estimation of erosion risk, a practice-related approach was designed for REPRO, which allows to analyse and evaluate soil erosion on farm level. For this purpose, the Universal Soil Loss Equation (Schwertmann et al. 1987) was integrated. This empirical model is based on an evaluation of more than 10,000 loss measurements on plot level, with the factors relevant for erosion processes summarised in an equation system. Losses (A) are determined by multiplication of the factors. A detailed description of the factors is given by Schwertmann et al. (1987) or in DIN 19708.

$A = R \cdot K \cdot C \cdot L \cdot S \cdot P$

A Mean annual soil loss [$t\ ha^{-1}\ a^{-1}$]
R Erosivity of rainfall
K Erodibility of the soil
L Slope length
S Slope steepness
C Soil cover and cultivation factor
P Protective measures against erosion

- R-factor: The term erosivity includes the rainfall or rainfall characteristics at a site as cause of soil loss. For its estimation, data of the mean annual or summer precipitation are used, and region-specific regression functions are then applied for calculating the R-factor.
- K-factor: The term erodibility stands for the stability or „resistance" of a soil to erosion with consideration of the soil properties. Soil valuation maps are used whose descriptive signs are the basis for K-factor determinations.
- L-factor: The L-factor characterises the influence of surface runoff on the loss of soil. Increased slope lengths enhance runoff and the carried load of soil. The surface runoff is computed by a geographic information systems (GIS) including hydrological modelling (Van Remortel, 2001). The input data contain relief information in form of a Digital Elevation Model (DEM).
- S-factor: The S-factor considers the relationship between slope steepness and surface runoff in form of the flow rate. A rising slope gradient leads to an acceleration of surface runoff, thus increasing shearing strength and transport capacity – soil losses go up. This factor as well is determined by use of GIS. After calculations of the slope steepness, the S-factor is computed (Hickey 2000).

- C-factor: The C-factor is determined according to the methodics by Schwertmann et al. (1986: pp. 38). Crop-related relative soil losses (RBA) within defined growth periods are linked with the temporal distribution of the R-factor (RRA). The REPRO model considers the grown crops and the dates of respective field operations (cropping periods) on the level of the regarded management unit (subfield). The required information on RRA or RBA is furnished by databases.
- P-factor: This factor includes the applied conservation measures which can be selected when entering details of the production technology (for example consideration of contour).

The factors are marked by a high spatial variability which requires a differentiated approach, i. e. to enter spatial data (geodata). Therefore, the calculation method combines REPRO with a Geographic Information System (GIS) which offers all necessary functions for handling and analysing spatial data. The spatially variable factors are computed by GIS, the farm-specific information on management details by REPRO. Thus, a complete and blanket coverage analysis of soil erosion becomes possible.

The obtained results can be interpreted for different levels and reference units (Table 1): The mean loss of soil can be indicated for the total arable area (fields) of a farm or broken down to each single field or even subfield. Apart from this, the method provides a spatial depiction of the results in form of maps, thus disclosing sectors where soil losses are especially high (Fig. 2).

Table 1 Excerpt from a table showing erosion estimates and loss information on farm and field level

Area								Soil loss
Name	Size	Distribution of erosion classes within field						Mean
Field	[ha]	< 0.2	0.2 - 1.0	1.0 - 2.0	2.0 - 3.0	3.0 - 6.0	> 6.0	[t ha^{-1} a^{-1}]
A 01	4.8	0.33	0.40	0.19	0.06	0.02	0.00	**0.9**
A 02	3.3	0.05	0.27	0.36	0.28	0.03	0.00	1.5
A 03	3.8	0.08	0.30	0.22	0.19	0.20	0.01	1.9
...	...	...	...	...	...	...	...	...
A 21	1.4	0.00	0.17	0.25	0.31	0.24	0.02	**2.4**
Farm	...	...	...	...	...	...	...	**1.9**

The method allows to elaborate suitable measures for reducing soil loss. It permits also to estimate the effects of the derived scenarios and to select the most appropriate variant. The consequences of these measures can be tested, again by REPRO, for their impact on other sectors (i. e. energy intensity, soil compaction, ..), in order to check the relationships of management measures with other farm sections.

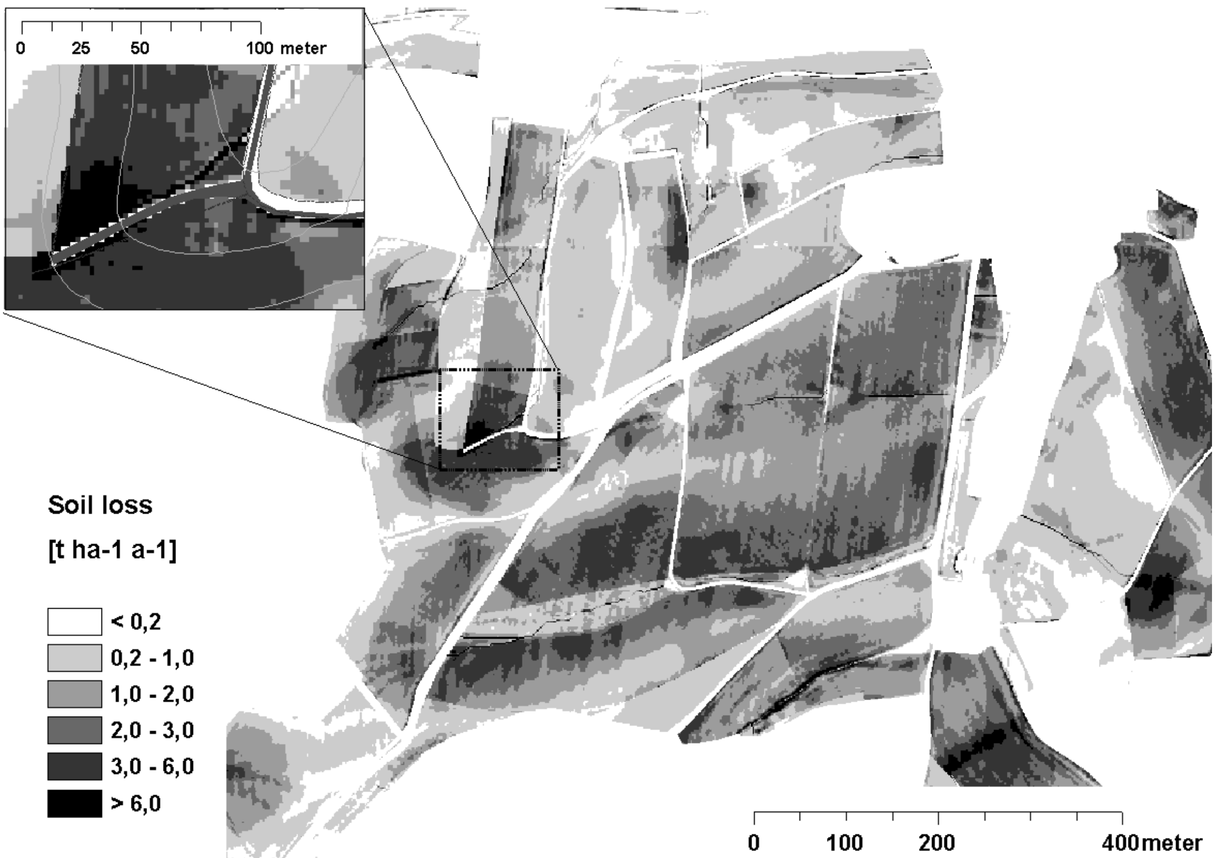

Fig. 1: Clip of a map with results of erosion estimates

The applied model was validated under the conditions of the Experimental Farm Scheyern. Below, important results are presented.

4 Soil erosion in an organically and a conventionally run farm

The Experimental Farm Scheyern is located in the Upper Bavarian Tertiary high-loess hills about 40 km northeast of Munich. Due to the site conditions (precipitation: 800 mm a^{-1}, temperature: 8.5°C, sandy loam, field valuation scores: 45 – 60), the farm has a high erosion potential. That's why soil erosion was an essential research subject in the experimental management system established in 1993 (ecological area 43 ha, integrated area 68 ha). Numerous measurements and activities towards erosion control have been carried out. On the basis of test results from long-term investigations which included also measurements of surface runoff and resulting soil loss in drinking water catchment areas (Fiener & Auerswald 2003), comparisons became possible between the computed results for the organically and the conventionally managed farm branches.

In the conventional farm system, the measured loss of soil from arable areas reached 2.3 t ha^{-1} a^{-1} vis-à-vis a computed loss of 2.8 t ha^{-1} a^{-1}. Supposing a methodical error of about 10 %, we confirm an adequate estimation of soil loss by the USLE under conventional management. In the organic farm, however, measurements reached 0.2 t ha^{-1} a^{-1}; the computed values amounted to 1.8 t ha^{-1} a^{-1}. This deviation by about 90% points out to the fact that the model pays no or only insufficient attention to the benefits of ecological management.

Table 2: Soil loss in the Experimental Farm Scheyern compared to measured and computed values in the conventional and the organic farm system, indicated as t ha-1 a-1; (a: Auerswald et al. 2003, Fiener & Auerswald 2001; b: Siebrecht 2006 unpublished)

	Conventional	**Deviation**	**Organic**	**Deviation**
Measured	2.5	-	0.2	-
Computed a	2.8	+ 11 %	1.7	+ 89 %
Computed b	2.3	+ 9 %	1.8	+ 89 %

First reflections on which factors could be influenced by organic farming were made by Kainz (2007). They can be subdivided as follows:

a) Surface cover and crop effects: The cultivation of grass, clover or grass/clover blends as practised in organic farming enhances soil stability additionally. This result remains effective beyond the year of cultivation and is therefore called carry-over effect. It is entered into the USLE in a simplified form. In the 1st year following cultivation the C-factor is adjusted to 20 %, in the 2nd year to 60 % (Wischmeier & Smith 1978, Table 5d; Fig. 1). However, we should take into account that under German conditions these effects can markedly deviate (Kainz 2006). Another aspect is the influence of weeds on the surface cover, which might lead to an additional protection and stabilisation of the soil. So far, however, nearly no investigations have been made into this subject. But also crop development and soil coverage are different between conventionally and organically managed fields, with corresponding consequences for erosion. A potato crop may elucidate the situation: Organically grown potatoes reach soil coverage sooner because mostly chit tubers are planted. On the other hand, the ground cover may rapidly shrink due to *Phytophthora inf.* infestation, where control measures are restricted.

b) Soil stability (erodibility) and infiltration: The abstention from mineral fertilisers and the spreading of farmyard manure as well are regarded as steps reducing soil losses. It is commonly accepted that manure improves aggregate stability and the infiltration potential of soils (Becher & Kainz 1983, Siegrist et al. 1998).

   Another aspect, which so far has not been investigated in detail, is the omission of chemical plant protection means (PPM) whose application might induce changes in the physico-chemical conditions. This again may have an impact on the microflora and the involved aggregate stability. The abstention from PPM in organic farming could also contribute to a reduction of the erosion potential.

   Numerous studies have demonstrated that ecological agriculture supports higher humus levels, and therefore sweeping statements, as obtained from the use of map information, may favour misinterpretations (Schlichting 1975, Schruft et al. 1982, Weiß 1988). The content of organic matter, however, has a decisive influence on the erodibility; decreasing levels will lower it.

The investigations made in the Experimental Farm Scheyern point out to the possibility of further effects, which are not commented here. In view of the described criteria of organic farming, we can conclude that this may contribute to a reduction of soil losses.

5 Discussion and prospects

The USLE turned out to be a reliable method for the prognosis of soil losses from farmed areas in Central Europe. The described approach underlines the possibilities for analysing soil erosion on farm level and helps derive optimisation strategies. The chosen methodics and the input data are suited as indicator for applications on national level. All input data are available, and the method has been sufficiently validated. For management practices deviating from the conventional standard, reliable statements can only be made after special adaptation of the methodics. For this reason, a project funded by the Federal Agency for Agriculture and Nutrition has been initiated with the target to adapt the method to the conditions of organic farming. It is expected to end in 2009.

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Functions, Assessments and Optimisation of Linear Landscape Elements

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Abstract

Linear landscape elements (LLE) in cultural landscapes are originated from human land use. Hedgerows, buffer strips, balks, tree rows, roads, waters, ditches and other linear landscape elements are especially widespread in intensively used landscapes. LLE are of high importance for diverse landscape functions, biodiversity or the landscape balance. The article discusses spatial explicit monitoring and data, assessment approaches and the optimisation of the spatial allocation of LLE in the context of landscape functions and ecosystem services. Examples are given for an inventory of LLE on a regional scale and for the assessment of the habitat structure for a key species of open arable landscapes. An optimal compromise for the distribution of linear landscape elements in an agricultural landscape is proposed and discussed.

1 Landscape Functions

As stated in Meyer et al. (2004) the discussion of functions "often relates to the landscape concept. According to Volk & Steinhardt (2002) land use acts as an interface between natural- and socio-economic systems. This implies that the term landscape is narrowly linked to the term land use, which reflects the "functional dimension" of the socio-economic situation in a region (Büchs, 2003). Regions defined as sub-national territories are historically grown spatial units, characterized by a common cultural context (traditions), language or a specific landscape. In a social context, regions serve as the basis of identity for its people (BPB 2004)" (Meyer et al., 2004).

The integration of a multitude of functions reveals the multifunctional character of landscapes/regions. Landscape functions, (defined as the capacity of the landscape to perform inherent services for society, economy and economy), are recently intensively discussed in terms of ecosystem services (MEA, 2005; De Groot, 2006). Bastian (2002) has distinguished three groups of landscape functions: production functions (economic functions); regulation functions (ecological functions); and habitat functions (social

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functions). Hence, landscape functions and potentials characterize the capability and usability of a landscape (concerning human needs, demands, goals and goods and services) in a broad sense (Bastian & Röder, 2002). The assessment of landscape functions has to take account of the complex interrelations between land use and the environment.

2 Landscape assessment

Meyer & Grabaum (2008) worked out that the evaluation of nature is an integral part of the process of environmental planning, management and decision-making (de Groot, 1987, 1992, 2006). In recent decades it has reached global importance. At local and regional scale, "landscape assessment for the planning and decision making processes is a key issue pertaining to sustainable landscape management (Bastian and Schreiber, 1994; Meyer, 1997; Palmer and Lankhorst, 1998; Lee et al., 1999; Clay and Daniel, 2000; Meyer, 2001; Nakamae et al., 2001). The development and application of the analysis and assessment methods for the inclusion of human demands of a social, economic and ecological dimension is the object of complex landscape ecological and planning research" (Meyer & Grabaum, 2008).

As stated in Meyer et al. (2004) "the assessment of these impacts is an important part in preparation and evaluation of policy decisions and can relate to a number of different spatial and temporal scales and disciplinary dimensions. According to Müller & Volk (2001) or Jessel and Tobias (2002) landscape assessment requires the existence of a subject (to carry out the assessment) and an object (which is to be valued) that are related to each other (in this case the changing effect between society and nature). The assessment object (in this case landscape) is normally not recorded as a whole, but instead through the use of a model" (Meyer et al., 2004). Meyer et al. (2007b) worked out that "in the last decades a multitude of assessment methods for landscape functions have been developed (Marks et al. 1989, Bastian and Schreiber 1994). The methods include detailed expert knowledge and result in spatial explicit maps of landscape risks. GIS-based models are used to analyse and to asses the landscape functions. New opportunities are given by the combination of mathematical methods and spatial data. Assessments of landscape functions are suitable layers for land use decision making, especially if a number of different and opposing assessments of different functions are available for the same region. For opposing goals a method for the integration (e.g. for land use decisions) is essential" (Meyer et al., 2007b). The author has developed the framework on multicriteria landscape assessment and optimisation (MULBO). MULBO offers a powerful alternative to a structured integration process of landscape planning (Meyer and Grabaum 2003, Meyer 2006, Meyer & Grabaum 2008). The assessment methods integrated as tools in MULBO are validated for the application in landscape planning (Gruehn 2005).

Two-dimensional land use options are the major outcomes of MULBO. These land use scenarios are used by planners to discuss and to develop solutions in the planning process. Recently a method of the spatial distribution of linear landscape elements has been integrated in MULBO by using optimisation techniques (see section: optimal allocation of LLE). Thus semi-automatic assessment and modelling techniques have been developed to support landscape planning instruments and the localisation of linear landscape elements. Optimisation methods (in the mathematic sense) are applied to find maxima and minima of functions. MULBO uses compromise programming to calculate land use alternatives or best compromises which leads to best solutions for two or more variables. Hence, alternatives are developed to solve more than one problem.

3 Impact of linear landscape elements on landscape functions

Linear landscape elements (LLE) in cultural landscapes are originated from human land use. Hedgerows, buffer strips, balks, tree rows, roads, waters, ditches and other linear landscape elements are widespread in intensively used landscapes. Meyer et al. (2007a) demonstrate that "LLE have a high impact on landscape functions, biodiversity and the landscape balance or on the value of cultural landscapes as described e.g. by Ihse (1995). Kantelhardt et al. (2003) discusses in this context the correlation between agricultural site conditions and the density of hedgerow networks. More scientific knowledge is needed about the links between the spatial distributions of different land uses and the effects of different linear landscape elements on landscape functions" (Meyer et al., 2007a).

Concerning the potential impacts of LLE on landscape functions in planning applications it has to be stressed, that LLE are of growing importance regarding the acceptance by stakeholders and the control of landscape functions in cultural landscapes. In Table 1 an overview on the impact of LLE on different landscape functions are given. Assessment approaches or models are available for most of the functions. The table is based on the list of landscape functions published by Bastian & Röder (2003). The impact of LLE on functions of the 3rd order has been added to the table.

