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The impact of Short Rotation Coppice (SRC) cultivation on the environment

Ioannis Dimitriou *, Christel Baum**, Sarah Baum***, Gerald Busch****, Ulrich Schulz*****, Jörg Köhn******, Norbert Lamersdorf*******, Peter Leinweber**, Pär Aronsson*, Martin Weih*, Göran Berndes******** and Andreas Bolte***

Introductory of the vTI series on SRC and environmental issues

The following six articles included in this series are describing potential effects and implications of the cultivation of Short Rotation Coppice (SRC) on the environment. The articles deal with the effects of SRC on water issues (Dimitriou et al., 2009), groundwater recharge with a spatial perspective (Busch, 2009), phytodiversity (Baum S. et al., 2009) and animal diversity (Schulz et al., 2009), as well as soil issues (Baum C. et al., 2009). Finally, the paper of Köhn (2009) deals with socioeconomic implications of SRC cultivation. The term SRC has prevailed to be referred to biomass productions systems cultivated for energy purposes using fast-growing tree species with the ability to resprout from the stumps after harvest, which occur in short intervals (i.e. 2 to 6 years). The management practices for SRC (soil preparation, weed control, planting, fertilization, harvest etc) are more similar to those of agricultural annual crops than forestry, although the species currently used in commercial SRC plantations in Europe are tree species such as willows (*Salix sp.*) and poplars (*Populus sp.*), i.e. fast-growing tree species with good coppice ability that produce much biomass even under very short harvest intervals. The reader should distinguish between SRC and Short Rotation Forestry (SRF), which is a broader term describing forest systems for biomass production (for energy purposes but also for others) using fast-growing tree species grown at denser spacing and elevated maintenance than in traditional forestry, typically harvested after 2 to 25 years depending on the desired end-product. We consider therefore that SRC falls within SRF and represents a more specialised and intense practice of SRF dedicated mainly for energy purposes. In the articles included in this series, we explicitly describe the effects of SRC on the environment, referring only in a few parts to effects of single stem trees used in SRF, since their use is currently broader than for SRC (e.g. poplars).

SRC for production of biomass for heat and/or electricity is considered as one of the promising means to contribute to meeting the European targets to increase the amount of renewable energy, and has been identified as the most energy efficient carbon conversion technology to reduce greenhouses gas emissions (Styles and Jones, 2007). Additionally, SRC cultivation in larger scale could help to meet social and economic targets of other EU policies (e.g. EU Rural Development, CAP reform). This combination of technological and political drivers has stimulated the interest in the growing and processing of biomass crops as a source of renewable energy. Different incentives for growing SRC have been introduced in several European countries. Currently, ca. 15 000 ha are being cultivated with willow SRC in Sweden, mostly in productive agricultural land. Smaller areas are cultivated in Italy (ca. 6 000 ha, mostly poplars), Poland (ca. 3 000, mostly willows), UK (c. 3 000 ha, mostly willows), Germany (ca. 1 500 ha, mostly poplars), and other European countries. Although these areas cannot be considered as extensive in comparison to

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other agricultural crops, a rapid increase of SRC in several European countries has been projected already in the short-term. For example, in Sweden the Swedish Board of Agriculture predicts a short-term increase of SRC to 30 000 ha (Jordbruksverket, 2006), the UK Biomass Strategy predicts that perennial energy crops will occupy some 350 000 ha by 2020 (DEFRA, 2007), and in Germany SRC cultivation areas may also increase markedly during next years due to a changing subsidise policy and the identification of high cultivation potentials for certain areas (e.g. 200 000 ha for the federal state of Brandenburg; Murach et al., 2008).

SRC is a perennial crop that differs from arable crops in a number of physical traits, and moreover, is managed quite differently. In particular, it is anticipated that SRC plantations will remain in place for a number of years (10 to 25 years depending on national regulation and market issues), thus taking the land out of arable rotations; harvest normally occurs in winter or early spring; the plants are deeper rooted and generally have a high water consumption compared to conventional crops; SRC is much taller (ca. 5 to 8 m at harvest) than other arable crops. Additionally, once established, SRC requires no annual soil cultivation and considerably less agrochemical inputs. Typically, much less nitrogen fertilizer is applied to SRC compared with arable crops. In fact, the vast majority of Swedish and UK SRC fields currently are not supplied with inorganic fertilizer at all. Minimal or no fungicides or insecticides are applied, although herbicides are needed during the establishment phase. As a result of the lower intensity, and particularly of the lower nitrogen fertilizer applications, SRC has a much lower carbon footprint compared with food or biofuel production from annual arable food crops (Heller et al., 2004).

A rapid large-scale shift from "conventional" agricultural crops to SRC will certainly have positive and negative implications on a range of environmental issues. A concentrated increase of SRC grown on agricultural land is anticipated in areas neighbouring power stations using biomass as a fuel (with approximate radius from power stations of up to 100 km). In such areas, SRC might need to be cultivated on a substantial fraction of all available agricultural land to fulfill biomass needs for fuel in power stations, simultaneously being economically and energy efficient. Where SRC is grown to supply small local heat and/ or power stations, plantations will be on a much smaller scale, although they may still be concentrated. This, coupled with the above-mentioned special features of SRC will surely alter the appearance of the landscape and have potential implications for the local water and soil quality, hydrology, carbon storage in soil, and biodiversity. The papers included in this series deal with the potential effect of SRC on all these issues, and speculate on the expected positive or negative impact of SRC cultivation at the micro- (field) and macro- (catchment and larger) scale based on the existing literature. Research results related in the above topics are presented, possible gaps in knowledge and future research aspects are identified, and assumptions for potential impacts of SRC cultivations are drawn.

For example, the paper of Dimitriou et al. (2009) examines the implications of SRC on water balances and water quality. The evapotranspiration (V_{ET}) rates reported in the literature for willow and poplar SRC seem to be somewhat higher than arable crops in most cases, but such values vary noticeably and are related to site-specific factors such as the local conditions in terms of soil type, temperature, groundwater level, the precipitation at the site, the planted species or clones, the age of the crop, and their interactions. Despite this, effects on water balances on catchment level have not been reported or justified. All these should be combined with the reported improved groundwater quality in terms of nutrient leaching when SRC is planted in a certain area. Therefore, if the identified potentially negative impacts would be considered and minimized, the average effect of SRC on water issues could be positive. To succeed in this, the authors suggest that the V_{ET} differences between SRC and arable crops should be better studied, comparing SRC fields grown for several years and respective arable fields in the same areas. Moreover, the relation of V_{ET} with local precipitation and other factors, i.e. root development and groundwater availability, should be closer examined and combined with positive effects of SRC on groundwater leaching compared to other arable crops.

To promote future decision-making processes with respect to the envisaged expansion of SRC on productive but also on marginal soils, the potential local impact of SRC on water needs to be extrapolated in the larger scale. This can be assessed within the framework of existing ecological and spatial planning as a tool for rapid qualitative assessments. Such an effort is conducted by Busch (2009) in his paper included in this series. He assessed the suitability of SRC for two municipalities in the district of Uelzen, Northern Germany, applying GIS analyses and based on the theoretical impact of SRC on groundwater recharge. The author calculates the ecological impact as a function of groundwater recharge for different SRC water use boundaries, but states that better and more precise data concerning SRC water use are needed to assess the ecological impact of SRC on water issues. Such impact assessments and ecological evaluations of landscape functions need to carefully consider site-specific conditions (soil type, climatic parameters etc) as well as existing environmental goals.

For this reason, it is interesting to consider the results of Baum S. et al. (2009) about the impact of SRC on phytodiversity, and the results of Schulz et al. (2009) on animal diversity, both included in this issue. In both papers it is suggested that there are indications about increased biodiversity in SRC in comparison with other arable crops. Protection and increase of biodiversity is a political commitment set by the European Union, and therefore it would be of key importance if biodiversity could be increased within the stand and/or in the surroundings when SRC replaces other crops in agricultural areas intensively managed. Concerning phytodiversity, and despite the lack of long-term studies that would enable better understanding on how SRC affects phytodiversity in time and space, there are indications that it would be increased if SRC is planted in areas dominated by agriculture or coniferous forest. Baum S. et al. (2009) identify also areas where SRC establishment might negatively affect phytodiversity, especially habitats of threatened species such as undisturbed peat land, forest wetlands, or areas adjacent to lakes or rivers. The authors present simple guidelines which favour biodiversity in SRC plantations, but they also underline that there are still a series of open questions to enable the best management of SRC in terms of increasing phytodiversity, which would be obtained if long-term studies can be initiated. Schulz et al. (2009) claim that research studies for animal diversity in SRC has been conducted mostly for birds and ground beetles, and that more research for invertebrates is needed. According to related literature, their diversity, equated with species richness, differs considerably in SRC in comparison to arable fields; whereas bird diversity in SRC is higher than in agricultural cropland, higher diversities of ground beetles have been found in arable fields. However, it has been found that animal diversity depends on a number of factors such as the age of the plantation, the tree species/clone, the plantation size, the habitat structures and the location of the plantation (surroundings and other uses). The authors, besides making recommendations for SRC management that will potentially increase animal biodiversity by encouraging habitat diversity, also point out that more studies in commercial SRC fields should be conducted. They add that the influence of the surrounding landscapes on the diversity of SRC and the influence of SRC on the diversity of the surrounding landscapes need to be considered, reminding as other authors do in this series, the importance of the decision for locating the plantations.

The potential effects of SRC on soil issues are described in detail by Baum C. et al. (2009), illustrating the multiple function that SRC can have when planted in a certain area. For instance, SRC can play a positive role as a carbon sink, therefore contributing to mitigate global warming impacts, mainly due to the annual leaf litter stored in the soil and minimum tillage and other soil management practices compared to other arable crops. The authors report that increased carbon sequestration has been reported to occur when SRC is planted on former arable soils, however, the amounts of carbon stored seem to be governed by the initial soil properties, and therefore approaches for the selection of most promising sites for carbon sequestration must be developed. The nutrient cycling in/from soil planted with SRC can also be advantageous compared to arable crops due to management practices and the leaf litter biomass and rhizodeposits. Furthermore, willows and poplars can be colonized by ectomycorrhizal fungi, in contrast to other arable crops, and consequently positive changes in soil microbial diversity and activity can be achieved in arable soils. Other positive impacts due to the non-tillage management and the high litter supply as the abundance and diversity of the soil fauna are also reported, such as an increased abundance of earthworms and an increased diversity of carabids. Finally, SRC as multi-purpose plantations can be used for phytoremediation of contaminated soils (e.g. extraction of Cd, Zn and other heavy metals, and degradation of organic compounds) when biomass is produced in such sites and improve the soil quality of moderately contaminated arable land. This is also the case on marginal land that can be returned to agricultural production after SRC cultivation for a number of years. The authors conclude that all the positive effects on soil ecology can be maximized with proper site selection and management adjustments, but there should be a balance between such management "modifications" and the sustainable production of biomass from SRC, keeping in mind that SRC is a commercial crop for production of biomass for energy competing high value arable crops.

This balance between maximum environmental benefits and maximum attained biomass production from SRC is a big challenge that all stakeholders involved in SRC cultivation (farmers, decision-makers, researchers, and others) should deal with. Despite all the above-mentioned expected positive environmental impacts of SRC, farmers need to be convinced to cultivate the crop, and this is typically achieved when the economic profit from the cultivation of a new crop such as SRC is equal to or higher than that of other established or conventional crops. Such issues are addressed in the article of Köhn (2009) included in this series. To encourage farmers to grow SRC instead of other crops in order to achieve environmental benefits, decision-makers should be prepared to contribute with direct or indirect incentives to the farmers. For instance, a potential economic compensation to the farmers could be a form of reward for those helping to fulfill national and European environmental goals already set and simultaneously keeping agricultural land into production. A prerequisite for such incentives and decisions, however, is to have scientific evidence concerning the quantification of those environmental benefits after SRC cultivation compared to other crops, and to evaluate the extent of benefits for the society. Such issues concerning the added value of SRC cultivation, when at the same time important environmental goals are achieved, should be one of the drivers for SRC cultivation, besides or in combination with drivers for producing biomass for energy to achieve renewable energy commitments.

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The impact of Short Rotation Coppice plantations on phytodiversity

Sarah Baum*, Martin Weih**, Gerald Busch***, Franz Kroiher* and Andreas Bolte*

Abstract

There is currently an increasing demand for wood as a renewable energy source. Plantations with fast growing trees, such as poplars and willows, have been established, grown in a short rotation coppice (SRC) system. A further increase of SRC plantations is expected in the future, but their effects on biodiversity are little known. We give an overview of the current state of knowledge on the phytodiversity in SRC plantations.

Many factors influence the vegetation in a SRC plantation. E.g. light climate and the tree age play important roles for species composition, species number and vegetation cover. The older the planted crop, the shadier the conditions for the ground vegetation, which is associated with a shift from annual to perennial and from light demanding to shade tolerant species. Furthermore, the land use history and the vegetation in the surrounding landscape have considerable influence on species composition in SRC plantations. The more diverse the surrounding landscape, the more species are able to establish in the plantation. Smaller plantations with longer edge habitats (ecotones) facilitate species immigration from the surroundings better than larger plantations. Smallscale structured plantations increase biodiversity.

When comparing SRC plantations with other land uses, diversity is often higher than in arable fields and coniferous forests, but lower than in oldgrowth mixed deciduous forests. If established in areas dominated by agriculture or coniferous forests, these plantations may increase regional diversity. Habitats of threatened species as well as areas adjacent to lakes or rivers should be avoided, whereas former arable lands and grassland fallows are generally well suited.

Keywords: biodiversity, energy crop, land use, landscape scale, poplar (Populus), sewage sludge, site preparation, species richness, SRC, willow (Salix)

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Zusammenfassung

Der Einfluss von Kurzumtriebsbeständen auf die Phytodiversität

In den letzten Jahren ist die Nachfrage nach Holz als nachwachsender Rohstoff zur energetischen Nutzung gestiegen. Dazu werden Plantagen mit schnell wachsenden Baumarten, sog. Kurzumtriebsplantagen (KUP), angepflanzt. Mit einem weiteren Anstieg kann gerechnet werden, wobei der Kenntnistand über die Auswirkungen der KUP auf die Biodiversität bislang gering ist. Der Artikel gibt einen Überblick über den gegenwärtigen Wissensstand zur Phytodiversität in KUP.

Die Begleitvegetation in KUP wird durch viele Faktoren beeinflusst. Licht und damit verbunden das Alter der Plantage spielen eine wichtige Rolle für die Artenzusammensetzung, Artenzahl sowie die Bodenbedeckung der Vegetation. Umso älter die Plantagen sind, desto weniger Licht steht für die Begleitvegetation zur Verfügung. Dies bewirkt eine Verschiebung von einjährigen zu mehrjährigen und von lichtliebenden zu schattentoleranten Arten.

Die vorherige Vegetation sowie die umgebenden Landnutzungstypen haben einen großen Einfluss auf die Artenzusammensetzung der KUP. Umso vielfältiger die Umgebung ist, desto mehr Arten können sich in einer KUP etablieren. Kleinere Plantagen mit längeren Randzonen sind besser für eine Besiedlung aus der Umgebung geeignet als größere Plantagen. Kleinstrukturierte KUP erhöhen die Biodiversität.

Verglichen mit anderen Landnutzungen sind KUP häufig artenreicher als Ackerflächen und Nadelwälder, aber artenärmer als alte, gemischte Laubwälder. In einer von agrarischer Nutzung oder von Nadelwäldern dominierten Umgebung erhöhen KUP oft die regionale Diversität. Es wird davon abgeraten, KUP in Gegenden mit seltenen Arten sowie an Seen und Flüssen anzulegen, während ehemalige Ackerflächen und Grünlandbrachen häufig gut geeignet sind.

Schlüsselworte: Biodiversität, Energiepflanze, Landnutzung, Landschaftsebene, Pappel (Populus), Klärschlamm, Bodenbearbeitung, Artenvielfalt, KUP, Weide (Salix)

1 Introduction

Since the early 1970s, many countries around the world have been developing new crops in order to increase the biomass resource base for production of bioenergy (Wright, 2006). Fast growing trees have been cultivated for many years in various European countries, with test-trials on willows grown in short rotation for the purpose of biomass production initiated in the 1980s (Kuzovkina et al., 2008). Commercial willow plantations are grown in a short rotation system on 15,000 ha in Sweden (Nordh, 2007), while in Poland the planted area is about 6,800 ha and in Germany less than 1,000 ha (Thrän and Seiffert, 2008). In general, short rotation coppice (SRC) plantations consist of fast growing trees or shrubs and are characterized by higher wood productivity in time and space than conventional cultivated forests, due to high juvenile growth rates of the trees. SRC plantations are mainly grown for producing wood fuel for heat and power production. The most important tree species grown in European SRCs are willow, poplar, aspen (including hybrids) and robinia, which all are characterized by fast juvenile growth, often with the capacity for asexual reproduction and an ability to regrow from rootstocks or stools. The plantations are established at high densities on arable land in spring and harvested in winter during vegetation dormancy when the ground is frozen. Prior to the plantation establishment, chemical or mechanical weed control is needed to minimize competition for resources and thereby allow for vigorous growth of the planted crop (Larsson et al., 2007). In many sites, especially in Central Europe, fertilization is not needed if the plantation is established on former arable land. When plantations are fertilized with sewage sludge, which is common in Sweden, the plantations act also as vegetation filters.