Table 1: Impact of LLE on landscape functions

Groups of functions	Impact of LLE
• functions of 1st order	
➢ functions of 2nd order	
o functions of 3rd order	
A - production (economic) functions	
• availability of renewable resources	
➢ production of biomass (suitability for cultivation)	
o arable fields (husbandry)	+
o permanent grassland	
o special crops (e.g. fruit-culture)	+
o wood (forestry)	
o game (hunting), fish (fishing, pisciculture)	+
➢ water accumulation	
o surface waters	+
o ground water	+
• availability of non-renewable resources	
o mineral raw materials, building materials	
o fossil fuels	
B - ecological functions	
• regulation of matter and energy flows	
➢ pedological functions (soil)	
o resistance to erosion/ to compaction	+
o resistance to underground wetness/ to drying out	+
o decomposition of harmful matters (filtering, buffering and transforming functions)	+
➢ hydrological functions (water)	
o groundwater recharge	+
o water storage/ run-off balance	+
o self-purifying power of surface waters	+
➢ meteorological functions (climate/ air)	
o temperature balance	+
o enhancing of atmospheric humidity	+
o influencing of wind	+
➢ regulation and regeneration of populations and communities (of plants and animals)	
o biotic reproduction and regeneration (self-renewal and maintenance) of biocoenoses	+
o regulation of organism populations (e.g. pests)	+
o conservation of the gene pools	+
C - social functions	
• psychological functions	
o aesthetical functions (scenery)	+
o ethical functions (gene pools, cultural heritage)	+
• information functions	
o functions for science and education	+
o (bio-) indication of environmental condition	+
• human-ecological functions	
o bioclimatological (-meteorological) effects	+
o filtering and buffering functions (chemical effects - soil/water/air)	+
o acoustic effects (noise control)	+
• functions of recreation (as a complex of psychological and human-ecological effects)	+

On the given background in the following article we will show in a brief form the following examples:

- How to find data about LLE by the example of the biotope type mappings and how to use the data for inventories (monitoring).
- How to integrate LLE into assessment procedures by the example of a habitat suitability assessment for the Corn Bunting (Emberiza calandra).
- How to find optimal distributions of LLE to enhance different landscape functions by using compromise optimisation.

4 Inventories, biotope type mapping and LLE

Monitoring data for LLE are normally not available in Germany or in other Central European Countries. Some information about LLE can be found in different official databases. Landscape monitoring is invented for "the control and (the) prognosis for the status and dynamics of natural-technical ecosystems, i.e. landscapes, and refers to landscape components such as vegetation cover, soil cover, land use and regional landscape structures" (Bastian and Schreiber, 1994, pp. 186). In the following, different approaches for landscape monitoring will be discussed in the context of LLE. The overview illustrated by Riedel and Lange (2001) concerning 26 on-going monitoring programmes for the Federal State of Schleswig-Holstein does not contain any indicators related. In Switzerland, the programme "Observation of Regions in Switzerland" (BUWAL, 1991, 1994, 2001) of the Swiss Federal Agency for Regional Planning is systematically recording changes in landscape structures. The changes in soil utilization and regional organization have been documented under the title "Landscape under Pressure" in up to now four observation periods since 1972. The programme has differentiated eight key characteristics of change (in the landscape), which serve as a quantitative and qualitative record. The programme includes information on small structures (e.g. tree rows, hedgerows), forest fringes and flowing waters.

In Austria, regional structures and also the land use structures and land use mosaics (which exemplify the character of a cultural landscape), were ascertained with the indicator set suggested by Wrbka et al. (1999). This study documents the environmental situation using a list of cultural landscape types. The technical design for a landscape-conservation-strategy for the German Federal State of Saxony also contains approaches for landscape monitoring (Döring et al. 1999). For the purpose of landcare, land use areas with landscape conservation relevance (field groves, hedgerows, marshland, meadows, etc.) are classified on a natural-regional basis, as determined according to the assessment of land use and selective biotope mapping for the German Federal State of Saxony.

In general, there is still a lack of information regarding LLE and small structures.

The "biotope type and land use map" for Saxony (here after referred to as "biotope type mapping") is based on a "List of biotope and land use types". The classification is based on the interpretation of colour infrared aerial photos from the years 1992/1993 in the scale of 1:10,000. The biotope type mapping contains both line and land use information. Line information entails details such as hedgerows, rows of trees and smaller running waters. Land use information entails arable land and grasslands as well as larger running waters and lakes. Furthermore, the biotope type mapping for Saxony is subdivided into seven levels. The data are hierarchically subordinated according to main group (for example waters), sub-group (such as running waters), resource/biotope type (such as cane brakes), characteristics (such as terraced silting-up vegetation) as well as other statements regarding utilization (developmental condition close to nature). Correspondingly, information at different levels of detail can be derived. Comparable data records also exist for most of the Federal German States. In the superordinated secondary level *Main Group* (MG), the land use area and line information are differentiated. The evaluation of biotope types in this study has been carried out for all main groups and for selected main groups in a more profound evaluation. *Waters* (MG 2), *tree groups, hedgerows and bushes* (MG 6) as well as *settlements, infrastructure,* and *green areas* (MG 9) are of primary interest. The classification scheme is based on the biotope and land use mapping of Saxony (SLUG, 1994).

A short example of a landscape inventory has been worked out by Meyer et al. (2003) and published recently by Meyer & Hirt (2006) and Hirt et al. (2007). The information of the biotope type mapping has been sorted and classified to natural regions and main soil types for the Mulde River Catchment in Saxony (with a size of 2,700 km²). Different data levels (biotope type mapping, soil map and map of natural regions) have been combined with GIS (Geographic Information System) for this study.

An example of a regional scale analysis shows that the main LLE are roads. Further common LLEs` are waters and tree rows (Fig. 1).

Further diffentiation shows that the length of all flowing waters is about 10 % of the total length of all LLE in the natural regions. They are not distributed evenly between the different natural regions.

In the northern natural regions (natural region 1 and 2) higher share of ditches can be observed compared to the hilly and mountain regions in the South of the catchment

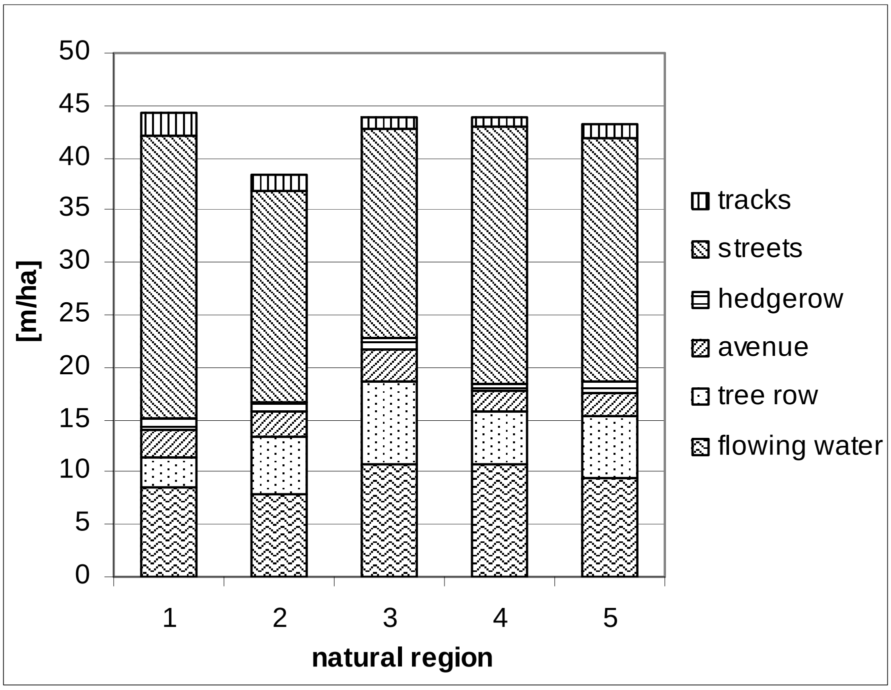

Fig. 1: LLE of 5 natural regions in the Mulde River Catchment in Saxony (in m/ha) (Hirt et al., 2007)

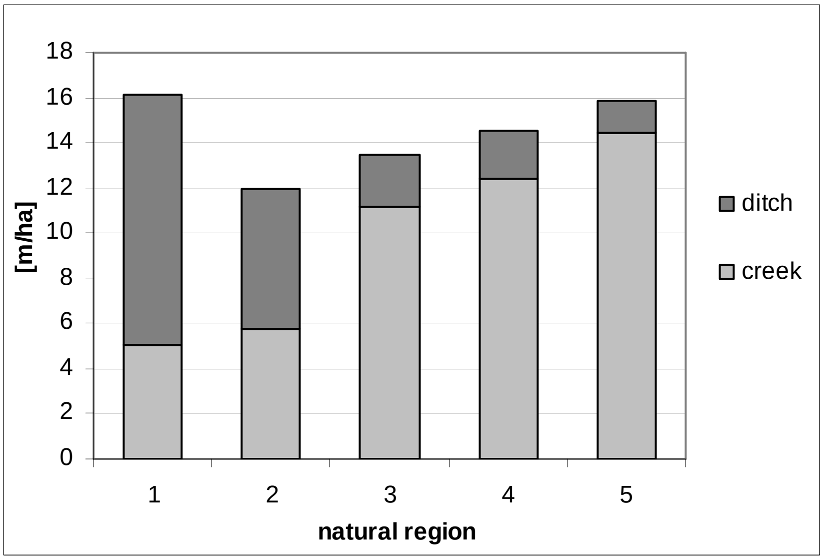

Fig. 2: Ditches and creeks differentiated for natural regions in the Mulde River Catchment (in m/ha) (Hirt et al., 2007)

(natural regions 3 to 5) (Fig. 2). This example stresses the regionally differentiated landscape transformation by man. This information leads to different activities on conservation and development planning.

5 Assessment of the impact of LLE

In general the assessment of LLE is applicable for all the landscape functions listed in Tab. 1. The assessment practice shows that most of the common assessment procedures and models, e.g. the ones described by Bastian & Schreiber (1994), are focused on the areal assessment of landscape functions. These models have not been developed to include the impact of LLE. Due to this fact adoptions or new developments of assessment procedures are needed. We have applied MULBO´s enhanced capabilities to assess the effect of different distributions of LLE in conservation management.

Based on a generalised habitat description the habitat preference of an avian species was assessed by means of landscape structural parameters. A static, rule-based spatially explicit non-probabilistic model was used to support decision-making and to find optimal LLE distribution. As described in detail by Meyer et al. (2007b) a predictive model has been developed. Using the Corn Bunting (Emberiza calandra) as an example the parameters necessary for an assessment of habitat characteristics of birds in Central Europe have been discussed in this paper on the basis of structural and context information. For this species a spatial analysis and assessment procedure supported by GIS has been developed. It was used to define rules and assessment categories and subsequently applied to the example of an open agricultural landscape in Saxony-Anhalt. The analysis and assessment model yielded good results even which scarce landscape-structural entry data. The model is well suited to support the process of the decision-making in spatial planning. The model framework presented in Meyer et al. (2007b) can be modified and applied to other species.

Figure 3 shows the strong correlation of LLE and the habitat requirements of the Corn Bunting, as key species in open arable landscapes. Hedgerows, tree lines, village fringes, unpaved country roads and field buffers offer different habitat opportunities for the Corn Bunting.

Comparable assessments have been worked out and adapted by the authors also for the assessment of the functional problems of wind erosion, water erosion, recreation, landscape retention, biotope connectivity and habitat suitability assessments for different species. The high value of LLE for most species in open arable landscapes should be stressed in general. The example of the Corn Bunting, as described above shows the high value of LLE for this single species. Similar results could be expected for other species.

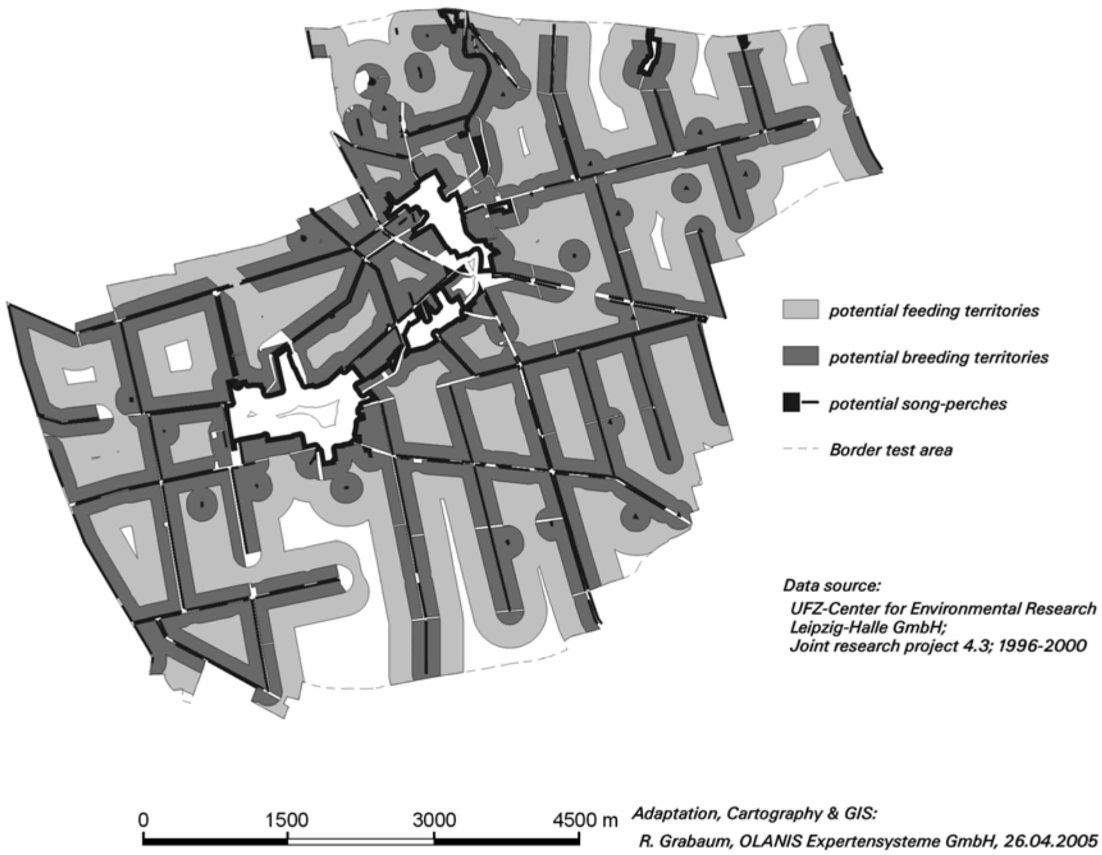

Fig. 3: Assessment of potential feeding and breeding territories and song-perches in the test area Barnstädt (Saxony-Anhalt) (Meyer et al. 2007b)

6 Optimal allocation of LLE

Grabaum et al. 2006 developed a framework to create and to optimise the distribution of LLE. On the basis of the MULBO framework (see Meyer 2006, Meyer & Grabaum 2008) a generation of a potential line grid has been developed and combined with assessment procedures. A GIS tool "Line Generator" has been developed to solve this type of problem by using the geometric concept of the compass rose to produce new line grids. The integration of existing LLE which could be found in landscape is possible.

On the basis of a new potential line grid assessments on the habitat suitability for the Corn Bunting and on the wind and the water erosion risks have been worked out. The procedure was aiming an integration of several landscape problems to be solved by a new distribution of LLE. The overall goal was to enhance the habitat quality for the Corn Bunting and to reduce the risk for wind and water erosion in an intensively used arable landscape by the localisation of a small amount of new LLE. The detailed opti-

Fig. 4: Assessment of wind erosion risk of new distributed LLE in the case study area Barnstädt as a result of an optimal compromise of 3 landscape functions on the data set "line grid set 500" m (Meyer et al 2007a).

misation framework and the technical application of the methods can be found in Meyer et al. (2007a).

The result of an optimisation run on the wind erosion risk is demonstrated in Figure 4. The location of new LLE (black lines) is distributed on places where positive effects clearly found for all the 3 functions integrated in the calculation. For the scenario runs only a small length of new lines has been chosen. This was applied to demonstrate the opportunities of the method. The need for the creation of new landscape elements can easily communicated to the relevant stakeholders' by discussing alternative optimisation scenarios. A comparable approach has been applied in the Project "IUMBO" funded by the DBU (Deutsche Bundesstiftung Umwelt) in the years of 2002-2006 (Mühle & Meyer 2005).

7 Progress

The structure of a landscape is highly relevant for research and planning. The fulfilling of the requirements of the Water Framework Directive (WFD) or the implementation of comprehensive catchment planning need information and techniques to solve complex landscape problems. Hence, LLE are of particular interest, as the need for the integration of different aspects into planning and into modelling has increased. By implementing the WFD in Germany, the restoration of LLE could be a valuable contribution for example to reduce nutrient inputs into rivers. But different other aspects on the evaluation and assessment of LLE and landscape functions should be taken into the focus of interest, too.

When using spatial explicit support systems for land use decision making the author estimates that today digital data of sufficient quality for spatial assessments are available. As the usage of GIS is nowadays a common method in several planning levels, a framework based on multiple landscape assessment tools is very useful for the development of different land use scenarios. By integrating the meaningful input of stakeholders and the knowledge of different sciences into the assessment procedures the framework MULBO offers scenario choices. The applied examples in agricultural landscapes explained in this paper, have shown that on the basis of diverse landscape scenarios farmers and institutional decision makers have the chance to start a transformation process for the landscape.

The implementation of the method into practice should be based on intensive discussions with land owners and stakeholders. Usually scenario results will be the basis of formal planning when focusing the length of the line elements e.g. on ownership and technical needs of land usage. There acceptance will be a step further towards the sustainable use of the land.