The demand for wood as a renewable resource for energetic use is currently increasing due to increasing energy use, the decline of fossil fuels and increasing energy prices. Further arguments for increased biomass demands include global environmental problems related to climate change in connection with CO_2-emissions and political requirements. As a result, demand is expected to continue to increase in the future. The cultivation of fast growing trees on agricultural land is a viable alternative for the production of renewable resources, particularly because these trees can achieve high biomass yields with relatively low input of nitrogen fertilizer and are regarded as efficient nitrogen users (Karp and Shield, 2008). However, knowledge is scarce about the effects of SRC plantations on the environment. The objective of this paper is to give an overview of the current state of knowledge about phytodiversity in SRC plantations and to derive recommendations for phytodiversity management in SRC stands. The overview is based on a survey of the literature mostly from Europe.

2 Establishment and management of SRC plantations

Rotation times and planting densities

In central Europe, there are currently three recognised kinds of rotations. In mini-rotation, which is the main cultivation method for willow, the trees are harvested after two or three years. The tree density is 16,000 to 20,000 per hectare. Midi-rotation takes four to six, at most ten years with a tree density of 6,000 to 9,000 per hectare and is often used for poplar. The third rotation type is maxi-rotation, suitable for trees like aspen, sycamore, basswood, mountain ash as well as alder and takes 10 to 20 years with densities between 1,500 and 3,000 trees per hectare (NABU, 2008). These data are only approximations because of the strong dependence of growth rate and adequate planting densities on site conditions.

Site preparation

Because of the enormous quantity of other seeds in the soil of agricultural fields and the weak competitiveness of young willow and poplar plants, action has to be taken to facilitate a successful establishment of SRC crops (Larsson et al., 2007; Stoll and Dohrenbusch, 2008). There are chemical and mechanical measures for preparing a field. For economic reasons chemical treatment is recommended in most cases before establishing a SRC plantation (Boelcke, 2006; Stjernquist, 1994), although sometimes only mechanical methods are used (Sage, 1998). However, the options of mechanical treatments have not yet been fully explored (NABU, 2008). For creating optimal conditions it is common practice to plough or grub up to 30 cm depth and harrow afterwards like in conventional agriculture. Treatment is recommended in autumn for cohesive soils whereas spring is the best time for more loose soil, so that already germinating seeds can be ploughed in (Larsson et al., 2007; Röhle et al., 2008). It is common to apply a broad-spectrum herbicide after ploughing the field in autumn (Boelcke, 2006; Burger et al., 2005; Fry and Slater, 2009; Schildbach et al., 2009). In spring, before planting, the field is grubbed (Schildbach et al., 2009), ploughed (Burger et al., 2005) or harrowed (Boelcke, 2006; Burger et al., 2005). Application of a pre-emergence herbicide is often recommended (Boelcke, 2006; Burger et al., 2005; Fry and Slater, 2009).

Undersown crops to repress ground vegetation are not recommended, because these would be strong competitors to the planted sprouts (Boelcke, 2006). Thus, experiments with undersown clover resulted in increased sprout mortality, sites with no treatment had a reduced crop growth rate, whereas herbicide application directly before or after sprout insertion stimulated crop growth (Wolf and Böhnisch, 2004).

If there is good nutrient supply from former land use, nutrient fertilization is not needed in the establishment year (Boelcke, 2006; DEFRA, 2004; Fry and Slater, 2009; Larsson et al., 2007). Annual nutrient extractions are low (Boelcke, 2006) and the crop is harvested in winter, when the trees are defoliated. The main part of nutrients is allocated to the leaves and therefore remains in the field after leaf abscission (Schildbach et al., 2009).

In comparison with conventional farming practices, SRC plantations require substantially less pesticide and herbicide treatment (Ledin, 1998; Perttu, 1998). Species composition is particularly strongly affected by herbicides applied during the establishment phase due to its impact on competition. This can cause long-term changes in the local species diversity. Herbicide application in mature stands has the potential to remove nearly all ground flora (Gustafsson, 1987), but in most cases herbicide applications are not necessary after establishment of commercial SRC plantations (Larsson et al., 2007).

Mechanical treatment like ploughing, harrowing or grubbing influences species composition as well, with the time of treatment playing an important role. The seeds from the surroundings influence the field strongly when cultivated in autumn, whereas this seed spread is insignificant in spring. Due to the fact that plant species are either spring or autumn-germinating, species of one of these germination types will be supported depending on whether soil cultivation is arranged early or late in the season (Gustafsson, 1987).

Many hardwood species (such as willows and poplars) are poor competitors in the juvenile plant stage when grown in a non-native environment. The poor competitiveness of the hybrid species grown in most commercial plantations makes weed control an extremely important management action, but implies also that the planted hybrids are unlikely to invade the surrounding areas and affect biodiversity (Weih, 2008a).

Sewage sludge

In Sweden it is common to use sewage sludge as fertilizer in willow SRC plantations. The practice may solve a waste problem, but is debated, because of environmental concerns (Dimitriou et al., 2006; Hasselgren, 1999). The sludge is normally dewatered and applied in spring after winter harvest every 3 to 5 years. Nutrient losses and leakage to the groundwater zone are reduced by applying sludge to an actively growing crop instead of bare soil (Hasselgren, 1998; Hasselgren, 1999). Hasselgren (1998) recommends application of 5 tonnes dry substance of sewage sludge per hectare per year. This amount should produce no adverse effects on soil, groundwater or vegetation. Sludge is also mixed with wood-ash from district heating plants (Dimitriou et al., 2006). Via irrigation of nutrient-bearing water such as wastewater from households, collected run-off water from farmlands and leachate of landfills, plantations can be used as vegetation filters for treatment. It is possible to locate plantations as buffer strips for capturing the nutrients in passing run-off water (Berndes et al., 2008).

Sludge application as a fertilizer may influence the ground vegetation and has been reported to affect ground vegetation cover (Hasselgren, 1999), but very little knowledge of sludge application on phytodiversity is currently available.

3 SRC effects on vegetation

Species composition

Species composition in SRC stands depends very much on light intensity which is highest in young plantations due to the lack of canopy closure. Light intensity is also dependent on the planted tree species and greatly influences the development and composition of the ground vegetation. For example, species that demand large amounts of light and nutrients, along with mild temperatures, typically colonize the plantations in the early stage (Delarze and Ciardo, 2002), in which the ground vegetation is dominated by annuals (Delarze and Ciardo, 2002; DTI, 2004; DTI, 2006). As a consequence of increasing canopy closure, radiation and temperature decrease, ground vegetation shifts from the initially ruderal and pioneer species towards woodland species (Britt et al., 2007; Delarze and Ciardo, 2002; Kroiher et al., 2008), and from annuals and biennials towards perennials (DTI, 2004; DTI, 2006). These changes are in accordance with the typical succession of dominant vegetation, i.e., short-lived species are usually more common early in succession whilst long-lived species usually dominate at later stages (Townsend, 2003). The shift from light demanding to shade tolerant species is likely to occur at some degree even after harvest, but has not yet been thoroughly investigated due to a lack of long-term surveys (NABU, 2008). Thus, DTI (2004) found that recently established SRC plantations are dominated by low vegetation cover dominated by annual species characteristic for disturbed ground, whereas plantations cutback after one year growth showed a higher vegetation cover, although still including a high portion of annuals. In contrast, Fry and Slater (2009) recorded almost equal proportions of annuals (34 %), short-lived perennials (39 %), and long-lived perennials (35 %) in the establishment year of willow SRC plantations grown on former grassland sites, where most of the species were typical of arable habitats or areas of recently disturbed ground. In year one and two they found a decrease of annuals, whereas short-lived perennials increased in the first year and remained constant in the second year. The proportion of the long-lived perennials increased slightly, but not significantly, and did not return to dominance like it had been the case before the establishment of the SRC plantation.

In many cases, only few species with regional conservation status are found in SRC plantations (Britt et al., 2007; DTI, 2006; Gustafsson, 1987; Vonk, 2008; Weih et al., 2003). Some rare species can occasionally be found in older stands (Gustafsson, 1987). Half of the willow plantations Gustafsson (1987) surveyed in southern Sweden were dominated by ruderal species like *Cirsium arvense, Galeopsis tetrahit* and *Urtica dioica*. *Urtica dioica* and grasses dominated the ground vegetation of 21 poplar plantations in southern and central England surveyed by Britt et al. (2007). On a plantation with poplars, hybrid aspen and willows in Bavaria (Germany), Heilmann et al. (1995) recorded five years after establishment mainly species typical of agricultural weed communities: 54 % of the species were perennial, among these mainly grasses, plus 6 % woody species as well as 40 % ephemeral species. This composition suggests the relatively undisturbed development of the vegetation (Heilmann et al., 1995).

The few rare or endangered species occasionally found in SRC plantations are predominantly light demanding pioneer species recorded in the first years of a plantation and disappearing with increasing plantation age. For example, Weih et al. (2003) found not a single rare species in 21 young poplar stands grown in Sweden, but Vonk (2008) found in a Dutch survey the orchid *Epipactis helleborine*, which has conservation status. Kroiher et al. (2008) recorded a higher number of rare species in SRC plantations in northern Germany: six out of 77 identified vascular plants are on the Red List of threatened species. These species have their main distribution in nutrient-poor habitats (Kroiher et al., 2008). Also a poplar plantation in Switzerland hosted many rare species: 18 out of more than 220 recorded species were on the Red List, for example *Ranunculus sceleratus*, *Carduus crispus* and *Carex riparia* (Delarze and Ciardo, 2002). The relatively high occurrence of rare species is probably related to the great tree distances at this Switzer site and the resulting favourable light and temperature conditions. The number of Red List species declined with increasing canopy closure of the poplars after two years, implying that the shortening of rotation time probably supports the establishment and/or survival of endangered species (Delarze and Ciardo, 2002).

The plant colonization of a plantation occurs from the surrounding area, the soil seed bank and through living vegetative tissues like rhizomes, tillers or living roots in the soil (Gustafsson, 1987; Stjernquist, 1994; Weih, 2008a). Therefore, the former vegetation and land use plays an important role for the composition of the ground flora in these plantations. The influence of former vegetation generally decreases with increasing age, but the magnitude and temporal development of the changes differs between land uses. A willow plantation in Sweden showed similar species composition compared with the meadow that used to be on this site, and a stabilization of the vegetation occurred four years after plantation establishment (Gustafsson, 1987). In contrast, changes were much more apparent on a former peatland site, in which no stabilization was recorded four years after establishment: half of the original species had then disappeared and those species still remaining had a very low cover (Gustafsson, 1987). Apart from former vegetation, also management regime greatly influences the floral composition in poplar and willow stands, as the results by Fry and Slater (2009) demonstrated.

Ground vegetation cover

As a consequence of the necessary field preparations prior to plantation establishment (Larsson et al., 2007; Stoll and Dohrenbusch, 2008), ground vegetation is very sparse when the crop is planted. Ground vegetation cover increased in the four years studied after establishment of willow plantations in England. Vegetation cover varied between individual plots, with some plots still dominated by bare ground even years after establishment (DTI, 2004). The vegetation cover of recently planted and cutback plots of year one was 10 % on average. In the last year of this four year study the average was 45 % (DTI, 2004). In recently established willow plantations, DTI (2004) detected higher vegetation cover in plots near the plantation edge compared to the plots situated closer to the center of the plantation, while the increase in vegetation cover over time tended to be more pronounced at the edges of the plots than in the interior.

According to calculated Evenness, older willow plantations show a higher heterogeneity than younger ones (Fry and Slater, 2009). The total number of species covering > 10 % increased throughout the three-year growth cycle, whereas the number of grasses covering > 10 % stagnated in the second year and decreased thereafter as fewer more competitive species like *Holcus lanatus* and *Dactylis glomerata* became increasingly abundant (DTI, 2004).

After harvesting a plantation, the cover of the ground flora increases (Heilmann et al., 1995), as it is expected from succession theory (Townsend, 2003). Still, vegetation cover is lower in willow plantations cutback after one year of growth than in recently planted stands (DTI, 2004). Although radiation would be expected to be one of the most important drivers for vegetation cover, Gustafsson

(1987) found no correlation between willow cover, ground vegetation cover and species number in southern Sweden. These plantations were not older than three years. It is expected that a longer rotation time would reduce the vegetation cover (Gustafsson, 1987) and species number, as was found by Heilmann et al. (1995).

Ground vegetation cover is also dependent on the planted crop. Different species and genotypes have different growth habits and are differently affected by habitat conditions. There is an increasing gradient in ground vegetation cover from poplar to hybrid aspen and willow due to differences in radiation climate resulting from different leaf phenology, growth habit and biomass of the trees (Heilmann et al., 1995).

Stoll and Dohrenbusch (2008) showed the influence of the former land use on biomass production and found the productivity of the ground vegetation to be higher on former grassland than on former arable land.

Species richness

Species numbers ranging from around 10 to more than 220 were recorded in willow or poplar plantations in Sweden, the Netherlands, the UK, Germany and Switzerland, with a trend to increasing species richness with decreasing latitude (Burger et al., 2005; Delarze and Ciardo, 2002; DTI, 2004; Gustafsson, 1987; Heilmann et al., 1995; Vonk, 2008; Weih et al., 2003). The number of species usually increases in the first two years after establishment and decreases thereafter with increasing age of the plantation (Delarze and Ciardo, 2002; DTI, 2004; Gustafsson, 1987; Wolf and Böhnisch, 2004). This pattern probably can be attributed to deteriorating light conditions on the ground over time, so that the conditions become increasingly similar to traditional forests. Especially the number of endangered species decreases over time (Delarze and Ciardo, 2002). DTI (2004) found more than six times more species in plantations cut back after one year of growth compared to recently planted ones.

A positive edge effect was recorded for species number in a similar way as was seen for ground vegetation cover (DTI, 2004). For example in willow and poplar plantations grown in southern and central Sweden, species numbers decreased with distance from the edges (Augustson et al., 2006; Gustafsson, 1987; Weih et al., 2003). A generally positive edge effect on species numbers was also found during the first two years after plantation establishment in UK investigations of willow stands grown on former arable land or grassland pasture (DTI, 2004; DTI, 2006). However, in the third and fourth year there was no relationship found between species number and distance from the edge, and a great proportion of total species numbers were only detected at the edges of the plots (DTI, 2004; DTI, 2006).

The above mentioned studies suggest that colonization occurs predominantly from the surrounding landscape, so that the location of a SRC plantation in the landscape context is critical for species numbers. The more diverse the surrounding landscape, the more species can colonize the plantations and thereby increase biodiversity (Weih, 2008a). Furthermore, the former vegetation normally influences the composition of the ground flora, especially in early stages (Gustafsson, 1987).

Plantation size and shape also seem to be important for biodiversity, with higher species numbers recorded at the edge of a plantation than within it (Augustson et al., 2006; DTI, 2004; DTI, 2006; Gustafsson, 1987; Weih et al., 2003). Stands of equal size have longer edges if they are long and narrow than if they are round or square. On one hand, longer edges support the immigration of seeds from the surrounding landscape, for example via wind and animals. On the other hand, round and squared stand shapes benefit the diffusion within the plantation (Gustafsson, 1987).

The more heterogeneous and species rich the surroundings are, the more species are likely to reach the plantation and establish there, suggesting a small-sized structure to favour species diversity (Gustafsson, 1987; Weih, 2008a). For example, on eight and nine year old poplar stands in Germany, 0.3 hectares contained almost all of the 38 species found in the whole plantation, and it was concluded that 1 ha of a homogeneous plantation hosts all the species found in larger plantations (Lamerstorf et al., 2008; NABU, 2008). Furthermore, large monocultures have been speculated to be more fragile to diseases than mixed stands. DEFRA (2004) therefore recommends a mix of species clones in a stand to minimize negative impacts of Melampsora rust damage, which is the most common fungal disease in willows and poplars.