The method has a high potential to close methodological gaps when it is applied to the economic evaluation of ecosystems services. Presuming a monetary value of a landscape functional risk is available in terms of goods and services, the MULBO framework offers interesting ways for further research: First, assessments can be worked out on the field, farm or landscape scale, by integrating different aspects to evaluation and spatial decision making. Second, the problem solving by following the economic principle of optimisation and allocating funds into optimal compromises can be combined with planning needs of landscape planning instruments, Environmental Impact Assessment, Strategic Impact Assessment or Sustainability Impact Assessment.

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Examples for the Valuation of Ecosystem-based Tourism and Leisure Services with Choice Experiments

Jan Barkmann [1]

Presentation
selected slides

Overview

1. **The Choice Experiment Method**
2. **Case Studies**
   – sustainable tourism development options for Navarino Island (Chile)
   – preferences for nature- and landscape-based tourism in the domestic Chinese tourism market
   – sustainable development of Drakenberg hill from a leisure services perspective (Lower Saxony)
3. **Summary**

GEORG-AUGUST-UNIVERSITÄT
GÖTTINGEN

[1] Georg-August-Universität Göttingen; jbarkma@gwdg.de

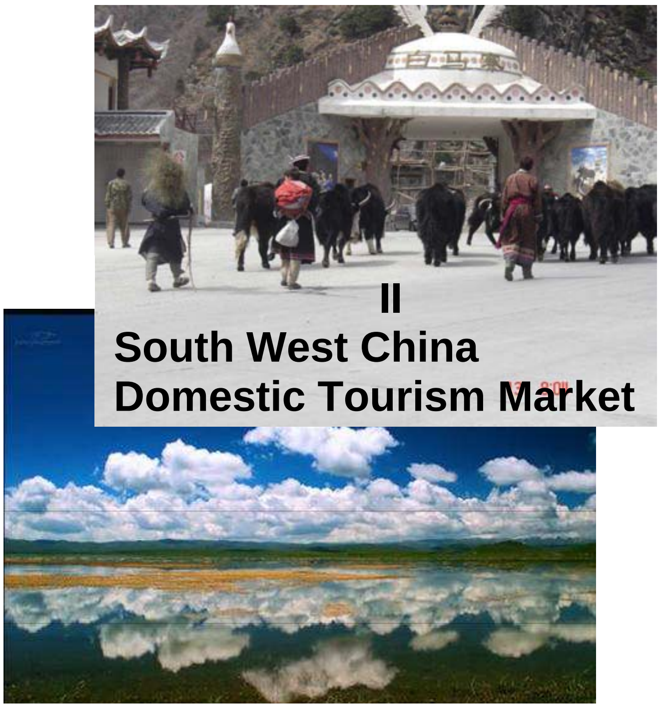
CE Methodology

- **Confront survey respondents with alternative choice options ('scenarios')**
  - different destination development scenarios/trips
- **Scenarios characterised by different attributes**
  - Restaurant selection (many, few, none)
  - Possibility to experience landscape (…)
  - Abundance of rare/beautiful species (…)
  - Tax increase associated with scenario/price of trip ($, $$, $$$)
- **Observe choices between scenarios**
- **Calculate influence of attribute on choices**
  - Influence relative to influence of tax change/price = maximum WTP for marginal unit of change

Case Studies

I
Navarino Island (Patagonia/Chile)

II
South West China Domestic Tourism Market



Case Study III: Drakenberge

Small-scale study on Peri-Urban Lei-sure Spot

Example of a Choice Card

	Alternative A	Alternative B	Present Situation
Attribute 1	good; 65% open	good; 65% open	good; 65% open
Attribute 2	no sheep / cattle (0)	lots of sheep / cattle (more than 500)	a few sheep / catle (about 50)
Attribute 3	for nature conservation and landscapecare	for nature conservation and landscapecare	no clearly determined aims
Attribute 4	many orchids and habitats	many orchids and habitats	few orchids and habitats
Cost Attribute	0 EUR	0 EUR	0 EUR
Choose your preferred alternaitve! ⇨	○	○	○

Put only one cross on this side!

Drakenberge - Results

Scenario Change	WTP / local resident / yr
1% more open land	1.03
1 additional sheep / cattle	-1.12
Conservation herding	30.28
"More" rare orchids	58.20

n=236, 2004; NL; adj. R^2=0.18

- **sheep/cattle has an inverted parabola utility function!**
- **WTP sufficient for re-introduction of sheep herding**
- **Better local environmental knowledge → higher WTP**

The Recreation Value of Forests

Volkmar Hartje [1]

Presentation
selected slides

Topics

- Reasons for recreation in forests
- Reasons for calculating recreation benefits
- Values and visitors
- Methods for estimating monetary values
- Methods for estimating recreation consumption
- Results from German speaking countries
- Research perspectives

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Reasons for Recreation in Forests

General nature experience dominates

quietness, fresh air, naturalness as main motives,
main activity: walking and nature watching

Specific activities secondary

biking, jogging, pick-nicking, collecting forest fruits

Specific attributes of forested ecosystems

tree composition, landscape structure: in German surveys for recreation secondary; but systematic treatment in economic valuation possible

Recreation infrastructure

in German surveys of lower importance;
again no systematic treatment in German studies

Reasons for Calculating Recreation benefits

- **General economic value of forests**
  recreational use as one function in multifunctional forests
  (compared to wood based revenues): To generate good will for the sector
- **Economic analysis of sectoral policies**;
  reasoning for afforestation, for change of forest structure; organisational structure
- **Economic analysis of national biodiversity strategy** (a component in multifunctional use)
- **Management strategies for sites / forests**
  Pricing strategies for visitors; In Germany unrestricted and free access
- **Basis for calculating subsidies**
  in Germany cost of recreation to forest owners reasoning for limited subsidies to sector
- **Basis for compensating eco-damage**
  US practice if damage results in use restriction
  based on corresponding EU Directive unlikely

Values and visitors

Central assumption: separable from other benefits

Recreational benefits are mostly use dependent

for o-nsite approaches use focus imminent; but for non-users significant value for option of use

Two numbers are important:

- **Monetary value of the visit:**
- = the core challenge of economic valuation; focus of most research
- **numbers of visitors/visits**:
- equally important, but usually less the subject of scrutiny
- often the subject of surveys from different disciplines

Central for a site specific demand function

Basis for an aggregate estimate

Methods for estimating monetary values

Travel costs methods:

3 basic types

- early phase: zonal models
- second phase: based on individual data
- third phase: choice between multiple sites: Random Utility models

Advantages

- observable data (limited as basis are surveys for individual data, thus reported behavior) →intuitively appealing (thus, non controversial)

Problems

- measure of opportunity costs of time
- applicability for planned sites
- Definition of market size

Methods for estimating monetary values

Contingent valuation study

- relies on stated preferences
- survey based
- relies on a constructed (not a real) market situation

Asks for maximum contribution

Advantages

- usable for all sites, not only for current user also for future users (option value of recreation)

Problems

- hypothetical situation

Methods for estimating monetary values

Choice experiments

Newer development in CVM:

Focuses on the valuation of individual attributes of the environmental resource concerned instead of an all - encompassing change of quality

For forest recreation: attributes of forests, supply of recreational infrastructure (trails, sites, asf.)

Respondents are asked to choose between alternatives with different attributes, their level and a corresponding price

Respondents are faced with choice sets

Relevance of choice sets tested in focus groups

Statistical analysis to determine choice probability

Methods for estimating monetary values

Choice set for biodiversity valuation

Tab. 2-6: Description of Biodiversity before and after the "LÖWE" - program

Attribut	Lüneburger Heide		Solling/Harz	
	Status quo	LÖWE	Status quo	LÖWE
Share of deciduous forest	30 %	60 %	40 %	70 %
Habitats for endangered and protected species	mittel	hoch	mittel	hoch
Species diversity	mittel	mittel	hoch	mittel
Age compositon of trees	niedrig	hoch	mittel	hoch
Landscape diversity	niedrig	mittel	mittel	hoch

Measuring /estimating recreation consumption

Alternative measures of visitor use

Recreation day, recreation visitor day, recreation visit, entrance permits, participation rates

Measurement on site

- Counting visitors at sites (entrances/gates), ticket sales, counting devices; problem: non-entrance visitors
- Counting cars in forest recreation parking lots; problem: occupancy rates
- On-site Monitoring: sampling/ surveying

Off-site measurement

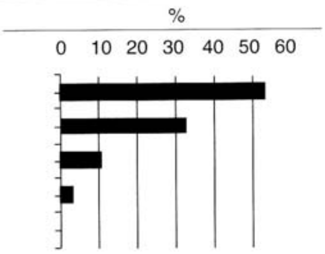
surveys, frequency of recreation, participation in activity

Visitor estimates in Germany

Frequency of visits

Table 8.23. Frequency of forest visits. (Question 2.2: How often did you visit a forest area in Germany or in a neighbouring state in your leisure time during the last 12 months (approximately)?)

Frequency of forest visits	*n*	% w.
Up to 10 times a year (< monthly)	357	53.6
10–50 times a year (< weekly)	217	33.0
More than 50 times a year (> weekly)	72	10.3
Don't know	17	3.0
Missing	1	0.1
Total	664	100.0



% w., weight %.

Table 8.24. Results from six regional investigations on forest visit frequency.

Source	Number of respondents	Frequency of forest visits (%) Seldom or never	At least once a year	At least once a month	At least once a week	Missing
Dertz and Nießlein (1993), Germany	1777	22	28	48		2
Sample-Institut (1995), Germany	1414	33	28	28	9	2
RES (1998), Germany	1069	35	33	20	7	5
Rozsnyay (1972b), Bremen	(?)	24	25	38	13	
Elsasser (1994), Hamburg	806	20	23	29	28	
Braune (1998), Lübeck	100	34		31	35	

Results in German speaking countries

Study	Schelbert 1988	Bergen Löwen-stein 1992	Löwen-stein 1994	Elsasser 1996	Baur et al. 2003
Method	CVM, TC	TC	CVM, TC	CVM, TC	CVM
Forest type	Urban / daily visitors	tourism region	tourism region	Urban/ tourism region	urban
Results CVM	3,30 CHF visit 430 CHF/y/ visitor		4,60 DM /day	114 DM/y /visitor: urban 100DM/seas/ visitor: tourism	15-100 CHF/y/user type
Results TC	TC 3,50 CHF/ visit	6,24-7,99 DM/ Day/ Visitor	2,30-8,80 DM /visit	1-8 DM/visit urban 1,10-13DM visit tourism	

Results in UK (Jones survey 2003)

Table 2: Studies of open-access woodland recreation value in Great Britain.

Value type	Recreation value unit	Valuation method	No. of studies	Date conducted[1]	No. of value estimates	Value range (£, 1990) (m = million)
Use	Per person per visit.	CV	8[a]	1987 – 1993	28	£ 0.28 - £ 1.55
Use + option	Per person per visit.	CV	3[b]	1988 – 1992	16	£ 0.51 - £ 1.46
Use	Per person per visit.	ZTC	3[c]	1976 – 1988	17	£ 1.30 - £ 3.91
Use	Per person per visit.	ITC	3[d]	1988 – 1993	16	£ 0.07 - £ 2.74
Use	Per person per year	CV	3[e]	1989 – 1992	7	£ 5.14 - £ 29.59
Use	Per household capital[2]	CV	3[f]	1990	3	£ 3.27[3] - £ 12.89
Use	FC forests/ conservancy[4]	TC	1[g]	1970	13	£0.1m - £1.1m
Use	Total UK value	TC	6[h]	1970 – 1998	6	£6.5m - £62.5m
-	All studies	-	30	1970 - 1998	106	-

Notes:
1 = Dates refer to the year of study survey rather than publication date.
2 = These studies use a once-and-for-all willingness to pay per household question.
3 = We have recalculated this figure by including those who refused to pay as zero bids.
4 = The FC at the time divided the area of Great Britain into a number of Forest Conservancies and large forests to which these estimates relate.
Study references:

Estimates of total value on a national level

Transformation of individual wtp for forest visits on all national forest visitors

CH Baur Ott 2005

Travel cost estimate of benefits of national average forest recreation: 544CHF/p/y (29 CHF/visit)

Swiss population above 18y: 5,8 Mio

→ 3,2 bio. CHF/y

D Elsasser 2001

benefit transfer of 12 sites from 3 (western) regions, CVM based

Yielding annual value between 100 and 130 DM/p/y

daily visitor only (64% of population)

→ total annual wtp 4,4 and 5,7 bio. DM (= consumer surplus)

1 € = 1,95583 DM

Rough regional WTP for forest recreation on ha basis

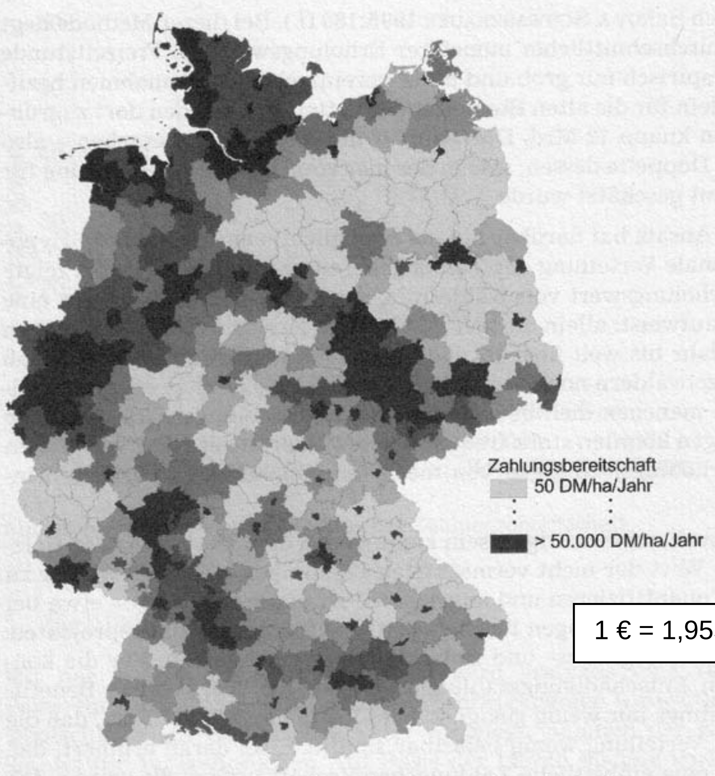
county based regional estimates of hectare values (rough first approximation):

- uses of average wtp of 117 DM/y
- estimates forest users on county basis (visitation probability); model weakest part in calculation
- multiplies by population (above 14y)
- divides by forested area

→ High variation between 50 and 50.000 DM/ha

→ High values in metropolitan areas with low share of forests

Regional WTP for forest recreation based on usage rate DM/ha



1 € = 1,95583 DM

Abbildung 2: Regionale Verteilung der Zahlungsbereitschaft

Research perspectives for Germany

Status of economic valuation of forest recreation in Germany:
•Few on-site surveys, few recent,
•few studies in north-eastern Germany
•So far no use of RUM, Choice experiments
•Studies mostly academic, Ph.D.
•First attempts at networking;
•Role of public agencies minor
Status of visitor monitoring
•Few studies: 3 studies 90s
•Few information in national forest report
•At selected sites by state forest agencies
Strategy:
Improve, widen economic valuation at individual sites; include substitution effects
Systematize visitor monitoring, develop visitor monitoring modelling

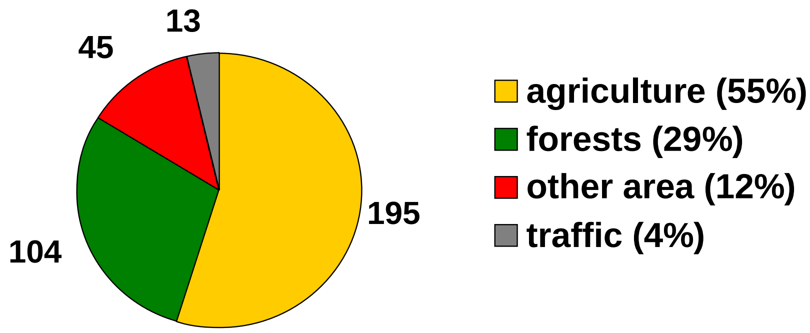
Ecosystem Services with regard to Hunting Activities in Germany

Ludger Wenzelides [1]

Presentation
selected slides

Hunting in Germany – Overview

Area of Germany: around 357.000 km²



Hunting area in Germany: around 320.000 km²

ca. 70.000 hunting districts (Reviere)

Hunters in Germany: around 348.000 (in 2006)