Land-use effects on local and landscape scale

Species number in willow and poplar SRC plantations has frequently been reported to be higher than in conventional agricultural fields (Augustson et al., 2006; Britt et al., 2007; Burger et al., 2005; DTI, 2004; DTI, 2006; Fry and Slater, 2009; Heilmann et al., 1995; Perttu, 1998; Wolf and Böhnisch, 2004). Especially in the first year after establishment of a poplar plantation, its species number can be much higher than in comparison with an intensive cropland (Burger et al., 2005; Wolf and Böhnisch, 2004). On SRC fields in Bavaria, the recorded species numbers were up to ten times higher than in adjacent agricultural fields (Burger et al., 2005). Not only species number, but also vegetation cover (DTI, 2004) and floristic heterogeneity (Weih et al., 2003) have been reported to be higher in SRC fields than in arable land. In comparison to arable fields and grassland fallow, willow and poplar plantations have been shown to contain more species than arable land and higher or similar species richness to grasslands (DTI, 2004; Fry and Slater, 2009; Heilmann et al., 1995).

In comparison to old-growth mixed deciduous forests, species richness of young poplar plantations was similar or lower (Weih et al., 2003). In line with this observation, Schmidt and Gerold (2008) suggested that SRC plantations are closer to a natural state than conventional cropland, but cultivated forests are the vegetation types closest to nature in this comparison.

Considering habitat demands, one would expect differences in species composition of SRC plantations and arable fields. The differences in habitat demand are reflected by the Ellenberg values and the species in poplar plantations are characterised by low light, pH and nutrient demands, but high moisture demand (Britt et al., 2007).

The contribution of SRC species composition to gamma diversity at the landscape scale depends strongly on landscape structuring, land-use variability and habitat diversity (NABU, 2008). Positive effects of SRC plantations were found in agrarian regions with uniform landscape structures where SRC sites are reputed to be a source for plant species richness (Gustafsson, 1987; Weih et al., 2003). In an area of northern Germany with quite uniform land-use patterns, Kroiher et al. (2008) found that even small SRC stands (1,600 m² area) contained 8 % to 12 % of the species number of the surrounding landscape (25 km² area) which is a considerable share when considering their limited extent.

4 Recommendations to manage phytodiversity in SRC plantations

As shown in the previous sections, there are many factors influencing species cover, species richness and the type of species occurring in SRC plantations. In general, the location for establishing a SRC plantation should be chosen carefully and consider both economic and environmental aspects. Depending on the location, SRC plantations can have positive as well as negative effects on biodiversity (Weih, 2008a). If being established in an area dominated by agriculture (Fry and Slater, 2009; Gustafsson, 1987) or coniferous forests, SRC plantations form an additive habitat and can increase regional structural diversity, whereas it is not advised to establish energy forests in areas with rare species, rich fens, forested wetlands and edge-zones bordering to lakes and rivers as well as on undisturbed peat bogs (Gustafsson, 1987). Cook et al. (1991) point out that the current and world-wide trend in favour of increased biofuel plantations puts wetland habitats at a high risk to become prime candidates for conversion to plantations. Due to their wetness, these habitats could be particularly interesting for SRC plantations and the conversion would entail a loss of biodiversity. From a nature conservation point of view, arable land is suitable for establishing a SRC plantation, if plantation sites do not affect adjacent or nearby protected areas, and if the establishment has no negative influence on endangered species or disturbs wildlife corridors (Schmidt and Glaser, 2009).

In summary, biodiversity in SRC plantations can be favoured by consideration of a few guidelines (after Weih (2008b)):

- Avoid areas with protection status for nature conservation and/or cultural heritage.
- Avoid very large plantation sizes – plant several smaller plantations instead.
- Locate the plantations close to existing native woodlands and/or incorporate 'islands' of native trees within large plantations.
- Leave buffer zones without any crop or with native vegetation at the edges of plantations.
- Plant several varieties (preferably of different gender) within the same plantation; different varieties may be planted in sections or parallel stripes in order to facilitate harvest actions.
- Apply chemical weed control only during plantation establishment.
- Do not apply more nutrient fertilizer than the biomass crops demand during a growing season.
- Try to plan harvest actions to be performed only when ground is frozen.
- Harvest parts of plantations in different years.
- Locate, design and manage the plantations in such a way that they maximize variation in habitat type and landscape.

5 Conclusions and future perspectives

Due to the expected increase of SRC plantations, more knowledge about how these plantations influence phytodiversity is urgently needed to enable sustainable management of SRC plantations and to gain extra benefits for the environment that might occur if the plantations are managed in a suitable way. Even though there has been research on the influence of SRC plantations on phytodiversity, there are still a lot of open questions which need to be clarified to enable the best management. Especially long-term studies are at this point of time still lacking.

As shown above, species composition and species number change over time in SRC plantations. Therefore, further research should focus on the question of what is an appropriate rotation duration to support species richness or endangered species. There cannot be a generalised answer to this question, because different plantation species as well as environmental factors play an important role in ground vegetation development.

Colonization of the SRC plantations happens not only from the surroundings, but also from the soil seed bank and through living vegetative tissues and is therefore dependent on the former usage. It should be clarified where the largest proportion of species in the SRC plantations comes from, in order to discover how strong the former use influences species composition of the plantations. Another interesting aspect is how long the influence of the former use lasts, which can be shown by long-term research.

Different tree species and clones are planted for energy crops, but little is known about how the different crops influence phytodiversity and about the variations concerning species composition and vegetation development on the ground over time.

Further research is also needed to clarify the contribution of SRC plantations to species diversity at the landscape scale, and correspondingly the influence of landscape factors on SRC plantations. Important variables include the crop species, the plantation age and the surrounding uses,. In this context it is very important to shift the experimental field design from the classical focal patch approach which focuses on the comparison of single patches or one patch and its surrounding to a mosaic-level approach - i.e. multiple sample patches have to be investigated in a given landscape mosaic. With this approach it is possible to address the question how on-site biodiversity is affected by the emergent properties of the surrounding landscape mosaic. These properties include the extent of habitat, composition of the landscape mosaic and spatial configuration of its elements. Thus, the quantification of the relative importance of determinants of species richness is an important task for biodiversity research. Further, this approach would allow the quantification of the specific contribution of SRC plantation patches to landscape species richness. Until now, this kind of approach does not exist for SRC biodiversity assessments. There is however experience from other investigations in agricultural environment to build on (Simmering et al., 2006; Wagner et al., 2000; Wagner and Edwards, 2001; Waldhard et al., 2003; Waldhard et al., 2004).

Using sewage sludge for fertilizing SRC plantations and at the same time solving a waste problem seems to be win-win situation as the planted crop acts like a vegetation filter. But in what manner sewage sludge application influences phytodiversity has not been analyzed until now.

SRC plantations can have clear benefits for biodiversity, but negative effects are also possible. As much knowledge as possible should be gathered about how the different factors influence phytodiversity for supporting sustainable management of energy wood plantations.

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Animal diversity on short-rotation coppices – a review

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Abstract

The animal diversity on short-rotation coppices (SRC) has not yet been investigated adequately. Most research conducted up to now did focus on birds and ground beetles. Their biodiversity, equated with species richness, differs considerably. Diversity of breeding birds is higher in SRC than in agricultural cropland, but generally lower than in forest ecosystems. Diversity of ground beetles is higher in arable fields than in SRC. In general the portion of forest species is lower in SRC than in typical forest habitats but rises with increasing age of stands.

The animal diversity depends on various environmental factors. These include the surrounding landscape, the shape and size of the plantation. For example, small and oblong SRC are more favourable due to the edge effect, for example. Besides these landscape ecological parameters, there are other important factors. Willows contain both a greater diversity and higher abundance in most animal groups than poplars. Some birds even prefer certain clones when selecting a nesting site.

The habitat structures of a SRC change as its age increases and the composition of the biocoenoses also changes as a result.

The biodiversity in SRC is enhanced most significantly by the structural richness within the poplar or willow blocks and in the peripheral areas. As a result, some animal species of conservational value can find a suitable habitat in SRCs. The increasing cultivation of SRCs can lead to a slight increase in biodiversity in cleared agricultural landscapes, but to significant adverse effects in landscapes with high conservational value. Further research is required, especially regarding species-rich invertebrate groups.

Keywords: Woody biomass, biodiversity, birds, ground beetles, environmental impacts, short-rotation coppices

Zusammenfassung

Tierdiversität auf Kurzumtriebsplantagen – eine Übersicht

Es liegen noch keine umfassenden Untersuchungen zur Biodiversität der Tiere auf Kurzumtriebsplantagen (KUP) vor. Am besten untersucht sind die Vögel und die Laufkäfer. Ihre Biodiversität, gemessen als Artenreichtum, ist unterschiedlich ausgeprägt. Während die Brutvögel KUP artenreicher besiedeln als z. B. Äcker, aber hier eine geringere Artenzahl als bspw. in Laubwäldern aufweisen, treten Laufkäfer auf Äckern in höheren Artenzahlen auf. Generell ist der Anteil an Waldarten in den Artenspektren gering, nimmt aber mit zunehmendem Alter der KUP zu. Die Tierartenvielfalt auf einer KUP hängt von verschiedenen Umweltfaktoren ab. Dazu gehört die umgebende Landschaft, die Flächenform und die Flächengröße. So sind zum Beispiel lange rechteckige Anlagen wegen des Randeffektes günstiger als quadratische. Neben diesen landschaftsökologischen Parametern gibt es weitere wichtige Faktoren auf der Bestandesebene. So weisen z.B. Weiden größere Artenzahlen und Individuendichten bei vielen Tiergruppen auf als Pappeln, und manche Vogelarten bevorzugen bestimmte Gehölzklone bei der Nistplatzwahl.

Die Habitatstrukturen einer Kurzumtriebsplantage ändern sich mit zunehmendem Alter und damit einhergehend ändern sich die Zusammensetzungen der Lebensgemeinschaften.

Begleitstrukturen und Strukturreichtum innerhalb der Pappel- oder Weidenblöcke können die Diversität auf KUP am stärksten erhöhen. Durch Begleitstrukturen und junge Offenlandstadien können naturschutzfachlich wertvolle Tierarten im Einzelfall gefördert werden.

In ausgeräumten Landschaften kann der Anbau von KUP eine biodiversitätsbereichernde Wirkung ausüben; in Landschaften mit hohem naturschutzfachlichen Wert kommt es jedoch zu gegenteiligen Effekten.

Es besteht noch Forschungsbedarf, gerade bei den artenreichen wirbellosen Tiergruppen.

Schlüsselwörter: Energieholz, Biodiversität, Vögel, Laufkäfer, Umwelteinflüsse

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1 Introduction

Energy use projections for North America and Europe predict that biomass will be an important source of renewable energy in the coming decades – and short-rotation woody plants will be the prime source of this biomass (Volk et al., 2004; Reeg et al., 2009). Short-rotation coppices (SRC) are production systems for generating wood in short time periods (Murach, 2009; Knust, 2009). For this, hybrids of willows (*Salix*) and poplars (*Populus*) that have been selected on the basis of their growth performance and ability to resist pests are planted in high density on areas of agricultural land (Knust, 2009).

But while technology is rapidly advancing to utilising woody perennials there are a lot of questions concerning environmental impacts of biomass and biofuel production (Firbank, 2007; Johnson et al., 2007; Porder et al., 2009; Rode, 2005). Thus, as more and more short-rotation coppices are planted, the question as to their influence on the biodiversity is also raised (Firbank, 2007; Schmidt & Glaser, 2009).

Nowadays, biodiversity is one of the overriding principles in nature conservation and land use. At the Convention on Biological Diversity in Rio (1992), the contracting states agreed to "protect and enhance biodiversity" within their national borders (Groom et al., 2006; Haughton et al., 2009). There are conflicts between biodiversity conservation and human land use activities, however, particularly in agricultural landscapes (Henle et al., 2008). An important aspect relevant to the future acceptance or rejection of the SRC by the general public will be whether they increase or reduce a landscape's biodiversity. The following is an attempt to summarise the current knowledge concerning animal diversity in SRCs.

2 Animal species diversity

Species are the fundamental units of biodiversity (Prance, 1995) and the number of animal species is the highest of all taxonomic groups (Wilson, 1988; Trevelyan & Pagel, 1995). The vast majority are invertebrates (Wilson, 1988), but the diversity of invertebrates is underestimated and poorly examined. Studies of animal diversity are complex, but surprisingly high numbers of species were discovered when such studies were carried out on agricultural sites. For example, more than 900 insect species have been found in monocultures of cereals which are supposed to represent very species-poor habitats (Potts, 1977 in Schneider, 2006). New insect species continue to be discovered on European agricultural land (as an example see the Sciaridae: Diptera compiled by Menzel & Schulz, 2007). Even less knowledge exists with regard to the diversity of soil animals. The "enigma of soil animal species diversity" cited by Anderson (1975) still applies today.

Vertebrates

In the overall diversity of animals in a SRC, as in other ecosystems, vertebrates make up only a small fraction. Most research has been conducted into the diversity of birds in SRCs. Tangible data and meaningful overviews from Great Britain (Sage & Robertson, 1996; Sage et al., 2006; Anderson et al., 2004), Sweden (Berg, 2002), Germany (Jedicke, 1995; Liesebach & Mulsow, 1995, 2003; Gruß & Schulz, 2008) and the USA (Christian et al., 1997; Christian et al., 1998; Dhondt & Sydenstricker, 2000; Dhondt et al., 2004; Dhondt et al., 2007) already exist on this subject. Londo et al. (2005) dealt with the habitat potential and theoretical avian diversity of willow SRC in the Netherlands.

The cited number of bird species in SRC differs from 8 to 60 species. Different bird species are associated with different age classes of SRC. The abundance of birds in SRC has also been shown to be linked with coppice stem or planting density and with increased weediness (Sage et al., 2006). But the different numbers of species are due to many further factors, such as variety of areal sizes, intensities of management, landscape context and regional species pool. These parameters will be discussed in the following chapters.

Up to now, very little research has been conducted on mammals in SRC. Species observed in SRC plantations in England included 17 mammals (compiled by Rowe et al., 2007) and trapping suggested that SRC provided a more attractive habitat for small mammals than arable land, with older coppices being more attractive, nevertheless it still represents a poorer habitat than hedgerow and scrub land (Coates & Say, 1999, in Rowe et al., 2007). Christian et al. (1997 and 1998) investigated small mammals in poplar SRC in USA. Shrews appear in well-vegetated patches on both young and older poplar plantations (Christian et al., 1998), young poplar plantations tend to be dominated by single rodent species (Christian et al., 1997). However, the composition of bird and small mammal fauna on SRCs varies among study sites and regions. Furthermore larger herbivores such as deer and rabbit can play an important role due to their browsing damage on trees and shrubs (Christian et al., 1998; Rowe et al., 2007; Helbig & Müller, 2008; Schulz et al., 2008a; Helbig & Müller, 2009). Bergstrom & Guillet (2002) suggest that although large herbivores are often supposed to damage SRC plantations, e. g. by barking or browsing, SRC could also be viewed as a resource for deer and hare in terms of the game value of these species. For many larger bird and mammal species, SRC with small area sizes merely represent partial habitats within larger home ranges.

Overall, it becomes clear from the cited works that the bird and mammal communities of SRCs are made up of species typically found in open land and woodland. Christian et al. (1998) did not observe any bird or mammal species on *Populus* plantations that did not occur elsewhere in the region. The most abundant bird species and small mammals on hybrid plantations are habitat generalists. Most of the bird species are regionally abundant, widespread and capable of using a wide variety of breeding habitats (Christian et al., 1998; Gruß & Schulz, 2008; Jedicke, 1995).

Invertebrates

Most studies of animal species diversity in SRCs have dealt mainly with vertebrates and left invertebrates largely neglected. The research was focussed on insects that directly populate poplar and willow plants and trees. Poplars and willows act as host to a large number of insects. This has been known for some time (e.g. FAO 1979), particularly since some insect species can emerge as pests. These include defoliating and boring species of the beetle (Coleoptera) and butterfly (Lepidoptera) orders, but also gallformers and sucking insects of other insect orders, such as sawflies (Hymenoptera: Symphyta), gallmidges (Diptera: Cecidomyiidae) and aphids (Sternorrhyncha: Aphidoidea) (Christersson et al., 1992; Gruppe et al., 1999; Helbig & Müller, 2008). Beetle species belonging to the *Phratora* and *Chrysomela* genera can cause major damage (Coyle et al., 2002; Helbig & Müller, 2008 and 2009). Leaf beetles (Chrysomelidae), which occur in remarkably high numbers in German poplar plantations, include *Chrysomela populi* (Helbig & Müller, 2008; Schulz et al., 2008b) and *Chrysomela tremulae* (Helbig & Müller, 2008).

Due to the overwhelming diversity of invertebrates, investigations have to be limited to individual indicator groups. Up to now, earthworms (Lumbricidae; Makeschin et al., 1989; Makeschin, 1994), web-spinning spiders (Blick & Burger, 2002; Blick et al., 2003) and butterflies (Britt et al., 2007; Haughton et al., 2009) have been investigated in SRCs. In other studies, epigeic arthopods are surveyed on the order and family level (Britt et al., 2007; Liesebach et al., 2000; Schulz et al., 2008b), or guilds such as flower-visiting insects (Reddersen, 2001) or as canopy-invertebrates (Sage & Tucker, 1997). The invertebrate group studied in greatest detail in SRC are ground beetles (Carabidae) (Allegro & Sciaky, 2003; Liesebach & Mecke, 2003; Britt et al., 2007; Schulz et al., 2008b; Lamersdorf et al., 2008). Species numbers ranging from 10 to 43 were discovered on SRC. These numbers are of little significance, however, if they are not related to adjacent habitats and to the various influencing factors. This will be enlighted in the following chapters.

3 Comparison with other forms of land use

Haughton et al. (2009) summarise that compared with cultivated areas of energy crops such as oilseed rape, SRCs have particular advantages as bioenergy sources:

- they are not food crops
- there is no annual cultivation cycle
- they achieve rapid growth with the potential to produce large yields with low fertilizer and pesticide requirements
- there are only a few disturbances in the growing period
- harvesting is carried out in winter and causes therefore less disturbance
- there is a greater richness of spatial structures.

This has an overall positive effect on the biodiversity. Animals that depend heavily on the vertical structure, such as many breeding birds, can benefit from the growth characteristics of SRC. This is particularly the case when SRCs are planted in an agricultural landscape with little structural diversity. Many insect groups benefit from the decreased use of pesticides in SRCs and earthworms e.g. are favoured by the longer soil rest period, for instance (Makeschin, 1989; Makeschin, 1994). Liesebach et al. (2000) demonstrate that a higher diversity of epigeal invertebrates is present in a SRC than in a barley field. Britt et al. (2007) found a greater abundance and diversity of butterflies (Lepidoptera) and a higher number of springtail species (Collembola) in hybrid poplar fields than in agricultural fields. Regarding arachnids, Blick & Burger (2002) and Blick et al. (2003) found more individuals and species of arachnids in German SRCs than on nearby agricultural crop land. Blick et al. (2003) found three times as many species of each of the groups Aranea, Opilionida and Pseudoskorpiones, and a total of 85 species in an SRC, compared with 72 species on the agricultural crop land.

But most research has been conducted on breeding birds and ground beetles (see chapter 2). According to this they will be considered in greater detail in the following.

Birds

Results obtained in the USA, the UK, and Sweden confirm that bird abundance and diversity is generally high in short rotation coppices (Anderson et al., 2004; Berg, 2002; Dhondt & Sydenstricker, 2000; Dhondt et al,. 2007; Sage & Robertson, 1996). Christian et al. (1997) found a greater avian species richness in SRCs in the north of central-western USA (Minnesota, Wisconsin and South Dakota) and more individual breeding birds than on agricultural crop land, but fewer than in woodland. Liesebach & Mulsow (2003) found more birds in a German SRC than in surrounding fields and fewer than in a neighbouring spruce forest. After analysing in depth the numbers of breeding birds in swedish willow SRCs, Berg (2002) came to the conclusion that bird species-richness in the SRCs was high compared with open farmland sites dominated by other crop-fields, but lower than that in forest edge habitats. Thus, SRCs also perform better than other biomass crops and arable crops when species-richness of breeding birds is considered only. Regarding to the habitat potential for endangered species or more specialized birds SRCs are generally of lesser value. In comparison to open grasslands, fallows or even arable lands they offer a considerably lower habitat potential for many avifaunistic elements of open lands – especially for demanding species (Gruß & Schulz, 2009; Rowe et al., 2007; Sage et al., 2006)

Ground beetles

A different picture emerges with regard to the ground beetles (Coleoptera: Carabidae) sampled by pitfall trapping, however. They are species-poorer in some SRCs than on agricultural crop land. Britt et al. (2007) found significantly more ground beetle species in arable fields than in poplars on english sites. Also, fewer species of ground beetle were found in various North German SRCs than on the neighbouring intensively farmed agricultural crop land (Liesebach & Mecke, 2003; Lamersdorf et al., 2008; Brauner & Schulz, 2009).

In the course of the investigations in Northern Germany Brauner & Schulz (2009) detected that particularly during the first years of a SRC species communities of persistent ruderal sites and species of arable weed societies reached in SRCs the greatest shares in species richness and abundances. The fraction of typical forest species rises significantly with increasing age of the stands. Thus, generally those species are only appearing in stands with longer rotation periods (see chapter 4.1).

Besides the age of stands of SRCs the distance to other forest patches might be of crucial importance for the number of occurring ground beetle species. Hence in larger long-term isolated forest patches fewer Carabids typical for forests were found than in smaller and younger forest patches that were less isolated (Gruttke, 1997).

The most in-depth research into ground beetles was conducted by Allegro & Sciaky (2003). They studied ground beetle assemblages for about ten years in different age groups of poplar stands distributed over a vast territory of the Po Valley in Italy. The number of species found in the different poplar stands varied between 10 and 39 without any correlation with factors such as habitat typology or age of the poplar stand (1 year old to 10 years old). But they captured 46 species in a maize-cultivated area (Allegro & Sciaky, 2003). In North-Eastern Germany Brauner & Schulz (2009) recorded the highest number of species in the recently planted or harvested SRCs. These are comparable to the numbers that could be found in the adjacent arable land.

Heterogeneity and spatial structures play a critical habitat role on SRCs. Many intensively managed poplar plantations have little ground vegetation, as a result of chemical or mechanical weed control early in the rotation cycle and of canopy shading later in the rotation (Christian et al., 1998). Others have gaps in the canopy that allow other plant species to grow, differences in canopy height or woody debris (Christian et al., 1998). This has strong influences on animal diversity.

Thus, it cannot be stated generally that SRCs have a positive effect on biodiversity. Instead, one has to differentiate between animal groups, between spatial structures and ecological conditions of SRCs. In addition, the respective landscapes concerned need to be considered and certain influencing factors such as age and form of the area taken into account. The following chapters will reveal how these and other factors can affect the animal diversity of SRCs.

4 Impact of environmental factors

4.1 Impact of age

It is not surprising that animal use changes with SRC age (Christian et al., 1998). Poplar and willow plantations change very quickly due to their rapid increase in height. Thus, the habitat conditions relevant to animals such as spatial structure, structural density, complexity of vegetation, shade and humidity also change. Berg (2002) measured increasing bird species numbers with increasing height of *Salix* plantations. Different bird species are associated with different age classes of SRCs (Sage et al., 2006; Jedicke, 1995; Gruß & Schulz, 2008). According to Christian et al. (1998) and Londo et al. (2005), three phases of an SRC can be identified: open area phase, shrub-like stands and tree-like forms. In the breeding bird studies carried out by Dhondt & Sydenstricker (2000), Berg (2002), Sage et al. (2006) and Gruß & Schulz (2008, 2009), the highest numbers of species and the greatest breeding density were found in the second, shrub-like development phase.

In poplar and willow SRCs of northern Germany the largest numbers of species (15 to 19) and individuals (45 breedingpairs/10 ha) were to be found in the 2 to 5 year-old plots but only 3 to 5 species (3.5 to 19 bp/10 ha) in the open area phase (Gruß & Schulz, 2009)

In the open area phase, after planting or harvesting of the SRC, despite the low number of species, sometimes species with conservational value were found. For example the skylark (*Alauda arvensis*) was observed to breed on recently harvested SRC in Britain (Sage et al., 2006) and in Germany (Gruß & Schulz, 2009).

As the age and growth height of SRCs increase, the avian species communities change and shift from open land species to forest species. This was proven not only for breeding birds (Dhondt et al., 2007; Sage et al., 2006; Gruß & Schulz, 2008), but also for web-spinning spiders (Araneae; Blick & Burger, 2002) and ground beetles (Carabidae; Allegro & Sciaky, 2003; Schulz et al., 2008b; Brauner & Schulz, 2009). Besides the change in species composition, the diversity of animal species declines again.

A short rotation coppice in northern Germany contained a lower number of breeding birds after 13 years' growth than in younger phases (Gruß & Schulz, 2009). Thirteen years is longer than the rotation period of 3 to 4 years common in practice, however and will not occur very often in future SRCs.

Allegro & Sciaky (2003) detected with increasing age of poplar plantations in the italian Po Valley the species spectra of ground beetles shift clearly from communities of the open areas to those of forests. A research in pine afforestations in the lignite open-cast mining area of the Lausitz (Kielhorn, 2004) indicates that numbers of species and abundances of characteristic forest species did not increase until reaching 15 years of stand age. Furthermore, the investigations of Brauner & Schulz (2009) show that only for longer rotation periods an increase of species typical for forests and their initial succession stages was evident. Here ground beetles of the forests occured in a poplar plantation of 8 years with 12 species as being 40 % of the total species amount and with 12 % of the total amount of individuals. However, these species mostly have a broader ecological amplitude.

Sixty-year-old poplar plantations in Poland represent an extreme example. Even in this case the ground beetle species numbers were lower than in the adjacent arable land (Ulrich et al., 2004) and the community contained mainly ubiquitous species with unspecific requirements. There were no habitat specialists not occurring elsewhere in the adjacent rural environments. Ulrich et al. (2004) conclude that poplar plantations neither reach species diversities of at least seminatural forests nor enhance regional species diversity.

4.2 Impact of choice of tree

Different hybrids of poplar species (*Populus* sp.) and willow species (*Salix* sp.) are selected for cultivation in short rotation coppices. Robinia (*Robinia pseudoacacia*) is increasingly used in areas of eastern Germany with low precipitation (Knust, 2009).

There are various reasons why the choice of trees affects the colonization of an SRC by animals. Because their structural richness is generally greater, blocks of willow are home to more breeding birds than blocks of poplar (Dhondt et al., 2007; Gruß & Schulz, 2008). Willow SRCs in England contained more resident and migrant songbird species than poplar SRCs (Sage & Robertson, 1996). Furthermore, the male and female flowers of the willow (*Salix viminalis*) are an important food source for bees, bumblebees and other flower visitors. Overall, willow SRCs contain more invertebrates than poplar SRCs (Sage & Tucker, 1997). As a result, willow SRC offer breeding birds more favourable conditions for nest-building and foraging (Sage et al., 2006).

The potential biodiversity of arthopods that directly populate the individual trees and plants – particularly phytophagous beetles and butterflies – can vary significantly between tree and plant genera. Kennedy & Southwood (1984) only found two species of insects on the *Robinia* genus, which is a neophyte in Europe, but 450 on *Salix.* Willows are known to provide a habitat for more arthropod species than most other trees. Brändle & Brandl (2001) compiled the numbers of insect and acarian species on various trees and plant genera and concluded that, in general, the *Salix* genus displays the greatest potential of animal diversity with 728 species, followed by *Quercus* with 699 species, *Betula* with 499 species and *Populus* with 470 species.

There are other differences within the tree and plant genera. According to Hondong (1994), 34 species of pollen-collecting wild bees populate *Salix caprea* (in southern Germany), of which nine are oligolectic, whereas only 16 species populate *Salix alba*, for example (of which five are oligolectic). In general, Thüring (2007) demonstrated, on the basis of literature reviews, a higher zoodiversity for native autochthonous tree and plant species (e.g. *Salix caprea*) than for allochthonous trees and plants (e.g. *Salix daphnoides*; summarised in Schulz et al., 2008b). On the whole, poplar species were cited as host plants for phytophagous insects less frequently than willow species (Thüring, 2007).

However, these theoretical numbers of species cannot be transferred to the willows and poplars in SRCs even in approximate terms. Much lower numbers of species of phytophagous arthropods are to be anticipated on the densely planted, young short-rotation trees with a lower richness of structure. Up to now, very little has been published on the actual colonization of SRCs by phytophagous insects – aside from the key pests studied by Sage & Tucker (1997) and Helbig & Müller (2009). Studies of the biodiversity of phytophagous insects (Coleoptera, Lepidoptera, Auchenorrhyncha) were carried out in poplar SRCs in northern Germany (Kreinsen, 2008; Kubis, 2008; Schulz et al., 2008). The species richness, which included 51 species of phytophagous beetle (Kreinsen, 2008) and 35 species of leafhoppers (Kubis, 2008), was on the low side. In addition, most species found are not associated with the poplars themselves, but instead are eurytopic or typically inhabit the grass and ground vegetation layer.

4.3 Impact of clone choice

Very few studies have been carried out into the influence of clone choice on colonization by animals. The research papers published by Dhondt & Sydenstricker (2000) and Dhondt et al. (2004) which focus on North American SRC plantations provide an initial point of reference. They report that certain bird species favour certain clones when selecting a nesting site. Dhondt & Sydenstricker (2000) found 41 % of the nests in the poplar clone S365, but only 24 % in the poplar clone NM6. This can obviously vary between bird species. For example, the American Goldfinch (*Carduelis tristis*) favours the poplar clones S25 and SA2, yet eschews NM6 (Dhondt & Sydenstricker, 2000). The American Robin (*Turdus migratorius*) behaves in the exact opposite way and favours NM6. The choice of nesting site appears to be influenced by the branching pattern of the respective clone. To increase the attractiveness for several breeding birds, Dhondt et al. (2004) therefore recommend a mix of different clones when establishing large-scale SRC plantations and not planting clones such as S301, as these are avoided by breeding birds.

Helbig & Müller (2009) also recommend polyclonal plantations, since these could minimize the damage caused by leaf-eating insects. According to Kendall & Wiltshire (1997) and Peacock & Herrick (2000), the defoliators display very diverse, species-specific food preferences, which depend on the chemical ingredients of the plants which differ among the clones.

Very little research has been conducted on the influence of different species or clones on the biodiversity of invertebrates, however. Initial observations have been made of caterpillars of various butterfly species which feed on willow and poplar hybrids in German SRCs. The butterfly species *Cerura vinula* and *Scoliopteryx libatrix* have been discovered on the poplar hybrid Max 5 (*Populus maximowiczii* x *nigra*) and the Eyed Hawk Moth (*Smerinthus ocellatus*) has been spotted on the willow hybrid Tora (*Salix schwerinii x Salix viminalis*) (Schulz et al., 2008b). Only isolated sightings were made of monophageous beetles that rely on Salicaceae, however.

Up to now, two mechanisms of action via which different willow and poplar clones can affect the biodiversity of animals are known about: by way of different structural patterns (e.g. branchings) in the case of breeding birds, and by way of plant ingredients in the case of herbivorous insects. It is also conceivable that foliation and consequently also shade have an effect on colonization by thermophile insects, for instance.

4.4 Impact of plantation size

Christian et al. (1998), Cunningham et al. (2004) and Sage et al. (2006) came to the conclusion that significantly more bird species with a higher concentration of individuals populate the periphery of the SRC and that the most obvious effect of plantation size on biodiversity is the higher proportion of edge habitat in small plantations. On large plantations lower overall bird densities were observed in plantation interiors than on edges (Christian et al., 1998). Sage et al. (2006) summarized that the interior of large SRC plots contained fewer birds than the edge-Zone (< 50 m). This ecotone effect also becomes apparent when other groups of animals are considered (Cunningham et al., 2004). From this it can be derived that under certain conditions, smaller areas of energy wood are more favourable from an animal ecology perspective, as they boast a higher proportion of peripheral areas than large-area SRCs (Londo et al., 2005). For the same reasons, elongated sites would be more preferable in comparison to quadratic sites, for instance.

This statement needs to be qualified for freshly planted and/or freshly harvested areas, however, as well as for SRCs which are highly heterogeneous due to the loss of individual blocks of plants, and which therefore feature a patchwork of trees and shrubs and open areas. Concentrations of breeding pairs in the peripheral areas and/or a decrease in density towards the center of the crop are less conspicuous here (Gruß & Schulz, 2008).

4.5 Impact of location

The biodiversity of an SRC is influenced to a large degree by the surrounding landscape. Berg (2002) emphasised the strong influence of adjacent habitats on bird community composition in the SRCs. He found major differences between bird communities depending on whether the SRC bordered onto woodland or open land, for example (Berg, 2002). In Ontario (USA) more bird species occurred on poplar plantations adjacent to both forest and open habitats than those in uniform settings (Christian et al., 1998).

The conservational significance of the surrounding landscapes also becomes apparent when initial comparisons are drawn between SRCs in western and eastern Germany. Thus, bird species on Brandenburg's Red List, such as the Corn Bunting (*Emberiza calandra*) and Wood Lark (*Lullula arborea*), which are completely absent from west German SRCs, have been spotted in freshly harvested poplar plantations (Gruß & Schulz, 2009). That might be ascribed to the stable populations and source habitats on agricultural land in eastern Germany (Flade et al., 2006).