Universität Trier

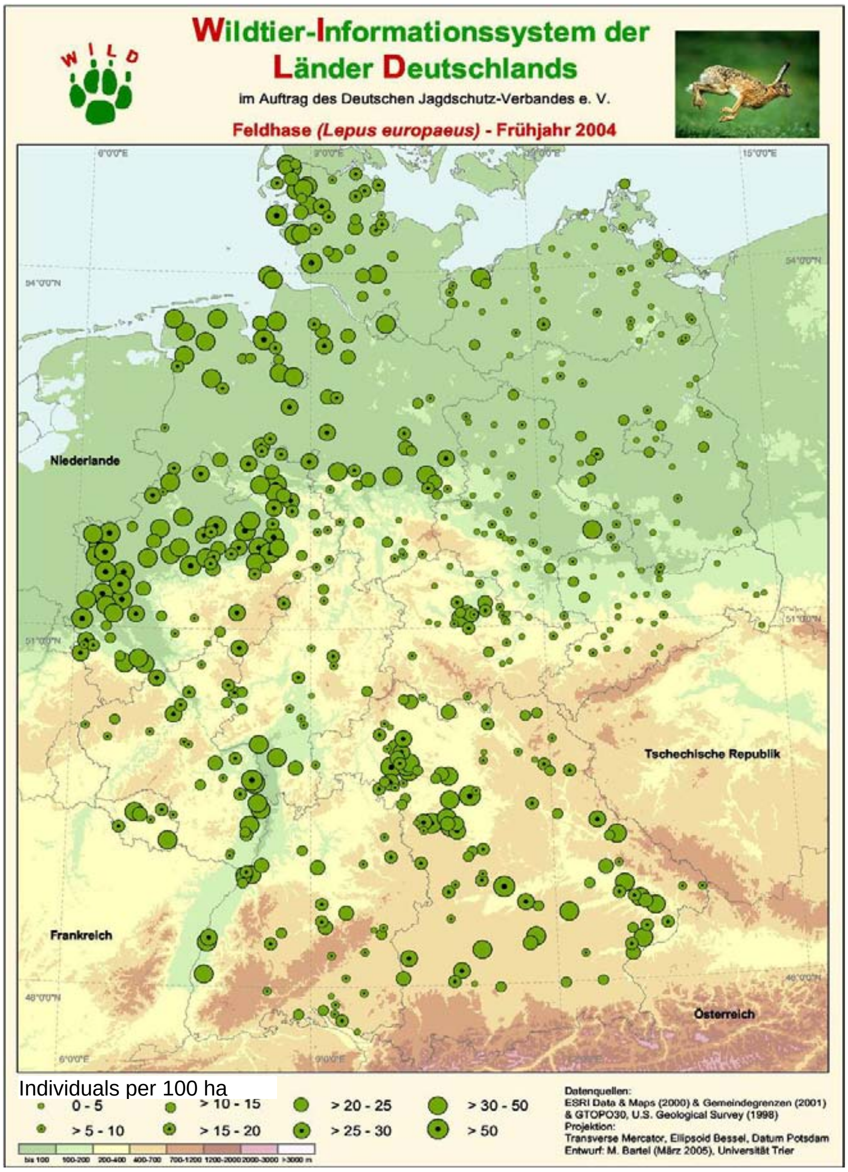
[1] University of Trier – Biogeography

WILD – Wildlife Information System of Germany

Since 2001

- colleting data countrywide
  - organized on the level of federal states
  - more than 700 reference areas (spring 2006)
- abundance of five animal species in these reference areas
  - European hare
  - Fox
  - Badger
  - Partridge
  - Carrion Crow

Abundance of european hare

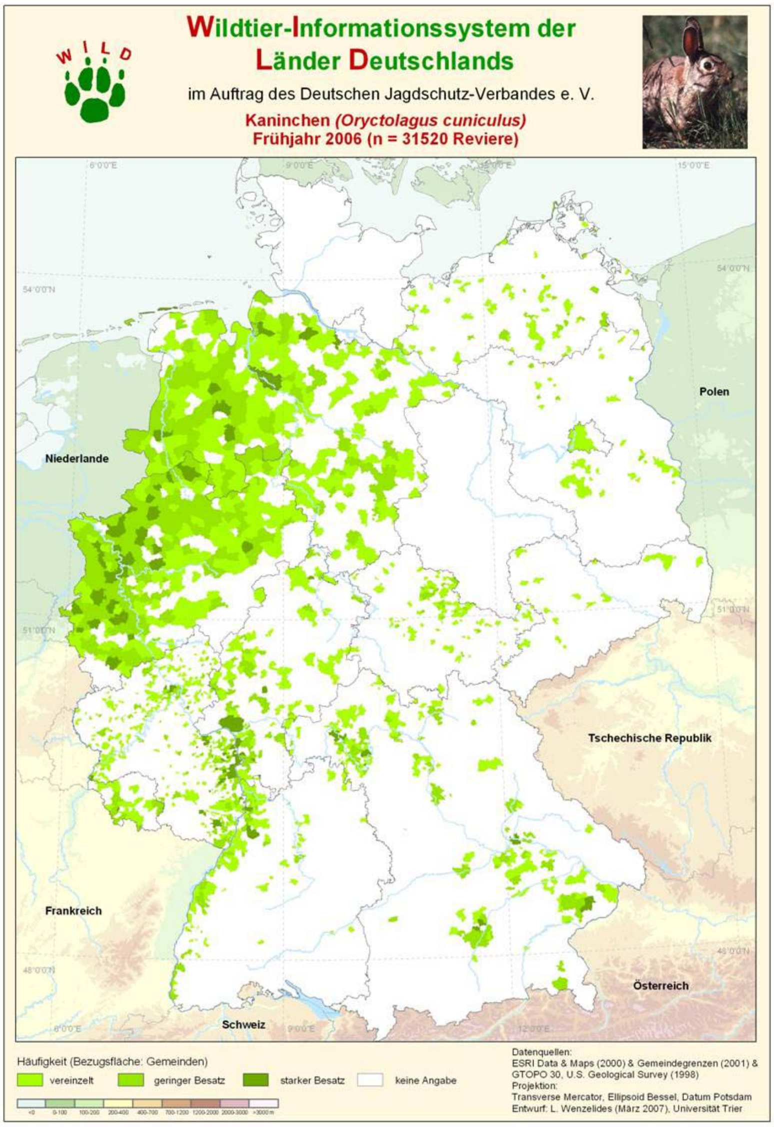


WILD – Wildlife Information System of Germany

Assessment of animal species occurrence in Germany

- spring 2006
- 30.000 questionnaires returned (roughly 50%)
- 26 species (more in Bavaria)
- list is a compromise
- includes Neozoa and animals reoccupying former habitats
  - European Otter
  - Racoon Dog
  - Nutria or Coypu
  - European Beaver
- mostly just occurrence yes/no
  - for some species more detailed

Occurrence of rabbit

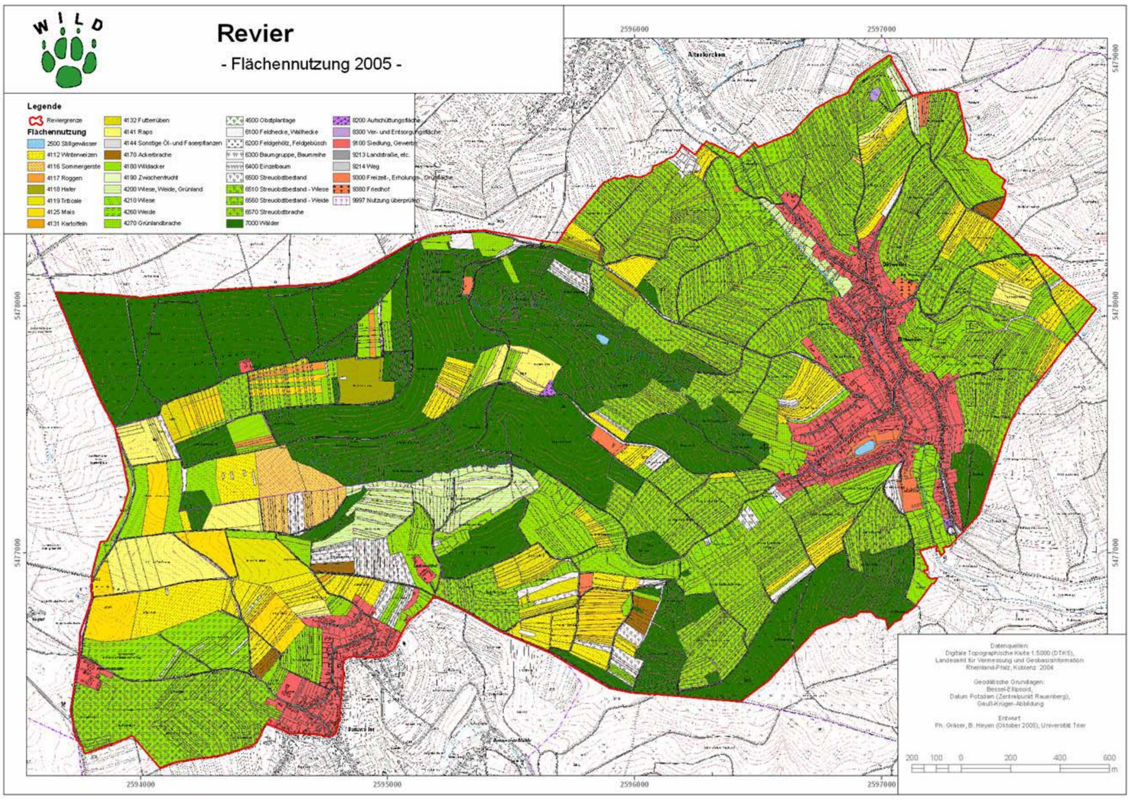


sporadic
low spread
high spread
no information

WILD – Wildlife Information System of Germany

Land use analysis

- representation of the land cover of the state
- GIS-mapping of the entire hunting district (digitalisation)
- landscape measurements derived with models (within ArcGIS)
  - landscape factors to characterize the hunting district
- answer tree analysis to identify important habitat factors
  - will also take predation into account



Ecosystem Services

- **Estimates of financial costs of hunting in Germany (2006)**

Hunting licenses	18.5 Mill €
Hunting insurance	16.0 Mill €
Lease for hunting district	**366.2 Mill €**
Hunting tax	65.0 Mill €
Activities related to habitat “improvements”	98.0 Mill €
Hunting safety	32.0 Mill €
Game damage compensation	61.0 Mill €
Hunting equipment, education	62.0 Mill €
Dog keeping activities	36.0 Mill €
Sum	**754.7 Mill €**

estimates from DJV 2007

- **Meat from game animals**
  - around 36.000 t in 2006
  - **worth estimated 182,5 Mill € (DJV 2007)**

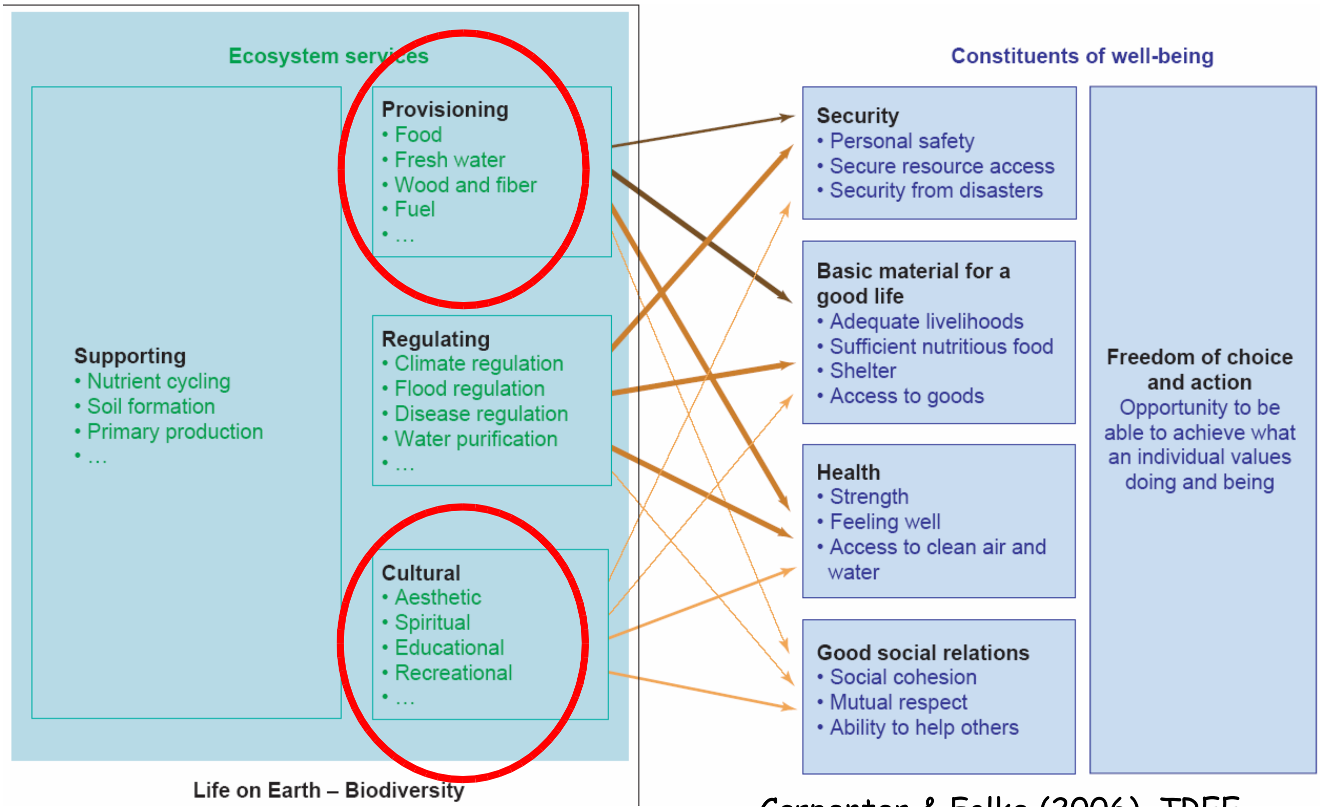
- **Amount of lease for hunting district depends on animal species presence**
  - **willingness to pay higher rates increases with species presence**
  - **more natural habitats will have more animal species present**

The Social, Economic and Ecological Importance of Recreational Fishing in Germany

Robert Arlinghaus [1]

Presentation
selected slides

Angling And Ecological Services Generated by Aquatic Ecosystems and Biodiversity

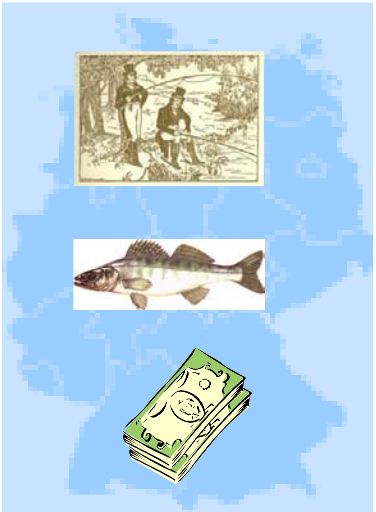


Carpenter & Folke (2006), TREE

[1] Junior-Professor at Humboldt-University of Berlin, Faculty of Agriculture and Horticulture, Institute of Animal Sciences, Research Group Inland Fisheries Management and Leibniz-Institute of Freshwater Ecology and Inland Fisheries (IGB), Department of Biology and Ecology of Fishes, ADAPTFISH Project Berlin, Germany, contact: arlinghaus@igb-berlin.de

Socio-Economic Importance of Recreational Fisheries

Recreational Fisheries	Commercial River and Lake Fisheries
> 400 000 ha	250.000 ha
2.6 – 4.1 million people (5%)	
Yield 45,000 t	Yield c. 4.000 t
6.4 billion €	11.7 million (caution)
52,000 jobs	1,875 jobs



sources: Hilge (1998); Arlinghaus (2004); Brämick (2004); Binnenfischereierhebung (2004)

Economic Impacts

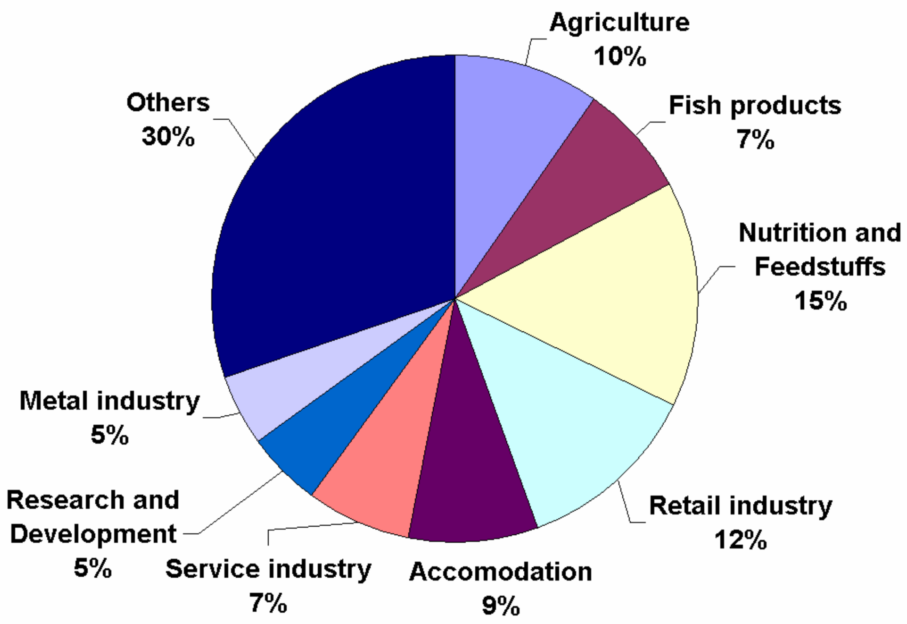
Often wrongly equalled with total economic benefits; for regions particularly relevant, if new money comes into the region/nation, e.g. by angling tourism

Impacts

Direct	Indirect	Induced
Expenses by anglers (Mean: 920 € per angler and year)	Expenses by industries receiving angler expenses (multiplier 1.8)	Expenses by employees of industries receiving angler expenses

Total economic impacts: 5,248 billion € / year; 52,000 jobs directly or indirectly dependent on expenses by anglers (input/output-analysis with DIW, Berlin)

Distribution of Job Effects Induced by Angling Across Industries
(Arlinghaus 2004, Berichte des IGB)



Economic Values

Net willingness to pay or consumer surplus; important for cost benefit analyses of environmental improvements