On the other hand, SRCs are supposed to affect the biodiversity of the surrounding landscape. Planting SRCs has a positive effect on the biodiversity in cleared landscapes, but a negative effect in valuable open countryside (Sage et al., 2006). In general, Berg (2002) and Sage et al. (2006) state that the planting of SRCs in open farmland plains will have positive effects on bird diversity by increasing the structural diversity of the landscape. In England Haughton et al. (2009) demonstrate that the abundance of total butterflies and most butterfly families was significantly greater in field margins surrounding SRC willow than in field margins of arable crop. However, Sage et al. (2006) also recognise possible negative effects and presume that it is highly likely that less widespread species would (also) be displaced by SRC or other energy crops, e.g. on arable land. They cite the rare Montagu's Harrier (*Circus macrourus*) and Corncrakes (*Crex crex*) as examples of affected bird species which are reliant on open country.

4.6 Impact of accompanying structures

In agro-ecosystems, species richness is often correlated with the spatial heterogeneity of the environment (Freemark, 1995; Hendrickx et al., 2007). Structurally rich blocks of trees and heterogeneously composed SRCs increase the diversity and the density of breeding birds (Berg, 2002; Sage et al., 2006; Gruß & Schulz, 2008). In particular, the diversity of vertebrates and invertebrates in SRCs can be greatly increased by accompanying structures in boundary and internal border areas. Marginal farmland habitats such as hedgerows and grass land margins provide valuable wildlife habitats (Anderson et al., 2004; Kühne & Freier, 2001; Röser, 1989). Such accompanying structures can also be represented in SRCs by sunlit interior borders (along the farm tracks between blocks of poplar) and by peripheral hedges, for instance. Few studies have been conducted in this area. Thus, for example, four butterfly species were found in the interior of an 8-year-old poplar plantation in Germany; however, 14 butterfly species were found in a relatively euphotic internal border with a width of around 15 m between two blocks of poplar (Schulz et al., 2008b; Brauner & Schulz, in prep.). In addition, a hedge approximately 3 m wide (which included field maple, hawthorn and other shrubs) at the edge of the SRC was studied. The largest number of ground beetles was discovered here (Sachs, 2007; Schulz et al., 2008a), as well as high densities of breeding birds, butterflies and grasshoppers (Gruß & Schulz, 2009; Brauner & Schulz, in prep.).

5 Recommendations

Up to now, in this review only the absolute numbers of animal species have been considered, without any attention being paid to the actual species concerned. The quality of the species encouraged or discouraged is crucial from a conservational point of view, however. Overall, small numbers of specialists and much larger numbers of generalists have been found in SRCs (see chapter 2). The detected spectra contained mainly eurytopic species which are common in woods, arable lands and ruderal habitats.

The surveys suggest that hybrid poplar or willow plantations are not likely to provide wildlife habitats of major significance, but are likely to contribute to increased biodiversity in many agricultural areas (Britt et al., 2007). Very few Red List species were discovered, however (Anderson et al., 2004; Berg, 2002; Gruß & Schulz, 2008 and 2009; Jedicke, 1995; Sage et al., 2006). Bird species of conservational value were most frequently found in recently planted or harvested SRCs, as some threatened open land breeders are able to find replacement habitats here (see chapter 3).

Like afforestation SRC offer chances and risks to nature conservation and landscape management (Klein, 1997). Overall, the conservational value of SRCs can generally be improved by encouraging habitat diversity. This includes the spontaneous development or establishment of hedges, external and internal boundaries, outer zones and borders – or even gaps within the SRC (Jedicke, 1995; Schulz et al., 2008c; Schmidt & Glaser, 2009).

Sage et al. (2006) state that if biomass crops become more widely grown, it is likely that economies of scale will encourage the planting of large, uniformly managed blocks, harvested at the same time. Under such a scenario, there could be a potential benefit in developing agri-environment options to encourage smaller plantings, the splitting of blocks by rides and hedges, and rotational harvesting in mixed age-class blocks (Sage et al., 2006). Jedicke (1995) proposes that, when planning SRCs, 10 % of the area should be reserved for smaller habitats, such as islands of field wood, stepped wood boundaries and strips of grass. Rowe et al. (2007) summarise that SRCs should be designed with a large edge to interior ratio, a mix of varieties and clones should be used, and willow clones with a range of flowering times. Furthermore, the use of pesticides should be limited (Rowe et al., 2007).

A crucial point, however, is the location of the SRC. Berg (2002) suggested that plantations should be avoided close to habitats of high conservation values, such as wet meadows. Conservationally valuable areas of Germany where SRCs should not be planted also include xerothermal grassland, marsh areas, ground nesting areas and river meadows (Jedicke, 1995; Völkl, 1997). For example the conversion of grassland into short-rotation poplar plantations causes a significant change in the diversity of species, but the cultivation of maize would be more unfavourable (Rösch et al., 2009). Set-asides, where birds and insects that favour fields and meadows can replenish their population, and the negative curtain effect associated with the occurrence of meadow breeders also need to be considered (Jedicke, 1995; Lee & Elsam, 2008). SRC on infields in forest-dominated landscapes will have negative effects, since the mosaic structure (i.e. mixture of open and forested habitats) will disappear, and relatively few forest species are favoured by *Salix* plantations (Berg, 2002).

Some open farmland birds, like sky lark, have been recorded in recently harvested SRC (Sage et al., 2006; Gruß & Schulz, 2009), suggesting that including a range of harvesting cycles in large plantations could reduce negative effects (Sage et al., 2006; Rowe et al,. 2007). Rowe et al. (2007) state nevertheless for English willow SRC, that the rapid growth rate may limit the effectiveness of this method, and a few bird species such as stone curlew (*Burhinus oedicnemus*) are likely to be negatively affected by establishment of SRC regardless of harvest cycle.

For the future, it is crucial that priority areas and restricted areas for SRCs are established on the basis of Gruttke (1997) and Jedicke (1995). Schmidt & Glaser (2009) make valuable suggestions. They have developed a decision algorithm that can be used to select and/or exclude areas that may be suitable for SRC on the basis of various nature conservation criteria.

6 Future research questions

In the future, the various development phases must be defined more precisely when studying biodiversity in SRCs, and their effects compared in greater detail. Christian et al. (1998) have already pointed out that, as plantations of a variety of sizes, ages, intensities of management, and landscape contexts become available, it will be imperative that biodiversity studies continue. This still applies, particularly with regard to the conditions in Central Europe. Regional distinctions and, in particular, the influence of the surrounding landscapes on the diversity of SRCs and the influence of SRCs on the diversity of the surrounding landscapes need to be considered. And the impact of hybridization of poplar and willow clones with autochthonous trees has to be scrutinized (Rotach, 2004; Schmitz et al., 2008).

Not all conclusions drawn with regard to biodiversity in SRCs in other countries are applicable to Germany. The extensive bird studies carried out by Berg (2002) were performed largely in Swedish *Salix* plantations, for instance. However, poplar plantations, which obviously have a lower biodiversity, dominate in Germany (Sage at al., 2006; Gruß & Schulz, 2008). No research whatsoever has been carried out in areas planted with robinia.

Furthermore, the distribution areas, ecological plasticities and population densities of animal species are of course different in Northern Europe than in Central Europe. This is apparent from the relatively extensive bird studies carried out in the past. Game birds, for instance (mainly pheasant and partridge), of which there is a relatively high abundance in British and Swedish SRCs (Berg, 2002; Sage & Robertson, 1996), have not been discovered in any Germany SRC up to now (Jedicke, 1995; Liesebach & Mulsow, 1995; Liesebach & Mulsow, 2003; Gruß & Schulz, 2008; Gruß & Schulz, 2009).

Furthermore, most studies into animal diversity carried out in the past have taken place in pre-commercial short-rotation coppices. But it makes a difference whether small sample areas or large plantations used for commercial purposes are studied. There were often gaps in the overall crop in pre-commercial plantations, due to poplar or willow hybrids of various origins being tested, and due to trees which had died. Such gaps in the canopy allow other plant species to grow and other animal species to settle. Therefore, these gap-habitats can considerably enrich the biodiversity and ultimately reflect structural complexity and abiotic conditions that are not to be expected in future commercial SRCs. Particularly in Germany, commercial SRCs which have been planted recently are considerably larger and more homogeneous than the sample areas.

In the future, large commercial SRC might generate high pest levels. Agroforestry has been assumed to reduce pest outbreaks usually associated with monocultures (Altieri & Nicholls, 2004). It should be investigated how such benefits from biological interactions could be achieved in SRC.

Furthermore, more indicator groups than previously should also be selected. Up to now, only the bird group has been studied in depth. But evidence from well-studied animal and plant taxa in Britain shows that areas rich in species for one group, such as birds, may be depauperate for a different group, such as the butterflies (Trevelyan & Pagel, 1995). Consequently, several indicator groups that complement one another in terms of their ecological properties and habitat requirements need to be studied in the future. These should include members of species-rich invertebrate groups and species groups with little mobility. More research is required to investigate the effect of SRC plantations on either euedaphic or hemiedaphic species of soil invertebrates (Rowe et al., 2007).

Comparisons with traditional coppices could be useful for deriving suggestions for enriching biodiversity in SRCs. Traditional coppices are centers of especially high biodiversity (LANUV, 2007) and it may be the case that the biodiversity in SRCs can be increased by adopting structures and characteristics typical of traditional coppices. For English plots, Sage & Robertson (1996) report that the songbird species using the SRC survey plots were similar to those reported from traditional coppice habitats. There is entirely contradictory information with regard to German plots, however (Jedicke, 1995; LANUV, 2007; Blankenstein, 2007). This is another indication of the different characteristics and environmental effects of woody biomass crops within the various countries of Europe.

In the future, in terms of conservation values, it seems to be worth to put more emphasis on qualitative aspects of SRCs' coenotic composition, its surroundings and the areas where SRCs are supposed to be established, respectively. So far most studies merely consider species-richness as a parameter of a conservation value. But, due to the prospective habitat loss especially for demanding or even endangered species of open lands by establishing SRCs the evaluation of this land use should focus more intensively on such distinguishing faunistic elements and accordingly the habitat potential of SRCs for those species. Target species for future monitoring programmes should then be derived from biocoenotic studies and subsequent nature conservational assessments (Schmidt & Glaser, 2009).

Furthermore, future in-depth research is recommended which deals with the similarities and differences in species composition and abundances that exist between SRCs and forest ecosystems.

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Effects of short rotation coppice with willows and poplar on soil ecology

Christel Baum*, Peter Leinweber*, Martin Weih**, Norbert Lamersdorf*** and Ioannis Dimitriou**

Abstract

Fast growing willow and poplar clones (*Salix* and *Populus* spp.) grown as short rotation coppice (SRC) on former arable soils lead to a long-term non-tillage management with increased litter amounts. Additionally, the litter composition is changed (e.g. increased C/N ratios and lignin contents) and thereby the litter decomposition can be retarded. An increased C sequestration in the topsoil of former ploughed arable soils can result from these effects. The leaf litter recycles a high portion of nutrients. In contrast to other crops, willows and poplar can be colonised by ectomycorrhizal fungi. This leads to the introduction of ectomycorrhizal fungi into arable soils and to changes in the soil microbial colonization and activity. The non-tillage management and the high litter supply can change the abundance and diversity of the soil fauna, e.g. increase the abundance of earthworms (Lumbricidae) and in spite of an increased diversity decrease the abundance of carabids (Carabidae). Willow and poplar clones are highly suitable for phytoremediation of contaminated soils (e.g. extraction of Cd, Zn and degradation of organic pollution) caused by their high biomass production in combination with high fine root density. Several soil ecological advantages of short rotation coppice compared to former arable soils with annual crops can be stated, however, more research-based knowledge is needed especially on the fundamentals of long-term effects and on the sustainability of effects after return to their former commercial arable use.

Keywords: Soil, carbon sequestration, soil organisms, mycorrhiza, phytoremediation, soil organic matter

Zusammenfassung

Einfluss von Kurzumtrieb mit Weiden und Pappeln auf die Bodenökologie

Kurzumtriebsplantagen (KUP) mit schnellwachsenden Weiden- und Pappelklonen (*Salix* und *Populus* spp.) führen auf vormals kommerziell genutzten Ackerböden zur Einstellung der Bodenbearbeitung in Verbindung mit erhöhten Streumengen. Zusätzlich ist die Qualität der Streu verändert (z. B. das C/N-Verhältnis und der Ligningehalt erhöht) und hierdurch kann der Streuabbau verzögert sein. Diese Faktoren können im Oberboden zu erhöhter Kohlenstoffspeicherung führen. Der Verbleib der Blattstreu im Bestand führt zu einer hohen Nährstoffrückführung. Im Gegensatz zu kommerziellen landwirtschaftlichen Nutzpflanzen können Weiden und Pappeln Ektomykorrhizierung ausbilden. Das führt zu einer Einwanderung von Ektomykorrhizapilzen in landwirtschaftliche Böden und zu Veränderungen in der bodenmikrobiellen Aktivität und Besiedlung. Der Verzicht auf Bodenbearbeitung und die hohen Streumengen können weiterhin zu Veränderungen in der Abundanz und Diversität der Bodenfauna führen, z. B. zu erhöhten Abundanzen von Regenwürmern (Lumbricidae) und trotz erhöhter Diversität zu reduzierten Abundanzen von Laufkäfern (Carabidae). Aufgrund ihrer hohen Biomasseproduktion und hohen Feinwurzeldichten eignen sich Weiden- und Pappelklone zur Phytoremediation von kontaminierten Böden (z. B. über die Aufnahme von Cd und Zn und den Abbau von Xenobiotica). Insgesamt können KUP auf vormals mit annuellen Kulturen bestellten landwirtschaftlichen Flächen damit verschiedene bodenökologische Verbesserungen bewirken. Informationsdefizite bestehen insbesondere weiterhin zu Langzeitwirkungen und zur Nachhaltigkeit dieser Veränderungen nach Rückkehr zur vorherigen Ackernutzung.

Schlüsselworte: Boden, Kohlenstoffsequestrierung, Bodenorganismen, Mykorrhiza, Phytoremediation, organische Bodensubstanz

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1 Introduction

Short rotation coppice (SRC) is defined here as a production system in which fast growing recoppicing species of trees and bushes are intensively managed to yield large quantitiy of woody biomass in rotations of about 3 to 10 years (Šlapokas and Granhall, 1991a). Fast growing species of willows (*Salix* spp.) and poplar (*Populus* spp.) are used in SRC on former arable sites as a renewable source of energy and for the phytoremediation of contaminated soils and water (Volk et al., 2004; Mirck et al., 2005). These tree taxa can be highly effective in wood biomass production in SRC as a constituent of the agricultural production in temperate conditions (Anderson et al., 1983; Stolarski et al., 2008).

Within the last thirty years soil ecological effects of SRC on former arable soils were investigated, e.g., to disclose effects on the C sequestration with possible consequences for the soil fertility or climate protection (Makeschin et al., 1989; Hansen, 1993; Grigal and Berguson, 1998; Post and Kwon, 2000; Hoosbeek et al., 2004; Weih and Bussel, 2006; Sanchez et al., 2007; Arevalo et al., 2009) and effects on the diversity, abundance and activity of soil organisms (Makeschin, 1991). Soil ecological effects of SRC on former arable soils were already reviewed by Makeschin (1994) and Mann and Tolbert (2000). Environmental effects of SRC were summarised by Ranney and Mann (1994), Joslin and Schoenholtz (1997), Jug et al. (1999), especially for Austria by Trinkaus (1998) and especially for Sweden by Perttu (1998) and Börjesson (1999). In the review of Makeschin (1994) lack of information was emphasized especially on long-term effects of SRC. The majority of more recent investigations continued to focus on short-term effects of SRC on soil properties a few long-term studies were published (e.g. Kahle et al., 2007). Besides the long-term effects, any indication of sustainability concerning changes in soil properties would be especially valuable for the ecological assessment of SRC on arable soils. Mann and Tolbert (2000) concluded that soil ecological benefits of SRC can be caused by the following mechanisms: i) the continuous plant cover intercepts rainfall and decreases erosion potential, (ii) the increased root development at greater depths stabilizes soil, improves nutrient uptake and reduces leaching losses, and increases organic matter input, (iii) litter and vegetation intercept surface runoff and enhance infiltration, and (iv) the cooler soil temperatures decrease the rate of decomposition. According to these authors, soil ecological benefits of SRC with fast growing tree species even on eroded sites were predicted to become detectable already in 3 to 5 years. Furthermore, beneficial applications of SRC for soil erosion control (Wilkinson, 1999) and phytoremediation of contaminated soils were reported (Keller et al., 2003). Selection of optimal conditions for beneficial environmental effects of SRC were summarised by Lamersdorf et al. (2008).

The present review describes the state of the art in the following aspects of soil ecology under SRC with willows and poplar on former arable sites: i) effects on C sequestration, ii) nutrient cycling from litter, iii) effects on soil organisms, and iv) applications for phytoremediation of contaminated soils.