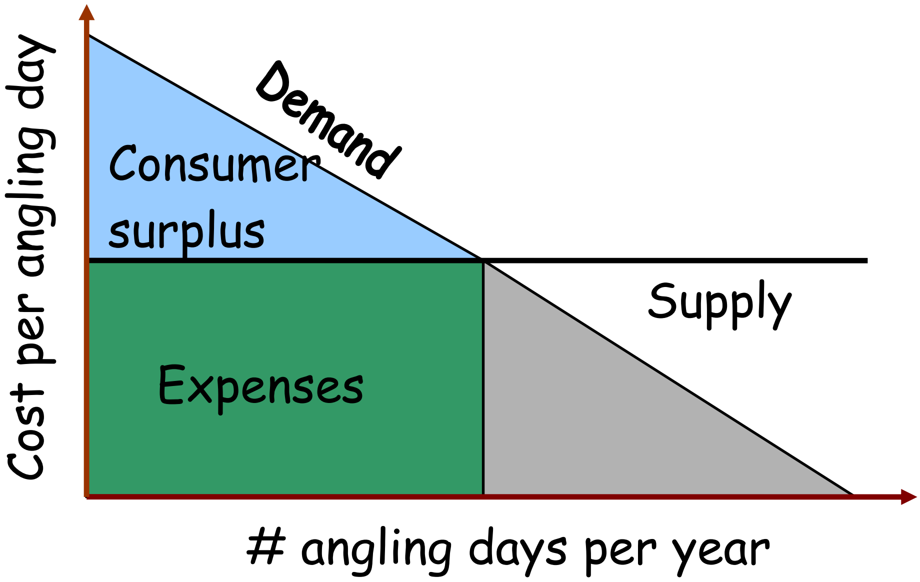
<u>Use</u>

- Consumptive (e.g., yield)
- Non-consumptive (e.g. nature experience)
- Indirect (e.g., reading about fishing)

<u>Non-Use</u>

- Option (personal use in the future)
- Bequest (use by other people in future)
- Existence (pure existence is valued)

Total Economic Benefits



⇒ contingent valuation, travel cost models or discrete choice models to estimate consumer surplus (maximum willingness to pay over and above current expenses)

Total Economic Benefits in German Recreational Fishing 2002 (Arlinghaus 2004, Berichte IGB)

	Non-use (population)	Use (anglers)
Mean per person (€/yr.)	21	134
Finite population (Mio.)	70.418	3.365
Total benefits (Mio. €/yr.)	937	296

Summary and Conclusions

- Recreational fishing in all industrialized societies, including Germany, most important fisheries form in freshwater ecosystems and many coastal ones
- Myriad of economic, social and ecological impacts and values, hence benefits, to society that are directly linked to a consumptive interaction with fish stocks
- These benefits are often overlooked, but need to be integrated in aquatic ecosystem management
- Potential negative impacts also possible, conflicts among stakeholders the rule, lack of empathy
- Tools to do so are available and have successfully been applied internationally (e.g., ecological cost-benefit analysis of habitat enhancement activities, many projects become positive if recreational values are explicitly considered, Willis & Garrod 1999)

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Economic Valuation of Ecosystem Services of Urban Open Spaces – Contribution of Urban Green to Life Quality in European Cities

Dietwald Gruehn [1],

1 Introduction

The paper title implies a connection between ecosystem services and urban open spaces on the one hand and a connection of ecosystem services of urban open spaces and life quality on the other hand. That leads to the following questions:

- What kind of ecosystem services respectively landscape functions are being provided by urban open spaces?
- Which ecosystem services or landscape functions of urban open spaces may contribute to life quality?

Due to the fact that both questions are linked with urban open spaces and urban open spaces can be very diverse with respect to their location, spatial extension and context, vegetation and infrastructure features as well as their state of maintenance, the contribution of urban open spaces to ecosystem services and life quality to a great extent may depend on the conditions in particular cases. At best the following ecosystem services are being provided by urban open spaces (LUTHER, GRUEHN & KENNEWEG 2002):

- Habitat functions
- Bio-climatic functions
- Air regeneration
- Groundwater recharge
- Water retention
- Noise attenuation
- Design respectively visual quality of city scapes
- Recreation function.

In contrast the (potential) effect of urban open spaces on human health and wellbeing is not terminatory investigated, yet. Under the perspective of life quality the above mentioned ecosystem services are not generally accepted as relevant. Even if life

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quality undoubtedly is dependent of external socio-economic factors, e.g. income, self-realisation, a considerable relevance may be assigned to ecosystem services of urban open spaces as listed below:

- Bio-climatic functions
- Air regeneration
- Noise attenuation
- Design respectively visual quality of city scapes
- Recreation function.

The contribution of urban open spaces to life quality is strongly connected with the spatial dimension. Therefore it may also be relevant for the spatial pattern of plots with its large variety of land values.

2 Background

In many European countries a contradictory debate on the value of and the benefits from urban open spaces is going on. On the one hand, they are highly regarded not only due to social, ecological and health related matters, but also increasingly to economic benefits resulting from them. Above all, they are seen as a means to counter migration out of cities and, thus, the deterioration of wide parts of the city centres. On the outskirts, their importance for the attractiveness of areas of complex housing has recently been recognised, at least in Germany. Finally, the provision and the quality of urban open spaces increasingly represent an important soft location factor for cities particularly with regard to attracting companies to settle down (FLL, 1999; LUTHER 2000).

On the other hand, however, recently less attention has been paid to urban open spaces in every day political and administrative life. This has led to a severe neglect of these amenities. As a consequence, they do not only to a great extent lose their positive effects on the environment in terms of ecosystem services, but might also actually be partly responsible for an accelerating decay of certain parts of cities because unkempt, littered and run down parks are often associated with crime and under-privileged neighbourhoods by the public (MAHLER 1998, URBAN PARK FORUM 1999).

Consequently, for a rational debate on the importance of (urban) open spaces and their economic effects it is important to find out more about which kinds of open spaces are appreciated by the public and what economic benefits actually result from them.

3 Methodology and Data

Because the provision of green areas is a public good, their appreciation is difficult to measure as market prices are not available. That is why special methods must be employed to obtain benefit information. Basically there are two methodological approaches for estimating economic values of non market goods: stated preferences and revealed preferences. The former one finds out individual preferences by asking people directly regarding their willingness to pay for certain goods (contingent valuation). The latter approach is to analyse the relationship between private market goods and public goods and to conclude the value of the public good from the price of the private good. That means the benefits resulting from public goods are reflected in private ones.

This method is basically adopted within the research projects presented here: Statistical analyses of land value data from European Cities based on random samples reveal new information regarding economic benefits of open spaces (LUTHER 2000; LUTHER & GRUEHN 2001; LUTHER & GRUEHN 2002; LUTHER, GRUEHN & KENNEWEG 2002; GRUEHN 2004, GRUEHN 2006).

The aim of our research is

- to discover the relationship between land value and the provision and quality of open spaces as well as
- to verify the value-increasing effects of urban open spaces by means of statistical methods.

The central hypothesis is:

- Open spaces or open space related criteria respectively have a positive influence on land value.

Generally land values are characterized by a large variation according to certain location factors. Due to the experience of practitioners in the field of real estate assessment, the factor 'centrality' seems to explain the greatest part of the total variation of land values. Despite of that empirical studies reveal that a central location does not explain more than 50 or 70 % of the total variation of land values. Hence, there is every reason to believe that location factors are not limited on the aspect of less or more central locations. It makes sense to prove other soft location factors, for instance open space related criteria.

Data acquisition comprises two types of data, dependent variables (e.g. land values) and independent variables (open space related variables referring to street sections or open spaces in the vicinity of the examined plots). Additionally control variables (urban density, urban development restrictions or allowances) were included to eliminate interpretation errors (Figure 1).

The following list gives some examples of open space related criteria:

- Distance (of the sample elements) to neighbourhood open spaces,
- Number of listed gardens within a radius of 500 m,
- Lack of local city parks in specific urban environments,
- Vicinity to open spaces and inshore waters,
- Number and size of nature reserve areas or areas of great natural value in the vicinity of the sample,
- Visual street quality,
- Urban fabric.

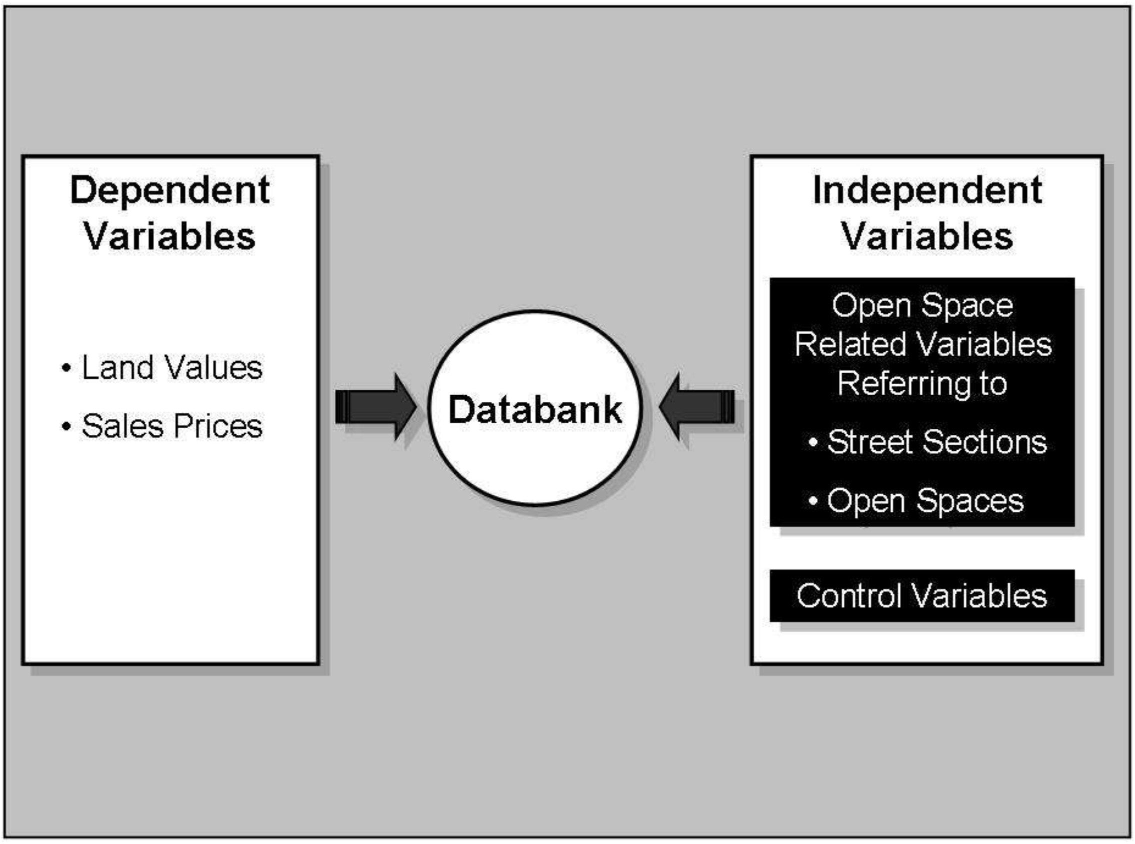

Figure 1: Data Acquisition

Hypotheses were formulated which stated that each of the above mentioned criteria has a positive effect upon land value. Whether or not the hypotheses were correct was determined through the application of specific statistical tests on the data collected. A confidence level of at least 95 % (i. e. a p-value ≤ significance level α ($\alpha = 0.05$)) is necessary to accept a hypothesis. If accepted under this condition, there is a significant connection between the sample and the population from which the sample was drawn. In other words, there is a significant effect upon land value of the criteria examined.

The ANOVA (analysis of variance) was the main tool for the statistical analysis. It is a method which helps to detect the effect of certain variables on one or several other variables (BACKHAUS et al. 1996). Its purpose within the survey was to examine if there is a statistically significant effect of the independent variables (location criteria) on the dependent variable (land value). If so, additional statements regarding the level of the influence can be derived. Thus, it is possible to distinguish and to weigh the individual location criteria with respect to their importance as regards land value.

In addition to the ANOVA, other test statistics were applied, depending on the quality of data available and on the specific questions to be researched: the t-Test and nonparametric procedures, like e. g. the KRUSKAL-WALLIS-H-Test and the U-Test of MANN and WHITNEY.

There is data available at University of Dortmund from about 20 European cities, mostly German cities (LUTHER 2000; GRUEHN 2004, GRUEHN 2006). This paper mainly focuses on data from Berlin (Germany) and Malmoe (Sweden).

4 Results

Below the most striking results will be summarised. The analysis of the data collected makes it plain that many of the criteria examined actually have a positive effect upon land value. This means that a higher quantity and/or a better quality of open spaces (e. g. small distance to parks or the existence of street trees etc.) results in higher land prices.

The data analysis revealed that the extent of the sample land values is unambiguously dependent on the distance of the statistical blocks to their next neighbourhood open space. Comparing the means of the different categories, it can be shown that land prices decrease with a growing distance to this open space category (Figure 2). The mean value of the blocks with a distance less than 400 m is about 170 €/m² higher than the mean land value of blocks with a distance more than 400 m (Figure 2). The mean difference is significant to a 99.9 % confidence level (t-Test).

The data analysis revealed that locations with a high density of listed gardens have significantly higher land prices than those with a lower density (Figure 3). If there are at least two listed gardens within the radius examined, then the plots are on average about 370 €/m² more expensive than those without such a feature. The variable

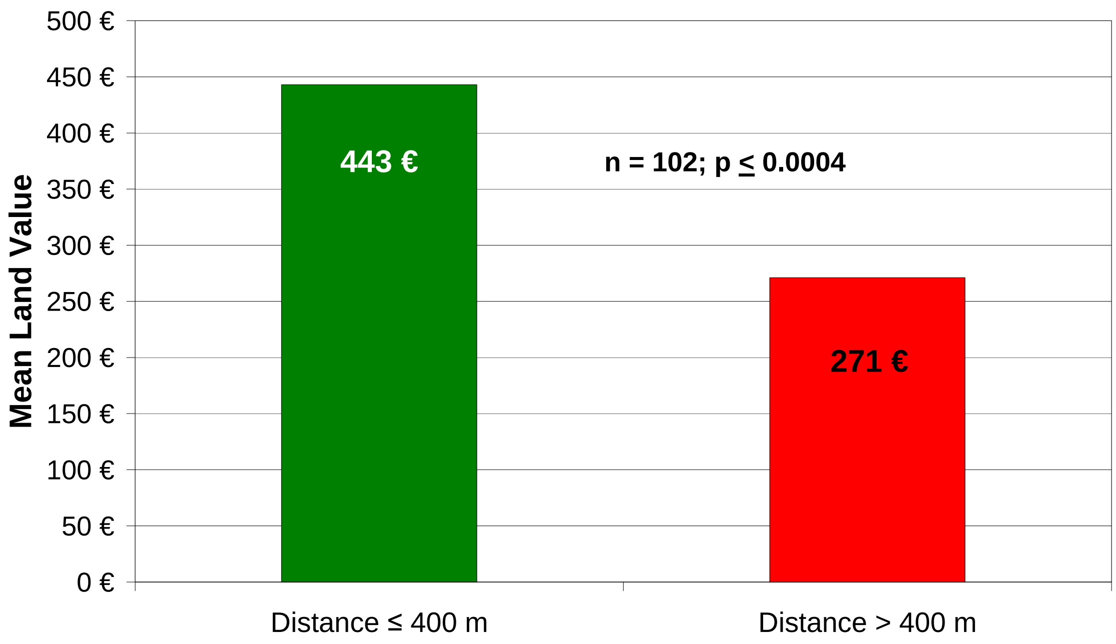

Figure 2: Effect of the Distance to Neighbourhood Open Spaces on the Mean Land Value in Berlin (t-test)

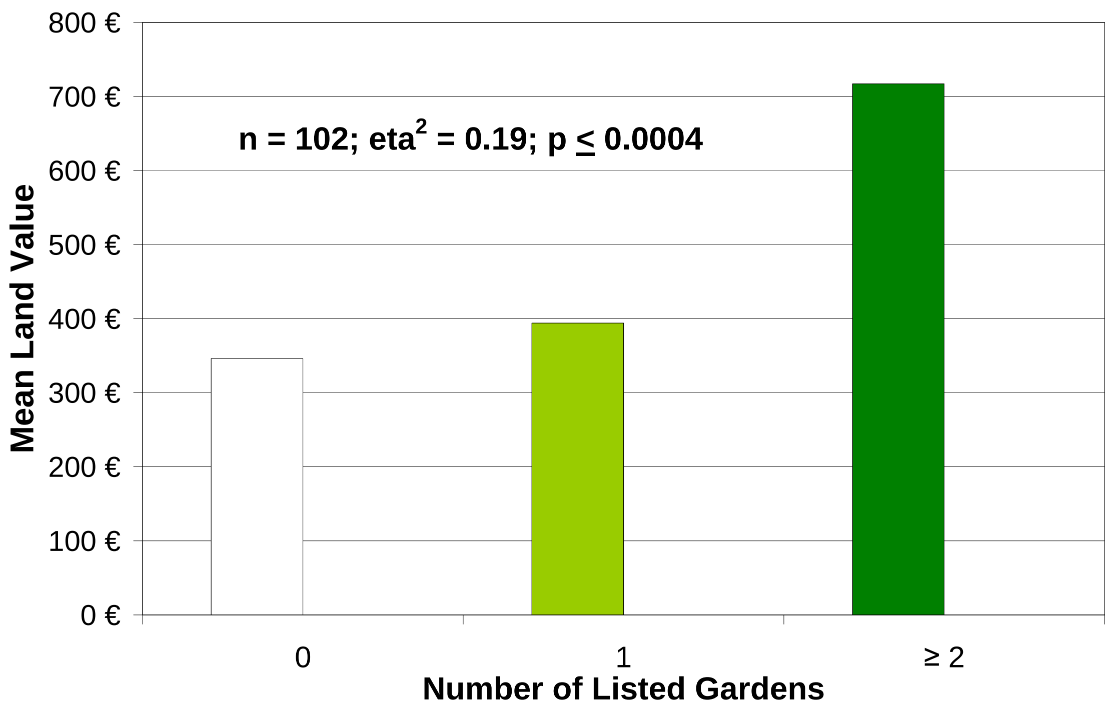

Figure 3: Effect of Listed Gardens within a Radius of 500 m on the Mean Land Value in Berlin (ANOVA)

"Number of listed gardens within a radius of 500 m" has an effect of 18.9 %, i. e. the variation of land prices is nearly 20 % dependent of this factor.

Figure 4 points out the effect of local city parks and missing of local city parks respectively, on the land value on residential quarters in Malmoe (Sweden). The data is valid for mixed development zones in the inner city, which are characterized by enclosed block development. Missing of local parks is linked with a significant negative impact on the mean land values in both cases, apartment houses and offices. The absence of local parks explains 13 % or 22 % respectively, of the total variation of land values in mixed development zones in the inner city of Malmoe. The effect is significant to a 99.9 % confidence level.

Figure 4: Effect of Missing of Local City Parks on the Mean Land Value in Residential Quarters in Malmoe/Sweden (ANOVA)

Figure 5 contains a synoptic presentation of the effect of different open space related criteria on land values within a radius of 500 m in randomly selected plots in German cities. With reference to the eta-square values as indicators of the level of influence "state of maintenance" proves to be the most important factor of the investigated variables. It explains nearly 20 % of the total land value variation. Figure 5 reveals that the intensity of design of urban open spaces is of similar importance for the land value in German cities, whereas the variables "accessibility", "ratio of flower borders" as well as “visual quality” have a less strong, but also significant effect (p-value ≤ 0.0004). The correspondent eta-square values range from 3.2 % to 6.4 %.

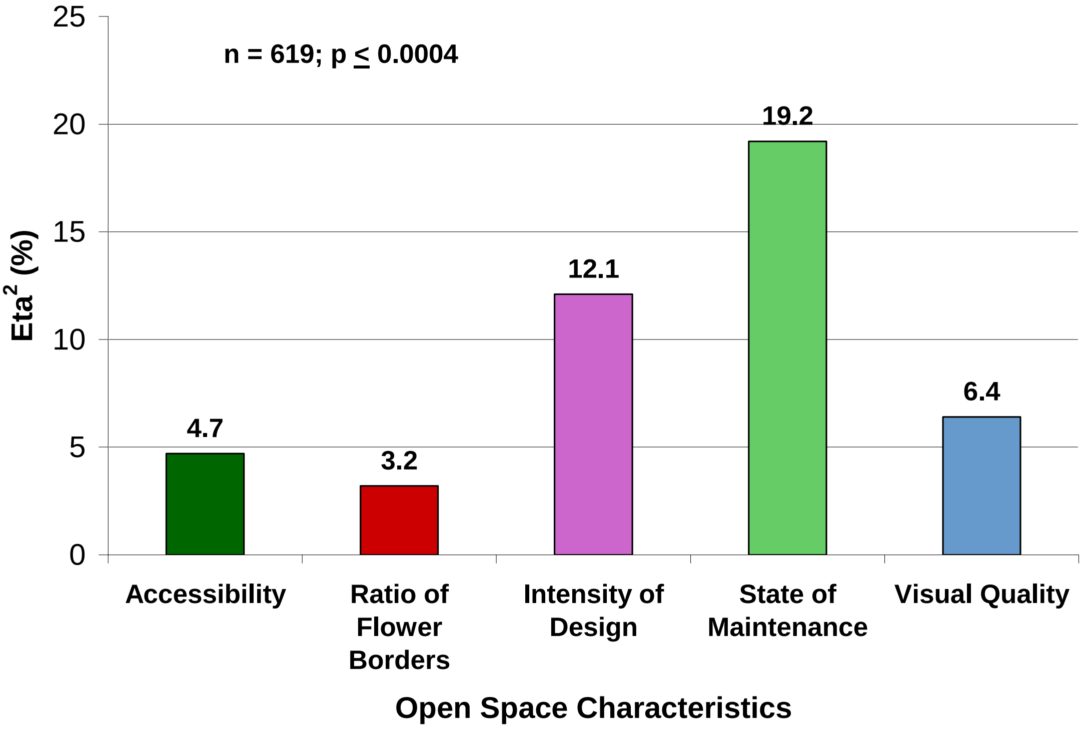

Figure 5: Effect of Open Space related Variables on Land Value in German Cities within a Radius of 500 m (ANOVA)

5 Conclusions

Regarding the above mentioned question it can be stated that a variety of criteria concerning the provision of open spaces have a clear and significant value-increasing effect upon land prices in Berlin and Malmoe. More expensive prices of private plots within a ″green environment″ are an expression of greater appreciation of residents for their surroundings. This discovery ought to be recognised by politicians when setting priorities in financial and tax decisions.

Against the background of the difficult financial situation of communities mentioned initially and in the light of the results found here, local authorities might consider if and how they might enlist property owners to contribute to the future funding of the open space system whether for the creation of new parks or for their maintenance. The amount to be contributed should depend on the benefit the property owners gain from the appealing quality of urban open spaces. This notion could be put into practice by modifying the council tax and business rates respectively. This procedure might represent one possible ”innovative approach” of financing open spaces and might, therefore, be of value to the Urban Green Spaces Taskforce mentioned earlier. Finally, the results could also deliver inputs for the discussion on shrinking cities. Beside other factors development and maintenance of high quality open spaces could be a means to raise life quality and image of all those cities suffering under population decrease and not being able to compete with ‘life quality champions’ like Vienna, Zurich or Geneve.

6 Summary

The contradictory debate in European Cities on the benefits from urban open spaces is going on. Open spaces are highly regarded not only due to social, ecological and health related matters, but also increasingly to economic benefits resulting from them. On the other hand, recently less attention has been paid to urban open spaces in every day political and administrative life. That not only induces a loss of positive effects of open spaces on the environment in terms of ecosystem services, it might also be responsible for an accelerating decay of certain parts of cities because littered and run down parks are often associated with crime and under-privileged neighbourhoods by the public.

For a more rational debate on the importance of open spaces and their economic effects it is crucial to find out specific characteristics of open spaces and what economic benefits actually result from them. Using the revealed preference approach the relationship between the private market good (land value) and the public good (open space) was statistically analysed within a survey of German and Swedish cities. Due to the fact that benefits resulting from the public good are reflected in private ones, statistical analyses reveal significant effects of urban open space related criteria on land values within the investigated cities. Finally political consequences are perspectively discussed.

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Climate Change and the Function of Urban Green for Human Health

Wilfried Endlicher, Marlén Müller & Katharina Gabriel [1]

1 Introduction

Urban Green as part of urban ecosystems can be divided into three different types: street trees, lawns/ parks and urban forests (BOLUND & HUNHAMMAR, 1999). These three types generally have different extensions as well as structures. The influence they exert on their surroundings depends on their size and composition (lawn, bushes, and trees).

In cities, vegetation generates a variety of services, e.g. noise reduction, air filtration and micro climate regulation. The latter is due to modifications in incoming and outgoing radiation fluxes, including the fluxes of latent and sensible heat, air temperature, wind conditions and air humidity. These effects contribute to mitigating the urban heat island on a local scale. This paper is analysing the thermo-physiological impact of urban green and discussing its importance for human health at present and in a changing climate.

2 The Impact of Urban Green on Human Comfort

Studies on the influence of vegetation on thermal comfort in men have been carried out in several cities worldwide. JAUREGUI (1990/1991) found the Chapultepec Park (500 ha) in Mexico City up to 2-3 °C cooler than the surrounding neighbourhoods. He even observed this cooling effect at a distance of about 2.000 m. Analyses of long-term data (1882-1984) of ten-year running average high temperatures in downtown Los Angeles show a cooling rate of 3 K during the first five decades. This is explained by an increase of gardens and irrigation of plants. Then, trees were replaced by asphalt leading to an increase of 0,5 K per decade (ROSENFELD ET AL., 1995). In an investigation carried out in Tel-Aviv, SHASHUA-BAR AND HOFFMAN (2000) detected the average cooling effect of urban green on air temperature to be about 2,8 K. The smallest effect (1 K) was observed in a street with heavy traffic, the highest in a small garden (4 K).

MAYER AND MATZARAKIS (2006) investigated the impact of street trees on thermal comfort in Freiburg, Germany. They analysed the diurnal variation of several meteoro-

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logical variables at five different sites (street canyons, below chestnut tree canopies) on a typical summer day. While air temperatures did not show significant differences between the five sites, the Physiologically Equivalent Temperatures (PET[2]) differ considerably (Fig. 1 and 2). Temperature differences that correspond with up to three grades of thermal perception were calculated. This is due to modifications in the radiation fluxes and the much lower mean radiant temperatures below the tree canopies. They conclude that "the higher the canopy density respectively the lower the sky view factor are, the more pronounced is the attenuation of thermal stress of people" (MAYER AND MATZARAKIS, 2006, p. 288).

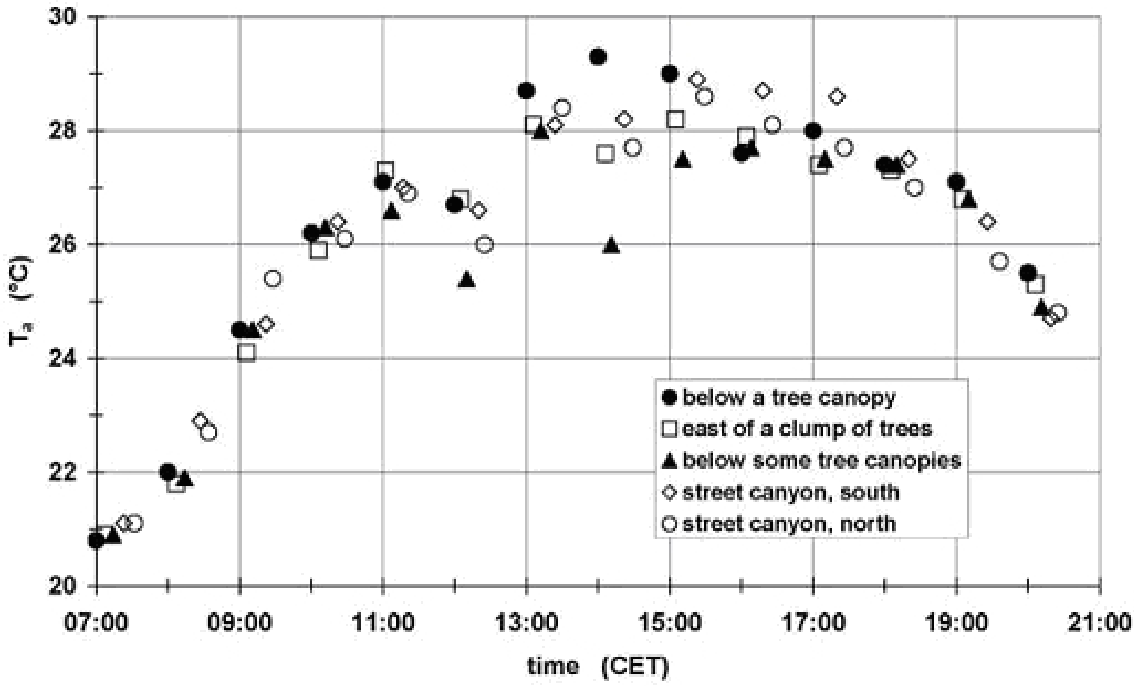


Fig. 1: Air temperature Ta (1.1 m above the ground) on a typical summer day (19th July 1999) at different sites in Freiburg (Germany).

Source: Mayer and Matzarakis, 2006

3 Thermal Comfort in the City of Berlin

Berlin, as Germany's biggest city, can be characterised as a remarkably green city with its more than 400.000 street trees, almost 13.000 ha of public green (including parks, squares, cemeteries and graveyards), and 16.000 ha of woods and forests. All in all, one third of the total area is covered by green (Berlin Senate Department for Urban Development, 2007).

[2] PET is a complex thermal index that takes into account meteorological, spatial (urban morphology) and personal parameters.

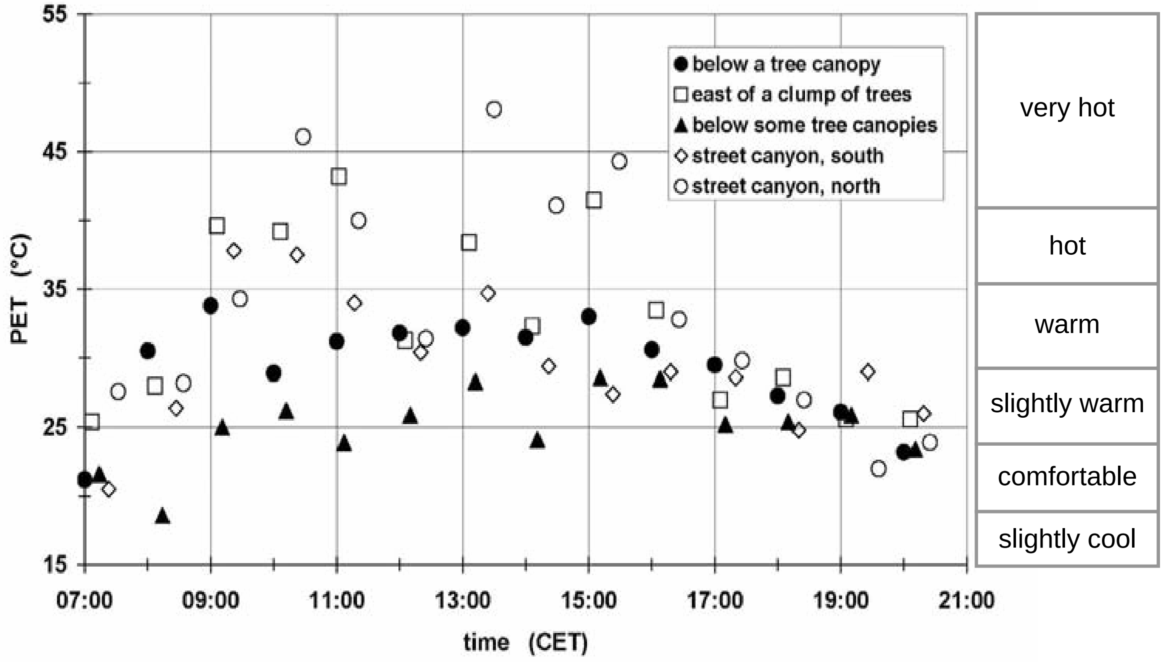


Fig. 2: Physiologically Equivalent Temperature PET (1.1 m above the ground) and grades of thermal perception on a typical summer day (19th July 1999) at different sites in Freiburg (Germany). Source: Mayer and Matzarakis, 2006 (modified)

Air temperature differences between two weather stations in Berlin, Alexanderplatz and Dahlem, were calculated for the summer months of 2003 (Fig. 3). In this comparison, Ale-xanderplatz stands for a typical inner city station, whereas the station in Dahlem is situated in the Botanical Garden. As can be deduced from the figure, Alexanderplatz in general is 0,5 – 2 K warmer.

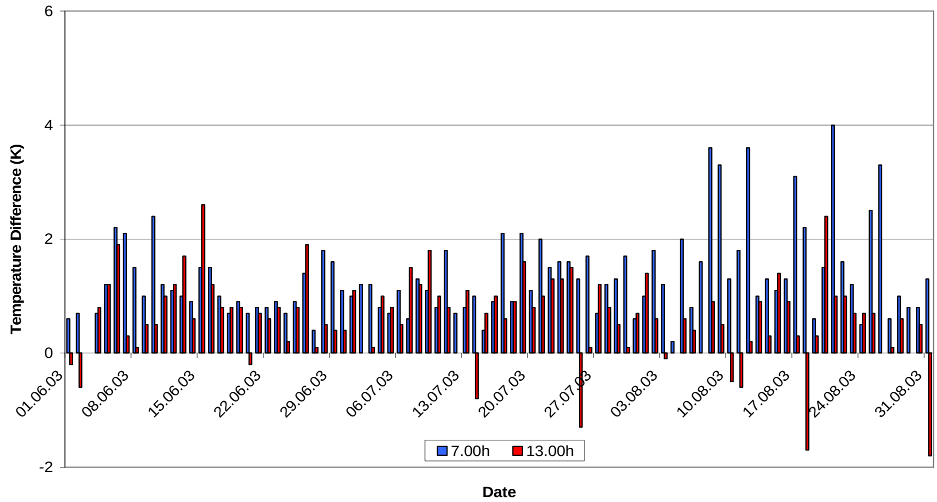
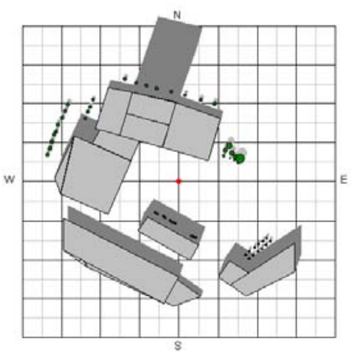
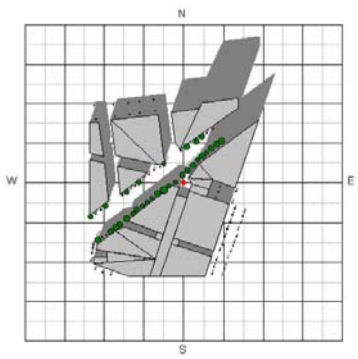
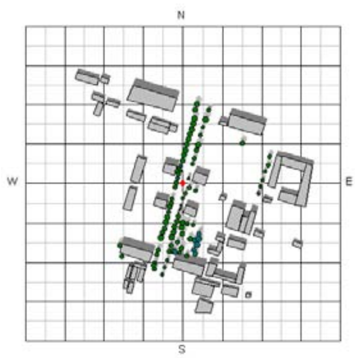


Fig. 3: Temperature differences between the weather stations Alexanderplatz and Dahlem, June to August 2003. Data: German Meteorological Service (DWD)

Three sample sites in the city of Berlin with different urban structures were chosen for studies of the thermal sensation in summer 2003. The main properties of these sites are resumed in Tab. 1. Alexanderplatz and Potsdamer Platz are located in the city centre, in densely built-up areas. About 22 % of the total district's area (Mitte) is covered by green spaces. Dahlem is situated in the south-west of the city and forms part of a district (Steglitz-Zehlendorf) with 35 % of green (1/3 parks and lawns, 2/3 urban forests). Additionally, 11 % of the total surface is covered by water surfaces (Berlin Senate Department for Urban Development, 2007).