2 Carbon sequestration in the soil

Potential options of C sequestration in arable soils via agroforestry, mainly focused on tropical tree species and management systems, were recently reviewed by Nair et al. (2009). It was estimated, that the area under agroforestry is currently worldwide about 1,023 million ha and that the C sequestration in this use depends on a number of site-specific biological, climatic, soil and management factors (Nair et al., 2009). The total C sequestration under SRC is significantly higher than under arable soils with annual crops, but still below the C sequestration in mature forests (Table 1, Boman and Turnbull, 1997). The C turnover under SRC on former arable soils is more similar to that in forests than in arable soils (Svensson et al., 1994). However, it is likely that C sequestration varies significantly between the tree genotypes even within one genus (Weih and van Bussel, 2006). The C accumulation after conversion of commercial arable sites to SRC was concentrated in the topsoil (Makeschin, 1994; Stetter and Makeschin, 1997; Neergaard et al., 2002; Dowell et al., 2009). Estimated rates of C accumulation in topsoil (0–40 cm) of arable sites were 40–170 g C m^{-2} yr^{-1} during the first decade following plantation establishment (Garten, 2002).

Table 1:

Average C sequestration in arable use, short rotation coppices and forests in t ha^{-1} (Boman and Turnbull, 1997)

Components	Arable use	Short rotation coppice	Forest
Leaves	4.0	2.5	2.5
Trunks	0	21.0	70.0
Weed	0.5	1.0	2.0
Litter	0.5	5.0	15.0
Roots	2.0	5.5	10.0
Soil	25.0	35.0	45.0
Total	**32.0**	**69.5**	**144.5**

The increased C concentrations in soils under SRC can be explained by: i) the non-tillage management and high annual amounts of leaf litter of in average 1 to 5 t ha^{-1} (Verwijst and Makeschin, 1996; Boman and Turnbull, 1997) and ii) the increased transfer of assimilates into external mycelium of mycorrrhizal fungi (Ek, 1997). The external mycelium of mycorrhizal fungi was the dominant pathway (62 %) through which C entered the soil organic matter (SOM) pool, and this input exceeds the litter and fine root turnover under poplar in SRC (Godbold et al., 2006). However, the annual leaf litter fall is the main source of easily-available C sources for the soil microorganisms as derived from hot water extracts (Huang and Schoenau, 1996). Gordan and Matthews (2006) predicted that the potential for C sequestration under SRC with willows is largest in soils whose C content has been depleted to relatively low levels due to aeration by annual deep ploughing in arable soil use. In summary, the publications agree in the general possibility of increased C sequestration by SRC on former arable soils, however, the initial soil properties govern the extent to which C is stored. Therefore, approaches for the selection of most promising sites for C sequestration by SRC must be developed.

Generally, soil C can be sequestered by various processes: (i) stabilization by interaction with mineral surfaces (Fe-, Al-, Mn-oxides, phyllosilicates) and metal ions, (ii) spatial inaccessibility of SOM against decomposer organisms due to occlusion, intercalation, hydrophobicity and encapsulation; and (iii) selective preservation due to recalcitrance of molecules from plant litter, rhizodeposits, microbial products, humic polymers, and charred SOM (Lützow et al., 2006; Marschner et al., 2008). Various of these processes such spatial inaccessiblility and organo-mineral interactions, e.g., in faeces of earthworms or recalcitrance of specific molecules in the litterfall or rhizosphere of SRC may be relevant for C sequestration under SRC but this has not been studied at the molecular level. Various chemical-analytical methods are applied in routine and research to describe the molecular composition of SOM such as compound specific wet-chemical extractions, nuclear magnetic resonance (NMR) spectroscopy (Kögel-Knabner et al., 2008) and analytical pyrolysis and mass spectrometry (Leinweber et al., 2009). However, none of these sophisticated methods has been applied to look at alterations in the molecular composition of SOM under SRC. Therefore, it is almost completely unknown which chemical alterations the SOM undergoes, if SRC are established and stand for long time at previous arable soils. Such a basic understanding, however, is urgently needed if predictions for the long-lasting C-sequestration under SRC shall be based on sound scientific evidence.

3 Nutrient cycling from litter

SRC on former arable soils affects the soil nutrient turnover i) by its biomass and rhizodeposits and ii) by its management. The litter biomass quality and quantity is influenced by soil properties (Rytter, 2001) and species or genotypes (Ericsson, 1981; Weih and Nordh, 2002; Lukac at al., 2003; Cotrufo et al., 2005). For example the N concentration of abscised willow leaves, differed among genotypes of the same tree species (Weih and Nordh, 2002), affect litter quality. The litter decomposition rate differed tree species-specific (Püttsepp et al., 2007) and effects of the tree species were superimposed upon other controls of the litter decomposition under SRC (Šlapokas and Granhall, 1991b). Meiresonne et al. (2006) investigated the hydrological fluxes, atmospheric deposition, litterfall, and soil percolation of the most important nutrients in an 18-year-old poplar plantation on a well-drained silt loam soil during 2 years. In this study around 80 % of total nitrogen input (6.6 kmol·ha^{-1} in years 1 and 6.5 kmol·ha^{-1} in year 2) originated from litterfall and after nitrification only a negligible amount of nitrate leached during the growing season. The yearly uptake of N by the poplar ecosystem in this study was equal to the input, of which more than 50 % was accounted for by the leaves. This indicated very efficient N cycling. Total deposition of base cations originated from two processes, dry deposition (Mg^{2+} and Ca^{2+}) and canopy leaching (K^+ and Ca^{2+}). Litter input of Ca^{2+} represented about 83 % of the total input (stand deposition + litterfall), Mg^{2+} about 61 %, and K^+ less then 50 %. Percolation of base cations at 1 m depth was very limited. Rather high Ca^{2+} and K^+ contents of the woody biomass can lead to high exports at harvest. Meiresonne et al. (2006) concluded, that the nutrient cycling in the poplar stand was very efficient, with no significant nutrient losses.

The ratio of aboveground biomass to fine root biomass production of lysimeter-grown willow varieties ranged from 0.4 to 1.2 (Rytter, 2001). Also the root-to-shoot ratio of willows varied between genotypes (Weih and Nordh, 2005), which is likely to influence fine root biomass and turnover. The fine root characteristics of poplar varied clone-specific (Al Alfas et al., 2008) and were affected by the management, like irrigation and coppicing (Dickmann et al., 1996). The average daily fine root growth (m m^{-2} day^{-1}) of aspen (*Populus tremula*) was positively correlated with soil temperature at 10 cm depth (r^2 = 0.83–0.93) (Steele et al., 1997). The fine root biomass under clones of the willows *S. viminalis* and *S. dasyclados* in a SRC on former arable soil in Estonia was vertically concentrated (39 to 54 % of the total fine root biomass) in the uppermost 10 cm of soil (Heinsoo et al., 2009). Thus it is not surprising that the fine-root turnover was mentioned as a substantial constituent of the nutrient cycling under willows (Rytter, 1999; 2001). Median fine root life span of poplar (*Populus deltoides)* varied from 307 to over 700 days and increased with depth, diameter and nutrient availability (Kern et al., 2004).

In addition to plant-genotype effects, SRC can cause several changes in soil chemical properties (Kahle et al., 2005), which affect the soil nutrient turnover. Again, this depends on the initial soil properties at the sites. The C/N ratio in the topsoil under SRC on former arable soils slightly increased (Stetter and Makeschin, 1997), and the soil pH decreased in the upper 0 to 10 cm of soil by about 0.5 to 0.8 units while the cation exchange capacities decreased by about 15 % (Jug et al., 1999). During the planting and establishment of SRC on former arable soil initial high nutrient losses are possible (Granhall and Šlapokas, 1984; Makeschin, 1994; Jug et al., 1999) because tillage promotes the mineralization and weed control reduces the organic matter input. However, in established SRC sites low nitrate losses were measured even in combination with an annual N fertilization of 150 kg N ha^{-1} and explained by the fast plant growth (Bergström and Johansson, 1992; Mortensen et al., 1998). The average annual nutrient uptake and removal by wood biomass were 18 to 54 kg N ha^{-1}, 10 to 70 kg Ca ha^{-1}, 3 to 9 kg P ha^{-1}, 6 to 36 kg K ha^{-1} and 1 to 5 kg Mg ha^{-1} in rotation periods of five years (Jug et al., 1999). The annual nutrient uptake of two poplar clones in France reached 92 kg N ha^{-1}, 15 kg P ha^{-1} and 87 kg K ha^{-1}. The total uptake of nutrients varied significantly in dependence on the soil texture (Rytter, 2001). About 60 to 80 % of the nutrients taken up returned to the soil through litterfall which reached about 4 to 5 t ha^{-1} a^{-1} at an age of 7 to 8 years (Berthelot et al., 2000). At degraded arable sites the nutrient supply and growth of poplar was significantly promoted by application of grass mulch (Fang et al., 2008).

4 Effects of SRC on soil organisms

4.1 Soil microorganisms

Soil microbial communities are important regulators of processes such as nutrient cycling and decomposition, and can offer protection against pests and diseases. These microorganisms rely predominantly on organic matter provided by root exudates, secretions and other rhizodeposits, including root turnover. Therefore microorganism communities are greatly influenced by plant species and genotype. For example, the diversity of saprotrophic microfungi in the rhizosphere depended on the willow variety grown in SRC plantations (Šlapokas and Granhall, 1991a; Baum and Hrynkiewicz, 2006). The vertical distribution of soil microorganisms was changed under SRC on former arable sites caused by the non-tillage management. This means that the microbial biomass in the soil increased in the upper 5 cm of soil but decreased in subsoils compared to the arable soil under SRC (Makeschin, 1994).

Mycorrhizal fungi are an important component of the soil microbial community, forming symbiotic associations with most land plants and mediating a range of crucial ecosystem processes including nutrient cycling, organic matter decomposition, C sequestration and soil aggregation (Zhu and Miller, 2003; Olsson and Johnson, 2005; Smith and Read, 2008; van der Heijden et al., 2008). For example, mycorrhizal fungi frequently are essential for plant nutrition, most notably in the provision of phosphorus and nitrogen to the host plant (Smith and Read, 2008). Furthermore, they form various symbiotic interactions with other soil organisms affecting soil ecology and biodiversity (e.g. mycorrhiza helper bacteria and plant growth-promoting rhizobacteria) (Zimmer et al., 2009). Two major types of mycorrhizal fungi, the arbuscular mycorrhizal fungi and the ectomycorrhizal fungi form symbiotic associations with most land plants. Arbuscular mycorrhizal fungi are common in arable soils, since the majority of arable crops forms arbuscular mycorrhizae, whereas in SRC plantations on former arable sites ectomycorrhizal fungi are usually introduced after long-term absence of host plants of ectomycorrhizal fungi. Although mycorrhizal colonization often has been documented in poplars and willows grown on former arable land (Baum et al., 2002; Khasa et al., 2002; Püttsepp et al., 2004; Trowbridge and Jumpponen, 2004), little is known about their functional role in biomass plantations. Interestingly, poplar and willows can form associations with both arbuscular and ectomycorrhizal fungi, known as 'dual colonization' (Lodge, 1989). However, it is usually one type of mycorrhiza that dominates or exclusively colonises a given tree at a given time, and ectomycorrhizal colonization normally seems to exceed arbuscular mycorrhizal colonization in poplars and willows (Trowbridge and Jumpponen 2004, Kahle et al., 2005). Ectomycorrhizal fungi will become introduced into arable soils with poplars and willows since the portion of ectomycorrhizal taxa on the total diversity of basidiomycetes in arable soils is rather low and dominated by saprotrophic taxa (Lynch and Thorn, 2006). Examples of ectomycorrhizal fungi of willow and poplar species are listed in Table 2. This overview documents fungal taxa that can be preferentially introduced into arable soil by SRC with poplar and willows. However, such potential changes in the soil microbial diversity with increased spreading and activity of ectomycorrhizal fungi in the soil are scarcely proven although they can have significant ecological effects. For instance, members of basidiomycetes are the main decomposers of recalcitrant components of plant litter because they can produce lignin-modifying enzymes (Rayner and Boddy, 1988). Therefore, changes in their diversity can have significant consequences for the litter decomposition.

Table 2:

Ectomycorrhizal fungi on *Salix* and *Populus* spp.

Host plant	Ectomycorrhizal fungi	Reference
Salicaceae	*Cenococcum geophilum*	LoBuglio, 1999
	Lactarius controversus	Hesler and Smith, 1979
		Moser, 1983, Bills 1986
Salix spp.	*Amanita silvicola*	Gardes and Dahlberg, 1996
	Cortinarius decipiens	Jumpponen et al., 2002
	C. tenebricus	
	Entoloma sinuatum	Agerer, 1997
	Hebeloma crustuliniforme	Aanen et al., 2000
	H. helodes	
	H. lutens	
	Hymenogaster rubyensis	Fogel and States, 2001
	Inocybe fuscomarginata	Beeneken et al., 1996
	I. lacera	Jumpponen et al., 2002
	Lactarius aspideus	Arnolds, 1989; Watling, 1992
	Russula fragilis	Jumpponen et al., 2002
	Tuber magnatum	Granetti, 1987
S. caprea	*Cenococcum geophilum*	Hrynkiewicz and Baum, 2003
	Cortinarius atrocoeruleus	
	Hebeloma helodes	
	Laccaria spp.	
	Lacatrius pubescens	
	Paxillus involutus	
	Phialophora finlandia	
	Tomentella spp.	
	Tricholoma cingulatum	
	Tuber raeodorum	
S. viminalis	*Inocybe glabripes*	Baum et al., 2002
Populus spp.	*Paxillus involutus*	Heslin and Douglas, 1986
	Tuber albidum	Fontana and Palenzona, 1969
P. alba	*Scleroderma bovista*	Jakucs and Agerer, 1999a
	Tomentella pilosa	Jakucs and Agerer, 1999b
	Tomentella subtestacea	Jakucs and Agerer, 2001
P. nigra	*Inocybe fuscomarginata*	Beeneken et al., 1996
P. tremula	*Pisolithus* spp.	Godbout and Fortin, 1985
	Scleroderma areolatum	
	S. citrinum	
	Tricholoma populinum	Kreisel et al., 1990
P. tremula x tremuloides	*Amanita muscaria*	Hampp et al., 1996
	Cenococcum geophilum	
	Cortinarius uliginosus	
	Entoloma minutum	
	E. prunuloides	
	Hebeloma helodes	Kaldorf et al., 2004
	Inocybe geophylla	Baum and Makeschin, 1999
	I. microspora	
	Laccaria laccata	
	Laccaria tortilis	
	Lactarius controversus	
	Leccinium aurantiaca	
	Paxillus involutus	
	Phialocephala fortinii	Kaldorf et al., 2004
	Tomentella spp.	
	Tuber spp.	
P. tremuloides	*Amanita muscaria*	Cripps and Miller, 1993,
	A. pantherina	Cripps and Miller, 1995
	Cenococcum geophilum	
	Inocybe dulcamara	Cripps, 1997
	I. flavella	
	I. flocculosa	
	I. geophylla	
	I. giacomi	
	I. griseolilacina	
	I. lacera	
	I. longispora	
	I. mixtilis	
	I. nitidiuscula	
	I. phaeocomis	
	I. sindonia	
	I. squamata	
	I. whitei	
	Laccaria laccata	Cripps and Miller, 1993
	Lactarius controversus	
	Leccinium aurantiacum	
	Paxillus vernalis	Cripps and Miller, 1995
P. trichocarpa	*Amanita muscaria*	Trappe, 1962a
		Molina and Trappe, 1982
	Cortinarius croceocaeruleus	Baum and Makeschin, 1997
	C. uliginosus	
	Inocybe geophylla	
	I. microspora	
	I. umbrina	
	Laccaria laccata	
	Laccaria tortilis	
	Lactarius controversus	
	Leccinium aurantiaca	
	Paxillus involutus	
	Rhizopogon spp.	
	Tuber borchii	

Mycorrhizal colonization of *Populus* and *Salix* spp. varies greatly between species and genotypes (Khasa et al., 2002; Püttsepp et al., 2004), and also depends on soil properties and management effects (Loree et al., 1989; Baum and Makeschin, 2000; Baum et al., 2002). In intensively managed biomass plantations, the degree of mycorrhizal colonization and the diversity of ectomycorrhizal fungi appear to be lower compared to adjacent natural stands (Toljander et al., 2006). Furthermore, mycorrhizal colonization affects the leaf chemistry of willows (Baum et al., 2009), which, in turn, is likely to influence willow leaf resistance to pests such as herbivory insects. The effect of mycorrhiza on leaf chemistry varied between host plant genotypes (Baum et al., 2009). The interaction of plant genotype and herbivory can affect the leaf litter decomposition and alter the nutrient dynamics (Schweitzer et al., 2005). This example shows that willow genotype directly affects soil ecology in terms of mycorrhizal colonization, which could indirectly influence the cropping safety of the plantations through an effect on pest resistance. Understanding of these complex multi-trophic interactions is crucial to our general understanding of soil ecosystem function and the regulation of fundamental ecosystem processes. Furthermore, understanding of multi-trophic interactions could also support yield increases and cropping safety in willow and poplar biomass plantations in a most sustainable way, e.g. by the appropriate choice of willow varieties that favour pest resistance.