Tab. 1: Characterisation of the investigation areas, fish-eye photos and maps with shaded areas in August at 13.00h

Alexanderplatz	**Potsdamer Platz**	**Dahlem**
• esplanade in the city centre • surrounding buildings between 12 m and 128 m high • completely sealed surface • lack of vegetation	• densely built-up area in the city centre • general height: 35 m, two higher buildings • narrow street canyons, completely sealed • trees of different ages and heights	• suburb in the south western part of Berlin • villa development, buildings and trees about 10 m and 12 m high • high amount of vegetation (trees, bushes, lawn)
		
		

For these three sites PET was modelled by dint of the programme RayMan Pro (University Freiburg) using meteorological data of Berlin-Dahlem and Berlin-Alexanderplatz.

In the morning only very few days exceeded the comfort range (18 °C - 23 °C PET) at all stations. Alexanderplatz showed highest values of mean and maximum PET (Tab. 2). At noon, differences between the three sites are somewhat larger. Alexanderplatz and Dahlem had both quite a high amount of days with strong or extreme heat stress with 14 days and 10 days, respectively. This is as well reflected in the mean values. In the evening the two sites in the city centre were characterised by having highest values.

Tab. 2: Mean and Maximum PET in summer 2003 comparing three neighbourhoods in Berlin

Station	Time	Mean PET (°C)	Maximum PET (°C)
Alexanderplatz	07:00h	16,0	27,1
	13:00h	25,8	41,0
	19:00h	19,5	30,0
Potsdamer Platz	07:00h	14,7	24,7
	13:00h	23,2	37,6
	19:00h	19,8	30,4
Dahlem	07:00h	13,7	23,1
	13:00h	24,4	40,8
	19:00h	18,4	27,7

At Alexanderplatz, high incoming radiation due to a small horizon limitation resulted in high mean radiant temperatures as well as in high PET values. Additionally, the heat storage in walls and streets contributed to the higher temperatures.

At Potsdamer Platz the input of insolation during daytime is quite low due to the high horizon limitation, as can be seen in the corresponding fish-eye photo in Tab. 1. Thermal stress at noon was the lowest comparing the three sites. But in the evening, and partly in the morning hours, this diurnal shading effect of the buildings and trees was cancelled out by the heat storage in the narrow street canyons leading to higher mean and maximum PET at this site. In Dahlem, the high amount of outgoing thermal radiation at night and the evapotranspiration resulted in cool and cold thermal sensations at 7.00h. Especially during sunny days the incoming radiation increased noticeably until noon leading to several days with at least moderate heat stress (MÜLLER ET AL., 2006).

As seen above, the diverse urban structures have varying influences on the development of PET in the course of the day. The calculated differences between Alexanderplatz and Potsdamer Platz, two sites located close to each other, are evidence of the high influence of the building structure and the resulting shading effects on the human biometeorological comfort.

4 Heat as a Health Risk Factor

"Urban heat waves are among the deadliest of all weather emergencies" (STÉPHAN ET AL., 2004, p. 39). In summer 2003, a long lasting and exceptionally intense heat wave caused several thousands of additional deaths in Western Europe. "Most of the deaths were counted in urban regions […]. Urbanization and urban architecture have a profound effect on heat mortality. High night time and morning temperatures characterize the climate of dense populated areas" (RUDEL ET AL., 2005). This hitherto unique phenomenon in Western Europe is now discussed as a probable future scenario in the changing environment at the end of the 21st century when heat waves will be more frequent and intense (BENISTON, 2004; ENDLICHER ET AL., 2006; IPCC, 2007).

From 1991 to 2003, day-to-day data of air temperature and mortality in Berlin were analysed. The general trend observed in other European cities in August 2003 could not be confirmed. However, such an outstanding event took place in summer 1994 (Fig. 4 and 5). "Mortality in Berlin had increased to 167 % in relation to the mean values between 1991 and 2003" (GABRIEL & ENDLICHER, 2006, p. 227).

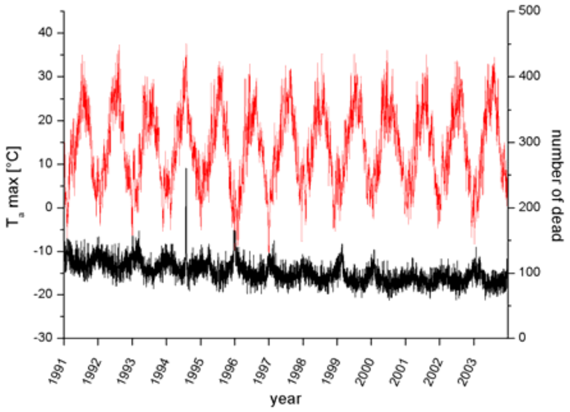


Fig. 4: Maximum daily air temperature (Ta max) and overall mortality in Berlin, 1991-2003

Data: German Meteorological Service, Federal State Office of Berlin

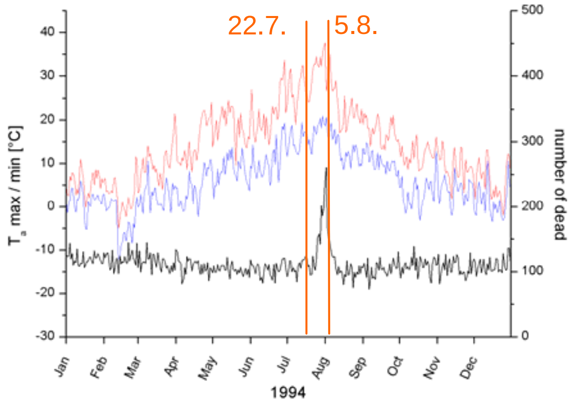


Fig. 5: Maximum and minimum daily air temperature (Ta max / min) and overall mortality in Berlin, 1994

Data: German Meteorological Service, Federal State Office of Berlin

In Berlin, August 2003 was not an extremely hot month when looking at the mean air temperatures at noon of the whole month (24,7 °C). Higher means (> 25,0 °C) were recorded during the months July 1994, July 1995 and August 1997. Compared to 1994, the small temperature difference between June, July and August 2003 is noticeable. It may have allowed the human body to adapt to the warmer temperatures before confronting the heat wave (Fig. 6).

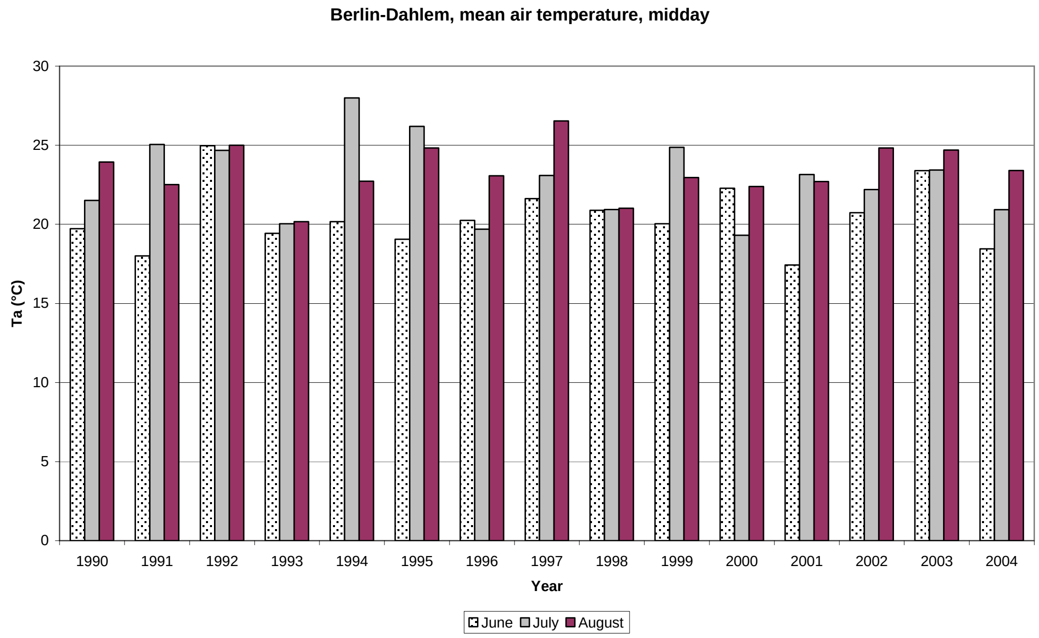


Fig. 6: Monthly mean air temperatures (T_a) at midday (13.00h/ 14.30h) in Berlin-Dahlem
Data: German Meteorological Service (DWD)

The summer of 1994 and 2003 were contrasted to get a better idea about the thermal impact of these high temperatures on the human body. PET was used to estimate the thermal load a standardised young male person may perceive in Berlin-Dahlem. The Fig. 7 illustrates the different grades of thermal stress during the months June to August about noon. PET between 29 °C and 35 °C stand for moderate heat stress, between 35 °C and 41 °C for strong heat stress and those exceeding 41 °C for extreme heat stress (VDI 3787, 1998).

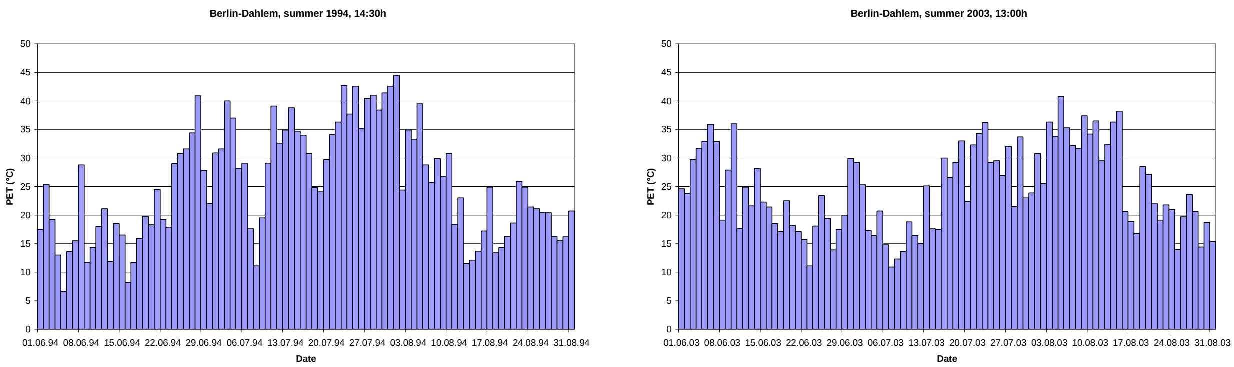


Fig. 7: Calculated Physiologically Equivalent Temperature (PET) in Berlin-Dahlem for summer months June to August 1994 and 2003 at 14.30h/ 13.00h
Data: German Meteorological Service (DWD)

5 Summary

In cities, global and local warming interfere with each other and will therefore lead to an intensification of estival heat stress. Heat waves are not solely weather phenomena. They are closely connected with human health, behaviour, social structures and medical care. Consequences for human health have to be taken into account since mortality and morbidity show significant peaks in large cities during heat waves and are expected to keep on rising affecting ten thousands of people during each strong event.

Future urban planning has to include the effects of climate change. Green areas already play an important role as they contribute to mitigating the urban heat island and thus reducing thermal stress (Jendritzky, 2007). In view of global warming, they will be one of the most important adaptation strategies in urban environments.

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Reduction of Airborne Particulates by Urban Green

Marcel Langner [1]

Many airborne particles have adverse health effects on humans. Vegetation can either increase or decrease their concentrations. Measurements on urban greens in Karlsruhe und Berlin show a homogeneous distribution of fine particles and significant filter effects only for coarser particles.

1 Introduction

Particles in ambient air are characterised by a high variability in many properties. Their size ranges from a few nanometres up to 100 microns, they vary in their chemical composition, their shape and surface structure. While not all particles are harmful to humans, at least two groups of particles have adverse effects on human health: bioaerosols like pollen and fungal spores and particles with aerodynamic diameter of less than 10 μm (PM_{10}), which are able to pass the larynx and therefore can penetrate into the trachea and the lungs.

Various epidemiological and toxicological studies have shown that exposure against PM_{10} and $PM_{2.5}$ (particles with aerodynamic diameter of less than 2.5 μm) has negative effects on the respiratory and cardio-vascular system (KAPPOS et al. 2004). It was estimated that PM_{10} caused about 40,600 cases of premature death in Austria, France and Switzerland in 1996 (WHO 1999). In the same study, health costs attributable to PM_{10} were calculated using a willingness-to-pay approach. Costs of mortality amounted to €36,525 million, costs of morbidity to €13,192 million. Based on the concept of the value of a statistical life, a reduction of PM_{10}-concentrations from 52.6 $\mu g \cdot m^{-3}$ to 30 $\mu g \cdot m^{-3}$ in the eight largest cities in Italy was calculated to reduce mortality worth €2,574 million (based on median willingness-to-pay) or €5,489 million (based on mean willingness-to-pay) per year (ALBERINI & CHIABAI 2007).

To mitigate the effects of particle pollution on human health, the Council of the European Union determined limit values for PM_{10} in the council directive 1999/30/EC (EU 1999). A 24-hour value of 50 $\mu g \cdot m^{-3}$ should not exceed more than 35 times a calendar year and the annual mean PM_{10}-concentration should not exceed 40 $\mu g \cdot m^{-3}$. The new limit values became effective on 1 January 2005. However, the daily limit values for PM_{10} could not be kept in many European cities in spring 2005 already. Hence, there is

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an urgent need for measures to reduce PM_{10}-concentration. This is especially true for cities, since particle concentrations are increased in urban agglomerations due to enhanced anthropogenic emissions by traffic exhaust and industrial activities.

There are two basic ways to achieve a particle reduction. Firstly, the emission of particles into the atmosphere could be decreased. Examples are the installation of filter systems into industrial plants and diesel engine vehicles (soot filters). Secondly, deposition of particles could be increased. Within this context, vegetation and especially trees are assumed to play an important role (FARMER 1993). Due to its large leaf surface relative to the ground it covers, vegetation can act as a biological filter for airborne particles. If urban vegetation proves to be an efficient sink for airborne particles, it will provide an important ecosystem service.

2 Influence of vegetation on ambient particle concentration – an overview

The effect of vegetation on ambient particle concentration is not restricted to particle deposition on leaf surfaces. Tab. 1 shows that vegetation can cause either decreased or increased particle concentrations. In both cases, there are direct and indirect processes. Whereas direct processes indicate direct removal or emission of particles by vegetation, indirect influence means an impact of plants on processes that directly govern particle concentrations.

Tab. 1: Influence of vegetation on ambient particle concentration

Decrease in particle concentration	**Increase in particle concentration**
Direct – Filtering by deposition	**Direct** – Emission of particles
Indirect Modification of airflow Reduced re-suspension	**Indirect** – Modification of airflow

Deposition of particles on plant surfaces is affected by different factors including micrometeorological variables like wind velocity, properties of the particles and variables of the surfaces like stickiness or the canopy structure (SEHMEL 1980). The most important particle property with influence on deposition velocity – a measure how fast particles are deposited to surfaces – is their size. Different models of particle deposition to leaf surfaces show a minimum of deposition velocity for particle diameters in a broad range of about 0.08 to 0.3 μm (ZUFALL & DAVIDSON 1998). Smaller particles have higher deposition velocities due to diffusion, larger particles are deposited faster by sedimentation and inertial processes. Detailed results from particle deposition on a single roadside tree will be discussed in the next section.

There are two main indirect processes leading to a decrease of particle concentration by vegetation. Firstly, a vegetation cover reduces the re-suspension of soil material or previously deposited atmospheric particles. This is caused by reduced wind speed in plant canopies and due to fixation of particles by roots. Secondly, a modification of airflow may lead either to an increase or decrease of particle concentration. Two examples will illustrate this:

- Polluted air flowing towards a dense vegetation stand is raised and therefore the particle concentration within the vegetation stand is lower than outside (fig. 1).
- When trees are placed in narrow streets with high traffic flow, ventilation in the street canyon is reduced and dispersion of emitted particles is hindered. This effect can be quantified using numerical micro-scale models like MISKAM. Dependent on the configuration of the canyon and the tree rows an increase in kerbside concentrations of 20 % (RIES & EICHHORN 2001) or up to 100 % (own calculations) can be found for an airflow perpendicular to the canyon.

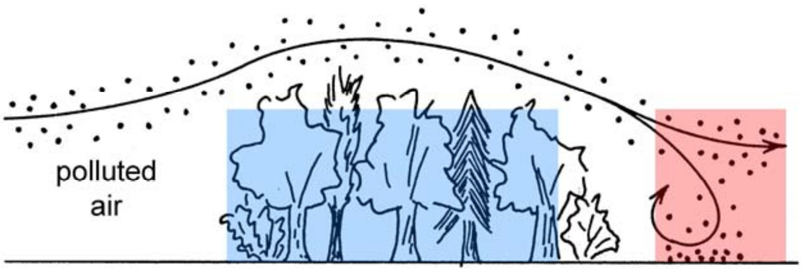

Fig. 1: Modification of airflow leads to areas with increased particle concentration (red) and decreased particle concentration (blue). After FELLENBERG (1999:41), modified

Fig. 2: Stand of *Ambrosia artemisiifolia* in Berlin-Steglitz (5 Sep 2006)

Vegetation may also increase particle concentrations by direct emissions of pollen and small plant debris or by release of biogenic volatile organic compounds which are converted in the atmosphere into nano-particles with diameter of less than 5 nm (O'DOWD et al. 2002). Beside 'classical' allergenic pollen types like *Betula* or *Artemisia*, in the future pollen from the neophyte *Ambrosia artemisiifolia* (Common Ragweed, fig. 2) may cause severe problems in urban regions in Middle Europe. *Ambrosia artemisiifolia* produces highly allergenic pollen and several environmental conditions typically found in mid-latitude cities, particularly raised temperature and CO_2-levels, are favourable for its pollen production (ZISKA et al. 2003).

3 Filter effect of a single tree

To investigate the filter effect of a single tree, an intensive measurement programme was carried out in Karlsruhe, a middle-sized town in Southwest-Germany with about 275,000 inhabitants during the vegetation period 2002. Within the crown of a roadside *Acer platanoides* (Norway Maple) and its surroundings particle distribution was measured, particle deposit on the leaves was determined and wet and dry deposition beneath the crown was collected. A detailed description of the methods and results can be found in LANGNER (2006).

Ambient concentrations of coarse particles were measured with passive Sigma-2 samplers according to VDI-Guideline 2119/4. In these samplers, particles are collected on adhesive transparent slides, so that black and transparent particles with diameters between 3 and 96 µm can be distinguished under an optical microscope. The results are shown for coarse particles between 3 and 12 µm and the entire coarse particles between 3 and 96 µm in fig. 3. Concentrations of both particle fractions were highest at the edge of the road and decreased with increasing distance from the road, both in the vertical and horizontal direction. However, while concentrations of the finer fraction decreased by 25 % from 12 to 9 $\mu g \cdot m^{-3}$, concentrations of the entire coarse particles decreased by 40 % from 25 to 15 $\mu g \cdot m^{-3}$. Hence, finer particles show a more homogeneous spatial distribution. Measurements of PM_{10} revealed an even more homogeneous distribution of PM_{10} within the tree crown.

These results can be compared with the dust deposit on the leaves in order to get a first clue of preferentially deposited particles. To determine the dust deposit, leaves were collected at 30 positions within the crown. Dust deposit was measured by washing the dust from the leaves using a brush. Like the distribution of ambient particle concentrations, dust deposit was highest near the road (fig. 4). However, dust deposit showed large spatial gradients with a decrease by 66 % at the first sampling and a decrease by 73 % at the last sampling in 2002. In connection with the distribution of airborne coarse particles, it can be concluded that mainly larger coarse particles are deposited on the leaves.

To quantify deposition beneath the crown and to get an estimation of the deposited PM_{10}, 20 wet-dry-samplers were placed beneath the crown. These devices collect wet and dry deposition separately in glass vessels. During the vegetation period 2002 about 2 kg of dust were deposited beneath the crown, from which 75 % were wet deposition. Since the deposited material consists not only of PM_{10} but also of coarser particles and leaching from the leaves, it was necessary to get an estimate of the PM_{10} fraction. Therefore, trace element analyses of PM_{10} and deposition samples were carried out. The two

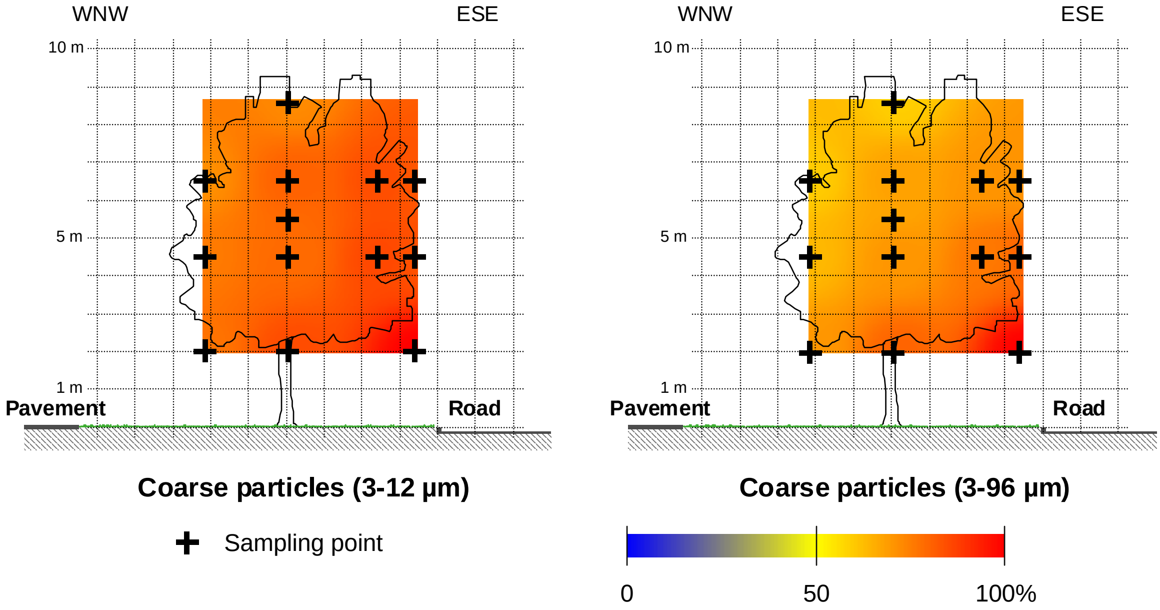

Fig. 3: Mean relative distribution of two coarse particle fractions in a plane perpendicular to the road. Measurements were carried out from 20 March until 18 December 2002, interpolation was done using the inverse distance method

elements lead and vanadium were chosen as trace elements for PM_{10}, because their concentration in PM_{10} samples was relatively constant. From the concentrations of lead and vanadium in the deposited material, a PM_{10} fraction of 20 %, i.e. 400 g was estimated.

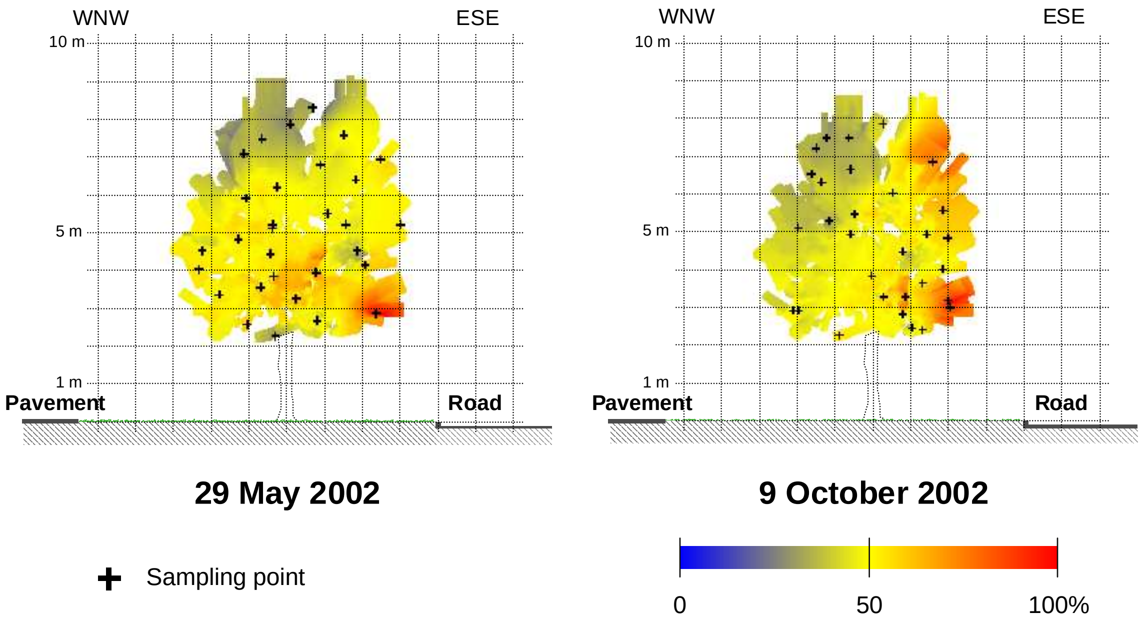

Fig. 4: Mean relative distribution of total dust deposit on the leaves
Shown are the results of the first and last sampling, interpolation was done using the inverse distance method

To evaluate this figure, it was compared with an estimate of particle mass emitted by road traffic beside the tree during the vegetation period. By using an emission factor of 150 mg PM_{10} per vehicle and km, the emitted mass of PM_{10} amounts to 3.5 kg. Referring only to these local emissions, a filter potential of 11 % can be assessed. Taking the urban background also into account, this potential decreases to fewer than 5 %, since local emissions are accountable for about 25 % of the PM_{10} concentrations at this site (see below).

4 Particle concentration on urban green areas

Measuring particle fluxes to vegetation is very elaborate, since it has to consider particle deposition on the leaves as well as throughfall and deposition beneath the canopy. Especially in heterogeneous environments like urban areas, it is associated with many methodological problems. In order to get first impressions of the filter effect of complex vegetation structures, it seems more promising to determine ambient particle concentrations on green areas and examine whether plants affect them.

To investigate particle distributions on vegetation covered urban sites, particle concentrations were measured on an urban green area adjacent to the investigated tree in Karlsruhe, and on a brownfield formerly used as a marshalling yard in Berlin, Germany's capital (3.4 million inhabitants).

PM_{10}-concentrations on the urban green in Karlsruhe were determined using two low-volume samplers in 2003 und 2004. Particles were collected on quartz fibre filters during sampling periods of two days. On sampler was located at a reference site about 45 m away from the road on a lawn, the second sampler was placed at various locations within the green area and at the edge of the road.

Fig. 5 shows the results of the measurements. At the edge of the road, concentrations are on average 23 % higher than at the reference site. This clearly identifies the road as a source of PM_{10}. If a vegetation structure like a hedge acted as a comparable sink for particles, concentrations should show a similar decrease. It is obvious from fig. 5, that concentrations of PM_{10} show no differences greater than the accuracy of the measurement device itself, which is in the range of 2 %. PM_{10} is very homogenously distributed within the urban green regardless whether plants are leaved or not. The homogenous distribution is also independent of the vegetation type in which PM_{10} was measured. Therefore, it could not be proved that a single vegetation structure has a significant lowering effect on PM_{10}-concentrations.

Similar results were obtained by measurements on a brownfield in Berlin. PM_{10}, $PM_{2.5}$ and PM_1 were determined with a GRIMM spectrometer (dust monitor #107). During a

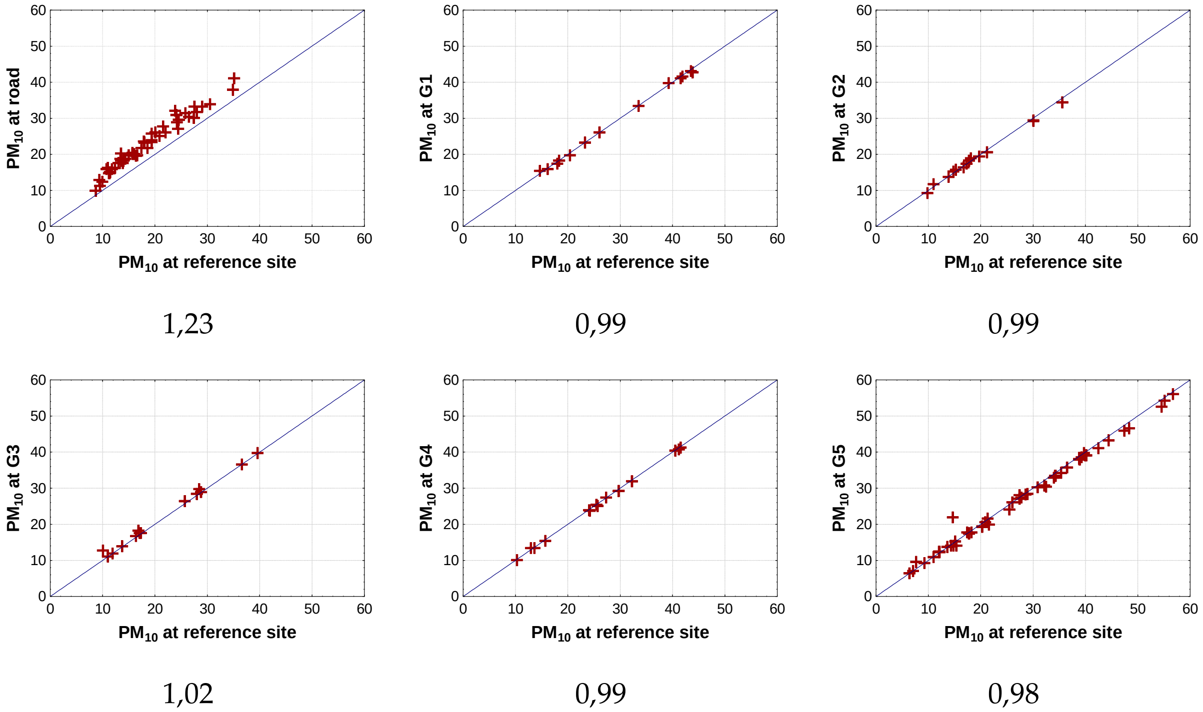

Fig. 5: PM10-concentrations at six locations on a roadside urban green in comparision with a reference site
The blue line is the bisector of the first quadrant, all concentrations are in µg·m^{-3}. The figures beneath the scatter diagrams indicate the mean ratio between concentrations at the site and the reference site. The sites G1 to G5 are characterised as follows:
G1: within a *Pyracantha* hedge, distance to the edge of the road: 25 m
G2: beneath a tree row, distance to the edge of the road: 35 m
G3: beside a *Pyracantha* hedge, distance to the edge of the road: 28 m
G4: on a lawn, distance to the edge of the road: 50 m
G5: beneath a group of trees, distance to the edge of the road: 58 m

campaign in summer 2006 at four locations with various vegetation types no significant spatial differences could be found.

On the same brownfield, concentrations of coarse particles were measured with seven Sigma-2 samplers over a period of one year (March 2006 until February 2007). One sampler was located at the edge of the adjacent road, six samplers were placed within the brownfield along two transects. The first transect passes through an area mainly covered by herbaceous plants, the second passes groups of shrubs and trees.

Coarse particle concentrations showed a higher spatial variability than the fractions of PM. The highest concentrations occurred at the roadside position where they were more than twice as high as on the brownfield. Factor analysis of the concentration data identified two factors. The composition of the factors differs between the positions, but black particles in the range between 3 and 24 µm appeared at every position in one factor. Since these particles consist mainly out of tyre wear, large soot particles and fly

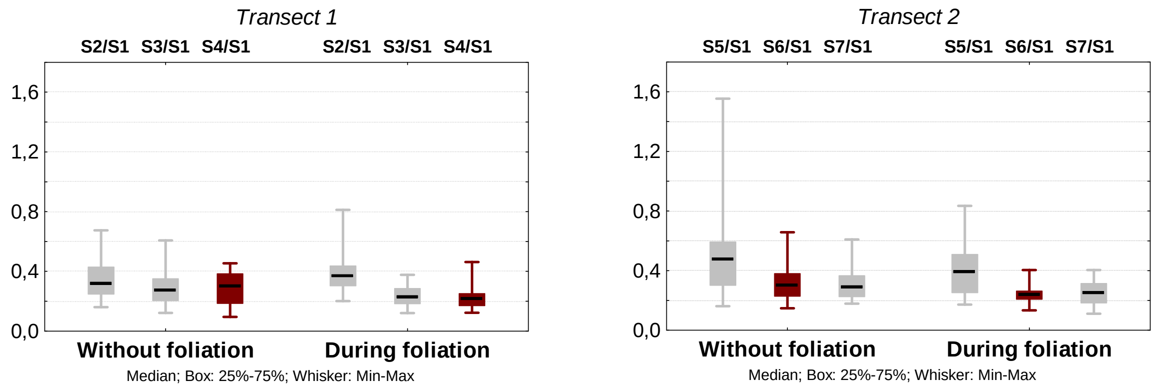

Fig. 6: Distribution of coarse black particle (3-24 μm) concentrations during the period of foliation and without foliation at sites S2 – S7
Concentrations are normalised to roadside concentrations (S1), significant ($\alpha = 0.05$) decreased concentrations during foliation are highlighted red

ashes (SCHULTZ 1989), this fraction can be used as in indicator for traffic emission. According to the U-test of Mann-Whitney this fraction showed significantly decreased concentrations during the period of foliation at two positions (fig. 6). Because transparent particles between 3 and 24 μm showed increased concentrations during foliation, the decrease of black particles is not likely caused by a modified airflow but by deposition.

5 Conclusions

Vegetation can both act as a source and a sink for particles with adverse effects on human health. While some species emit allergenic pollen, leaf surfaces are not an efficient sink for fine particles. Therefore, PM-concentrations are homogeneously distributed within urban green areas. However, some coarse particle fractions are well captured by vegetation. For planning purposes, it is important to ensure good ventilation near strong particle sources like roads so particles can be well dispersed. Since single vegetations structures have only a minor effect on PM_{10}-particles, there is the need for large green areas in urban agglomerations to lower concentrations of anthropogenic particles.

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