4.2 Soil fauna

The soil fauna is an important control of litter decomposition and bioturbation and in consequence of the nutrient cycling and plant growth. Since the soil macrofauna can be damaged by soil tillage a promotion during non-tillage SRC seems most probably. In agreement with this, the abundance of earthworms (Lumbricidae) (Makeschin, 1994), harvestmen (Opilionida) and woodlice (Isopoda) increased on experimental sites with poplars and willows on former arable soils after the conversion to SRC (Makeschin, 1991). The abundance of carabids (Carabidae) and spiders (Araneida) decreased after this conversion. However, under fast growing trees a greater diversity of carabids (Carabidae) was detected. Centipedes (Chilopoda) and millipedes (Diplopoda) did not change after conversion from arable to forest land use. Mineral fertilization had no significant effect on this faunal groups under SRC (Makeschin, 1991). The effects of poplars and willows on the soil fauna differed significantly. The supposed determining factor of the tree taxa-specific differences was a different water regime in soil resulting from a lower interception by the willows compared to poplar and differences in the leaf and root litter production (Makeschin, 1994). The abundance and diversity of soil mites (Oribatida and Gamasida) was negatively affected through soil cultivation for SRC during the first year after conversion arable land to SRC (Minor et al., 2004). In this experiment, the use or lack of tillage contributed to differences in the mite community structure and following initial disturbance. However, the abundance and diversity of soil mites increased in the long term under SRC (Minor et al., 2004). Diverse management effects (application of biosolids, chicken manure compost, urea fertilizer and black plastic mulch) on the soil mites (Acari) under willows in SRC were investigated by Minor and Norton (2004). They found, that urea fertilizer had no persistent effect on mite assemblages in SRC. Plastic mulch did not affect Mesostigmata, but had a lasting detrimental effect on oribatid mites. Mesostigmatid mites benefited from application of biosolids, while Oribatida were most adversely affected by this treatment. When plastic mulch and biosolids were used together, the effect of biosolids predominated. Composted chicken manure supported abundant and diverse populations of both groups (Minor and Norton, 2004).

5 Applications for phytoremediation of contaminated soils

Phytoremediation is defined as the use of trees and other plants such as grasses and aquatic plants, to remove, destroy or sequester hazardous substances from the environment (Glass, 1999). This chapter reviews the use of poplar and willows for cleaning/improving the soil. Poplars and willows can be used for several different types of phytoremediation for soil improvement, based on the function of the plants against hazardous compounds. These are "phytoextraction" (ability to accumulate large quantities in the above ground parts removed by harvest), "rhizofiltration" (absorption onto plant roots removed from aqueous waste-streams), "phytotransformation" (degradation or metabolization in the plant parts) or "phytovolatilization" (volatilization into the air from plant biomass), "phytostimulation" (degradation of pollutants in soil due to secreted plant enzymes or by plant stimulation of microbial biodegradative activity), "phytostabilization" (immobilization in the soil supported by plant exudates), and "phytomining" (extraction of large amounts of metals by plants) (Glass, 1999). Beside the direct effects of the plants during phytoremediation of contaminated sites, indirect effects like metal immobilization by increased SOM sequestration (see section 2) with increased concentrations of metal-chelating substances in the soil can be used. This can lead to a decreased bioavailability and thereby phytotoxicity of heavy metals and to a decrease the risk of metal leaching into the ground water.

Willows and poplars are no hyperaccumulators of metals or other hazardous compounds but they were preferred in commercial phytoremediation projects due to their fast and high growth, and the fact that agronomic practices for SRC easy management and good growth performance already exist. Besides the relative high growth, willows and poplars have been reported to evapotranspire high amounts of water (Persson and Lindroth, 1994; Bungart and Hüttl, 2004) and to tolerate high heavy metal concentrations in soil (Hammer et al., 2003; Laureysens et al., 2004). Furthermore, willows are tolerant to anoxic conditions (Jackson and Attwood, 1996). All the above traits enable growth under "difficult" environments, but under the scope of this paper poplar and willow SRC are primarily seen as a biomass production system. For this, productive soils should be preferred to achieve high growth, preferably in large-scale plantations. In many cases however, only moderately contaminated soils are available for

SRC cultivation and other contaminants can already exist in agricultural soils. Therefore, we will mostly focus on implications for large-scale willow and poplar SRC, although results obtained in the laboratory will be used to estimate soil ecological effects.

5.1 Phytoextraction of heavy metals

Extensive research related to phytoextraction of heavy metals, e.g. the ability to accumulate large quantities in the above ground parts removed by harvest has been conducted with willow and poplar. Willows have been reported from early stages of their commercial bioenergy use to take-up large amounts of Cd (Perttu, 1992; Riddell-Black, 1994). Initially most research was done on Cd uptake by willows but later the uptake of other metals together with Cd such as Cu, Pb, Zn, Cr, Ni, As was studied as well (Granel et al, 2002; Landberg and Greger, 2002; Ali et al., 2003; Vyslouzilova et al., 2003; Kuzovkina et al, 2004; Fischerova et al, 2006; Dos Santos Utmazian et al., 2007; Meers et al, 2007; Wieshammer et al, 2007). Metal uptake by poplars was studied at later stages since poplar gained constantly interest as alternative species to willow for biomass production for energy (Robinson et al, 2000; Sebastiani et al, 2004; Laureysens et al, 2004; Licht and Isebrands, 2005). The phytoremediation potential of willows and poplars, despite not being hyperaccumulators (Table 3), has been reported to be high based on the combination of high accumulation of metals in the plant tissues together with the high biomass produced (Aronsson and Perttu, 2001; Berndes et al., 2004; Rockwood et al., 2004; Licht and Isebrands, 2005; Puschenreiter and Wenzel, 2007).

Substantial related research was conducted in controlled laboratory conditions where individual willow and poplar plants were grown in contaminated soils (Landberg and Greger, 2002; Rosselli et al, 2003; Vyslouzilova et al., 2003; Sebastiani et al, 2004; Vandecasteele et al, 2005; Fischerova et al, 2006; Meers et al, 2007; Wieshammer et al, 2007) or in hydroponic systems (Kuzovkina et al, 2004; Dos Santos Utmazian and Wenzel, 2007; Dos Santos Utmazian et al., 2007). Very promising results for uptake of certain metals in willow and poplar plant parts were reported from those experiments and speculations for great potential for cleaning contaminated soils with willow and poplar were expressed. Although results from pot-trials have been validated in some cases in the field (Robinson et al., 2000; Sebastiani et al, 2004), concerns due to the different conditions between controlled small-scale experiments (often artificially mixed heavily contaminated soils and favorable plant growth) and large-scale field situation (often non-uniform and moderate contamination and lower plant growth) have been raised (Dickinson and Pulford, 2005; Dickinson et al., 2009). We also believe that such extrapolations from lab to field should be drawn cautiously, and although we will refer to results obtained in the laboratory, generalizations for implications in the field will be avoided.

Table 3:

Concentrations of heavy metals in the biomass of *Salix* and *Populus* clones at contaminated soils

Plant species	Plant constituent	Element	Concentration (mg kg^{-1})	Reference
Salix acmophylla	leaves	Cu	2.4 - 126.3	Ali et al., 2003
		Ni	2.9 - 139.1	
		Pb	1.9 - 180.4	
	stems	Cu	4.0 - 203.7	
		Ni	3.5 - 264.3	
		Pb	2.5 - 284.0	
	roots	Cu	6.8 - 624.4	
		Ni	4.3 - 746.3	
		Pb	3.1 - 1038.5	
Salix caprea	leaves	Cd	177.0	Wieshammer et al., 2003
		Pb	79.0	
		Zn	2034.0	
Salix fragilis	leaves	Cd	326.0	Wieshammer et al., 2003
		Pb	68.0	
		Zn	2413.0	
Salix matsudana x S. alba	stems	Cd	9.0 - 167.0	Robinson et al., 2000
Salix viminalis	stems	Cd	3.3 - 3.4	Keller et al., 2003
		Cu	12.0 - 14.0	
		Zn	240.0 – 294.0	
Populus deltoides x P. yunnanensis	leaves stems	Cd	12.0 - 62.0 6.0 - 75.0	Robinson et al., 2000

Many studies have also proposed the use of a range of chelating agents such as ethylendiamin-tetraacetat (EDTA), ethylenediamine-N,N'-disuccinic acid (EDDS), oxalic and citric acids, and others, to increase the positive metal uptake rates by willow and poplar plants (Hooda et al., 1997; Robinson et al., 2000; Schmidt, 2003; Hammer and Keller, 2002; Komarek et al., 2008). Despite the positive results for induced phytoextraction indicated in the previous publications, chelating agents have been reported to cause toxicity symptoms to the plants, leaching of metals and negative impact on soil biota have been reported, therefore questioning the potential future use of chelate-assisted phytoextraction (Evangelou et al., 2007; Dickinson et al., 2009). Another opportunity for the improvement of phytoextraction by willows and is the inoculation with mycorrhizal fungi and bacteria (Sell et al., 2005; Baum et al., 2006; Kuffner et al., 2008; Zimmer et al., 2009). The potential of this biologically based improvements of phytoextraction efficiency is by far not fully explored.

However, some critical points must be considered for successful phytoextraction with SRC. Great variations in metal uptake ability of willows and poplar have been reported in different SRC fields. This might depend on different contamination levels within the fields, and/or differences in the clone material used. Vandecasteele et al. (2002) suggested that Cd uptake in aboveground plant parts tends to increase with increasing Cd in soil. This was also reported in other studies with elevated metal concentrations where willows and poplars took up larger amounts of heavy metals in aboveground tissues (Hammer et al., 2003; Rosselli et al., 2003; Unterbrunner et al., 2007) than in less contaminated soils (Pulford et al., 2002; Klang-Westin and Eriksson, 2003; Dimitriou et al., 2006). Moreover, even spatial variability of contamination within one field might be responsible for great variations in metal uptake (Dickinson and Pulford, 2005). Differences in metal uptake by willow species and clones have been reported by Granel et al. (2002), Landberg and Greger (2002), Vysouzilova et al. (2003), Kuzovkina et al. (2004) and Meers et al. (2007). In analogy, Laureysens et al. (2004) reported great differences in the ability of poplar clones to take up metals. Therefore, to effectively use SRC to clean soils, much attention should be paid to the selection of the clone in relation to the contamination source and level at the site. However, Dickinson et al. (2009) suggested that predictable uptake patterns for all metals will be unlikely to be found for accumulation in aboveground biomass, and only genotypes that take up more mobile elements such as Cd and Zn can be selected for a specific site (Table 3). This was based on reported differences in the metal uptake among different families, species, clones, but also within individual plants. The mobility and plant availability of metals in soil might be also responsible for the great differences in uptake patterns. For example, Eriksson and Ledin (1999) indicated that plant available Cd concentrations in soil were reduced in a willow SRC field, but higher uptake of different metals in willow shoots were not found when plant available fractions differed due to pH changes in a field willow experiment (Dimitriou et al., 2006). In all, it seems that for cleaning soils a "site-specific" approach with pre-testing of several clones to identify the best performing ones for further use in large-scale should be performed in advance, although difficulties due to the heterogeneity of localization of the pollution are to be expected (Keller et al., 2003).

The above raises the question which soils can be satisfactory remediated by phytoextraction of heavy metals with willow and poplar SRC, and what strategies should be followed for best remediation combined with best economic value in a certain time frame. For instance, despite willow and poplar have been proved of equally good or better phytoextraction efficiency than other species (Rosselli et al., 2003; Fischerova et al., 2006), recent studies suggested that short-term remediation is not to be expected in heavily contaminated soils such as mine spoils or heavily contaminated industrial sites due to unrealistically long time scales needed (Dickinson and Pulford, 2005). Furthermore, such sites might be polluted in deep layers which cannot be cleaned with poplars and willows that are appropriate for rather shallow contamination (Keller et al., 2003) since most of their active roots are concentrated near the soil surface (Rytter and Hansson, 1996). However, large-scale SRC cultivation offers a great potential for cleaning moderately contaminated soil from metals as Cd, Zn, Cu, Ni, Se (Dickinson et al., 2009). Such moderately contaminated soils can be arable soils with elevated Cd concentrations from P fertilizer but also with undesired metal enrichments from continuous sludge applications. Berndes et al. (2004) calculated that 100 times more Cd would be removed by willow SRC than harvested by straw in Sweden if SRC will be grown in arable land with elevated Cd concentrations from phosphate fertilizer. These amounts would compensate for the atmospheric deposition each year and would drastically reduce the amount of Cd in arable soils in Sweden, but would give economic incentives for the farmer from compensations for reducing Cd in the soil (ca. 10 % of total revenue). Similar calculations were made by Lewandowski et al. (2006), suggesting that phytoextraction with willows cultivation for a certain period can allow the future use of moderately contaminated fields for more profitable food production, thus increasing farmers´ income.

Sewage sludge is applied to SRC in certain counties such as Sweden, Denmark and U.K. (Nielsen 1994; Aronsson and Perttu, 2001). Sludge contains P and N that are used as fertilizer to SRC, but also contains heavy metals that can accumulate in the soil when applied for many years. Therefore, an increase in biomass of SRC combined with increased metal uptake would result in a balance between metal input with sludge application and metal output with SRC harvest. Based on field results Dimitriou (2005) calculated that the amounts of metals applied with sludge and after the uptake in SRC stems was within legal limits for such practices. Furthermore, if Cd in soil would continue to reduce as in the first years of the experiment, a 26 % reduction of the total Cd in the upper soil layer was to be expected in 25 years. Significant respective reductions for Cu and Zn were also calculated. Similar results were reported by Lazdina et al. (2007) who also found increased metal concentrations in willow shoots compared to control by 4 to 8 % after sewage sludge applications. This indicated the potential for SRC fields to receive sewage sludge in consecutive years without drastically affect soil quality. To test the effect of long-term sewage sludge applications, several willow clones were grown in historically sewage sludge-loaded fields (Pulford et al., 2002; Maxted et al., 2007). Results underlined the potential for using willow to reduce metal amounts, but indicated great differences between clones in uptake of different metals at the same site.

Moreover, different patterns of metal concentrations were mentioned such as in either bark or wood or in leaves versus the shoots. Cd and Zn concentrations were generally much higher in the leaves than in shoots (Dimitriou et al., 2006; Maxted et al., 2007). Based on similar results it has been suggested that leaf harvest would significantly reduce the soil concentrations of these elements in SRC fields (Puschenreiter and Wenzel, 2007; Dickinson et al, 2009). Vandecasteele et al. (2005) suggested that Cd and Zn was accumulated in above-ground willow parts compared to the other metals accumulated in the roots. However, others suggest that most of the metals are concentrated in the roots and small amounts are accumulated in aboveground biomass (Landberg and Greger, 1996). Therefore, it has been suggested to remove both leaves and roots of SRC if a maximum soil cleaning effect is projected (Echevarria et al., 2006). However, harvest rotation is impossible if roots are removed so that this is not an option. Thus, species or clones that have highest biomass growth and potential ability to store more metals in the shoots at a certain site should be preferred for commercial SRC fields.

5.2 SRC and rhizodegradation of organic pollution

Besides the positive effects of SRC to reduce heavy metal concentrations in soil, willow and poplar SRC have been reported to remediate soils from various organic compounds (Schnoor et al., 1997; Aitchison et al., 2000; McMillan and Schnoor, 2000; Corseuil and Moreno, 2001; Predieri et al., 2001; Kelley et al., 2001; Ciucani et al., 2004; Ma et al., 2004; Ucisik et al., 2007), such as chlorinated solvents, explosives, petroleum hydrocarbons, cyanides, pesticides, and others (see Table 4). Soils polluted with such compounds are usually characterised as heavily polluted and are therefore not considered for production of agricultural crops. The plant roots degrade the different compounds in the soil and in most cases these are not absorbed in the harvested parts as it is the case with heavy metals.

Table 4:
List of organic compounds which were degraded by *Salix* and/or *Populus* spp.

Organic compound	Reference
Chlorinated solvents	
Trichloroethylene	Gordon et al., 1997; Shang et al., 2001
Dichlorophenol	Icisik et al., 2007; Shi Xiang et al., 2008
Perchloroethene	Larsen et al., 2008
Pentachlorophenol	Mills et al., 2008
Explosives	
TNT	Thompson et al., 1998; Brentner et al., 2008
RDX	Van Aken et al., 2004; Tanaka et al., 2008
HMX	Yoon et al., 2002
Petroleum hydrocarbons	Palmroth et al., 2002; Rentz et al., 2003; Zalesny et al., 2007
Ethanol-blended gasoline	Corseuil and Moreno, 2001
Cyanides	Ebbs et al., 2003; Larsen et al., 2004; Yu et al., 2005
Dieldrin	Skaates et al., 2005
Dioxane	Schnoor et al., 1997; Aitchinson et al., 2000
Pesticides	Burken and Schnoor, 1997; Predieri et al., 2001

Although the focus of this paper is on SRC systems producing biomass in productive soils (and therefore not heavily polluted with organic compounds), poplar and willow show ability to treat some compounds of interest in agriculture, such as pesticides, and their ability to remediate soils from contamination with such compounds should be examined more closely.

6 Concluding remarks

Soil ecological effects of willows and poplars in SRC on former arable soils can be various and are controlled significantly (i) by natural (e.g. initial soil properties, climate) and (ii) anthropogenic factors (e.g. former and recent management, selection of tree genotypes). In response to natural factors, the selection of suitable sites for SRC seems to be the only appropriate option. However, numerous investigations indicated that anthropogenic factors have a great impact on the soil ecological effects of SRC. This means that the selection of clones for SRC and the management might be valuable tools to increase ecological benefits and to decrease potential disadvantages of SRC on former arable soil. It was demonstrated that SRC can change the site-specific communities of soil organisms with positive (e.g. increased abundance of earthworms and introduction of ectomycorrhizal fungi) but also possibly negative (e.g. decreased abundance of carabids and of predominantly arbuscular mycorrhizal host plants) effects. Lack of knowledge was indicated especially on the sustainability of soil ecological effects of SRC with willows and poplar after return to annual arable crops. Since SRC in the temperate climate is at present usually a constituent in the commercial arable land use, the sustainability of positive and negative effects in the site management should be considered in future investigations.

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A review of the impacts of Short Rotation Coppice cultivation on water issues

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Abstract

Cultivation of Short Rotation Coppice (SRC) with poplars (*Populus sp.*) and willows (*Salix sp.*) for production of biomass for heat and/or electricity is energy effective and coincides with several environmental objectives. Since an increase of cultivation of poplar and willow SRC has been projected in Europe, the consequent implications on water issues arises. For instance, water use of SRC can be higher compared to annual agricultural crops or previous set-aside land, but water quality can be improved. The paper examines such implications based on a review of the existing scientific literature. Rates of evapotranspiration (V_{ET}) of SRC are reported to be fairly higher than arable crops, but reported values vary markedly and are related to site-specific factors such as the local precipitation and conditions (e.g. soil type, temperature, groundwater level), the species/sorts and the age of the crop, and their interactions. Despite the predicted local effects on water balances, effects on water balances/hydrology on catchment levels have not been reported or justified. This, combined with the reported positive effects of SRC on groundwater quality in terms of nutrient leaching, imply average positive effects of SRC on water issues, if the identified potentially negative impacts would be considered and minimized. For this, comparisons of V_{ET} between SRC and arable crops, and the relation of V_{ET} with local precipitation and other local factors (root development, groundwater availability) should be better examined and combined with positive effects of SRC on groundwater leaching compared to other crops. Upscaling of water issues for SRC is needed to promote future decision-making processes with respect to the envisaged broadening of SRC on productive but also on marginal soils.

Keywords: Short Rotation Coppice, poplar, willow, water quality, nutrient leaching, water balance, biomass, bio to energy

Zusammenfassung

Zum Einfluss von Kurzumtriebsplantagen auf den Wasserhaushalt – eine Übersicht

Kurzumtriebsplantagen (KUP) mit Pappel und Weide sind eine energieeffiziente und umweltfreundliche Möglichkeit zur Erzeugung von Biomasse für die Energiegewinnung. Die prognostizierte Ausdehnung von KUP-Flächen in Europa wirft die Frage nach deren Wasserverbrauch auf. Dieser kann bei KUP höher sein als bei einjährigen, landwirtschaftlichen Kulturen oder bei Stilllegungsflächen, wobei sich gleichzeitig die Qualität des Sickerwassers verbessern kann. Basierend auf einer Literaturauswertung beschäftigt sich die vorliegende Arbeit mit Fragen zum Wasserhaushalt von KUP. Dabei liegt die Evapotranspiration (V_{ET}) von KUP deutlich über der des herkömmlichen Ackerbaus, schwankt aber stark in Abhängigkeit von der lokalen Niederschlagshöhe sowie anderen Standorts- und Bestandesbedingungen (z. B. Bodentyp, Temperatur, Grundwasserstand, Art/Sorte, Alter der Kultur) und ihren Wechselbeziehungen. Auswirkungen auf den Wasserhaushalt auf Einzugsgebietsebene sind bisher kaum untersucht worden. In Verbindung mit möglichen positiven Auswirkungen von KUP auf die Wasserqualität durch eine Minimierung der Nährstoffauswaschung ergibt sich bei Vermeidung eines erhöhten Wasserverbrauchs ein insgesamt positives ökologisches Potential für KUP. In diesem Zusammenhang sollten Vergleiche zur V_{ET} zwischen KUP und landwirtschaftlicher Nutzung sowie die Bezüge von V_{ET} zur Niederschlagsmenge und anderen Standortsparametern (Wurzelentwicklung, Grundwasserverfügbarkeit) besser herausgearbeitet werden, um die positiven Einflüsse von KUP auf den Grundwasserabfluss darzustellen. Weiterhin muss die Übertragbarkeit von Ergebnissen auf Wassereinzugsgebietsebene gefördert werden, um politische Entscheidungen bezüglich der Ausbreitung von KUP auch auf Marginalstandorten besser absichern zu können.

Schlüsselworte: Kurzumtrieb, Wasserverbrauch, Wasserqualität, Pappel, Weide, Biomasse, Bioenergie, Eutrophierung Grundwasserschutz

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Introduction

Cultivation of Short Rotation Coppice (SRC) with poplars (*Populus sp.*) and willows (*Salix sp.*) for production of biomass for heat and/or electricity has been identified as the most energy efficient carbon conversion technology for reducing greenhouse gas emissions and thus meeting the ambitious European commitments to increase the amount of renewable energy sources (Heller et al., 2004; Cocco, 2007; Styles and Jones, 2007). Moreover, SRC cultivation seems to coincide with other social, economic and environmental EU policies and objectives, such as the EU Rural Development, the CAP reform and the Water Framework Directive (EEA, 2006). Therefore, this combination of agricultural and energy drivers has stimulated interest in growing SRC as a source of renewable energy, and different incentives have been introduced in different European countries. Already in the short-term, a rapid increase of SRC in several European countries has been projected. For example, in Sweden the Swedish Board of Agriculture predicts a short-term increase of SRC to 30 000 ha (Jordbruksverket, 2006) and the UK Biomass Strategy predicts that perennial energy crops will occupy some 350 000 ha by 2020 (DEFRA, 2007). In Germany the present area under SRC cultivation is less than 1 500 ha (BMELV, 2008) but may also increase markedly during next years due to a changing subsidise policy and the identification of high cultivation potential for certain areas, e.g. 200 000 ha for the federal state Brandenburg according to Murach et al. (2008).

If the projected increase of agricultural land with SRC would be realised and a shift to SRC instead of other usual agricultural land use systems takes place, a range of positive and negative environmental implications will arise. For water-related issues, such implications are connected to questions concerning plant water use and water use efficiency, groundwater recharge, soil erosion as well as water quality. SRC as a perennial and woody crop differs from arable crops in a number of physical traits and thus its management is less comparable to other usual crops. In particular, SRC plantations are expected to remain in place for a number of years (10 to 25, depending on EU and national regulations, but also on management practices) taking the land out of arable rotations which implies no annual soil cultivation. Furthermore willow and poplar plants are thought to be deep-rooting and to have high water consumption, their height is ca. 5 to 8 m at harvest, and agrochemical inputs in the form of fertilizers and pesticides are minimal after SRC is once established. The anticipated increase of SRC will probably be concentrated in agricultural areas neighbouring power stations using biomass as fuel, since an approximate radius to a maximum of 50 to 100 km from customers should be considered as appropriate with respect to economically and energy efficient operation strategies (Rosenqvist and Dawson, 2005). This, depending on the size and the type of the power station, will be decisive for the "market" needs for SRC and therefore for the surface to be cultivated in a certain region.

The above indicate that a qualified estimation about the impact on water-related issues is rather complex when SRC is introduced instead of other crops in a certain area. The impacts of SRC will depend on a range of factors such as the specific site (climatic and soil conditions), the species used (willows or poplars), the management and the amounts of biomass produced, the age of the crop, the previous land use, and their interactions. SRC is a rather new cropping system and although considerable research in several countries has been conducted on water-related issues, there is still the need to combine the so far acquired knowledge to better understand the anticipated impact of SRC in small (field) and in large-scale (catchment).

This paper will give an overview of the available literature to estimate the expected impact of SRC on water-related issues. Although some direct impact on flooding prevention using SRC in buffer strips might exist (Kuzovkina et al., 2004; Volk et al., 2006) this paper will not deal with such issues but will mainly focus on the impact on water use and water quality. Implications for water balances and water quality issues when SRC is introduced in areas previously cultivated with arable crops and/or other crops will be discussed and analysed based on the related literature. Further, gaps in current knowledge will be identified, and therefore some future research aspects on water-related issues will be proposed.

SRC effects due to water use patterns

General about water balance for SRC

The decision behind using willow and poplar cultivated as SRC for production of biomass for energy was mainly based on the characteristic of these species to be fast-growing producing higher biomass compared to other tree species, especially in central and northern Europe where the development of such biomass systems was initiated and adapted commercially (Christersson and SennerbyForsse, 1994). Increased biomass accumulation has been linked with high water use, especially in warmer climates. Willows in particular are known to grow in places with high water availability such as river banks. Coupled with the fast-growing feature of SRC, fears for high water use and consequent concerns about the effects on local hydrological balances and flow to neighbouring streams/rivers have been expressed in several reports predicting future biomass supply from agriculture (EEA, 2006; Eppler et al., 2007; EEA, 2008).

In general, water balance of plant crops can be described by the water balance equation

$$N_n = N - V_I = W + V_{ET} + A_{Vs},$$

where

N_n	=	crop precipitation,
N	=	field precipitation,
V_I	=	interception,
W	=	change of water storage in ecological systems (soil, dead and living plants),
V_{ET}	=	evapotranspiration
A_{Vs}	=	runoff, including infiltration (Schaefer, 2003).

Evapotranspiration (V_{ET}) can be distinguished in transpiration (evaporation over stomata in the leaves of the plants) and evaporation (evaporation from bare soil or water surface). Precipitation remaining in the vegetation layer is termed as interception.

The special features of both willow and poplar that differentiate their water use patterns compared to other crops are the fast canopy development and the high leaf area index (LAI) during vegetation period. These affect significantly rates of transpiration from leaves and interception evaporation from the canopy. Furthermore potential deeper rooting of SRC species compared to annual crops might favour higher rates of water consumption. There are several difficulties to accurately determine the V_{ET} from any agricultural field, since a matter of factors such as air temperature, air humidity and wind force interfere. Nevertheless, a series of studies has been performed so far to estimate in precision the V_{ET} in SRC fields with the aim to speculate about changes in water balances in relation to other land uses. For willows, most of them were primarily conducted in Sweden, since it was there that cultivation of SRC for biomass was initiated and then commercially practiced, and later in the UK for the same reasons. Similar research on poplars has been conducted in a range of countries with more temperate climate than Sweden such as in Germany and the UK, where poplar has been considered as more appropriate species than willow grown as SRC, gaining large interest during the last years.

Transpiration and evapotranspiration rates of SRC

From the several estimates for V_{ET} for poplar and willow, there seem to be variations in the figures reported. For irrigated and fertilized willow SRC grown in clay in south Sweden for four years, Persson and Lindroth (1994) simulated seasonal (May to November) V_{ET} between 360 to 404 mm. Persson (1995) estimated that the average seasonal V_{ET} (May to October) from six fields in different locations in south Sweden areas was 435 mm, confirming in a way the previous findings. For SRC poplar fields,

Table 1:

Indicative reported evapotranspiration (V_{ET}) rates from poplar (P) and willow (W) SRC stands in different countries

Stand/ shoot age	Site	Soil	Species	Precipitation (mm)	V_{ET} (vegetation period)	V_{ET} (year)	Source	Country	Annotation.
9/9	Methau	loamy loess	Po	690 (lta)	480		1	GER	
3/3	Neuruppin	loamy sand	Po	585		356	2	GER	
9/9	Neuruppin	loamy sand	Po	585		359	2	GER	
Diverse	div	clay soil	Wi	700 (lta)		500	3	GB	
8/8	Welzow	clayy sand	Po	749		404	4	GER	
8/8	Welzow	clayy sand	Po	749		388	4	GER	
2/2 to 5/2	Uppsala	loamy clay	Wi	352 prec. +222 irrigation during vegetation period	435		5	SE	Irr, Fert, Mean
3/3	Börringe	sandy loam	Wi	586 (lta)	360		6	SE	
6/3	Alyckan	sandy loam	Wi	641 (lta)	440		6	SE	
7/2	Brinkendal	sandy loam	Wi	641 (lta)	374	481	6	SE	
X/2	Silsoe	sandy clay loam	Wi	574 (lta)		441	7	GB	
X/2	Selby	sandy clay loam	Wi	643 (lta)		462	7	GB	
X/2	Cirencester	sandy clay loam	Wi	776 (lta)		594	7	GB	

Sources: [1]Petzold et al., 2008; [2] Knur et al., 2007; [3] Hall, 2003; [4] Bungart et al., 2004; [5] Persson and Lindroth, 1994; , [6] Persson, 1995; [7] Stephens et al., 2001

lta – long-term average
Irr – irrigated
Fert – fertilized
Mean – mean value calculated from different sites
Po – poplar
Wi – willow

Bungart and Hüttl (2004) estimated mean annual transpiration rates between 1996 to 2002 at the Lusatian mining region in Germany equal to 266 and 241 mm, for two different poplar clones, respectively (Beaupré, *Populus trichocarpa* x *P. deltoides* and Androscoggin, *P. maximowiczii*). V_{ET} was 404 and 373 mm, respectively. Annual V_{ET} of 356 mm and 359 mm for a 3- and a 9-year old SRC poplar plantation, respectively, located in Neuruppin, Germany, has been calculated by Knur et al. (2007). Another 9-year old poplar plantation estimated to transpire 480 mm water during the vegetation period (April to November) in middle Saxony, Germany (Petzold et al., 2008). In the UK, Hall (2003) estimated that ca. 600 mm will be used by SRC willow in a clay soil which receives precipitation of 700 mm. According to Hall (2003) this corresponds to an annual V_{ET} of about 500 mm. Allen et al. (1999) estimated average daily transpiration rates of about 5 mm in poplar plantations in South England throughout the vegetation period, which is much higher than the previous reported values in Sweden and Germany. An indicative compilation of reported evapotranspiration values for willow and poplar SRC can be found in Table 1, showing the rather vast differences and factors that these might be depended on.

To add to the uncertainty about which species is a higher "consumer" of water, Linderson et al. (2007) found that the estimated transpiration rate of a willow stand with different clones in south Sweden from April to October was between 100 to 325 mm (therefore markedly lower than the previous findings for poplar), whereas the Penman-Monteith transpiration in that area for willow reached 400 to 450 mm for that period. This was attributed to the relative high temperatures in the summer when the measurements took place. Busch (2009 – see this issue) reveals that there is a trend between total annual evaporation (evapotranspiration including interception) and mean annual precipitation. With respect to all limits due to sparse data, methodological uncertainty and varying modelling approaches, the given data show a trend of increasing total annual evaporation with higher annual precipitation. Currently this trend could be only derived for a "window" of annual rainfall between 580 mm and 800 mm. This window, however, covers a range of annual precipitation which is typical for large regions in Central Europe.

The above V_{ET} estimations indicate that safe predictions for V_{ET} level from an SRC stand cannot be granted. Even stem sap flow measurements for calculating the daily transpiration rates gave varying results. Hall and Allen (1997) reported that transpiration of individual poplar plants grown in UK reached 9 ± 2 mm/day in June, and suggested based on these results that an effect on reduced stream flow and aquifer recharge and in reduced peak flows is to be expected. Others using similar methods, however, such as Hinckley et al. (1994), Allen et al. (1997) and Hall (1998) found that poplar transpiration rates in June/July were 4.8 mm/day, 5 ± 1.8 mm/day and 6 ± 0.5 mm/day, respectively, therefore significantly lower than the value of Hall and Allen (1997). Although willows seem to reach higher LAI earlier in the vegetation period (peak at 6.9 LAI) compared to poplars (peak at 4 LAI) (Stephens et al., 2001), there are only few reports to our knowledge suggesting that V_{ET} from willow is higher than poplars after direct comparison, e.g. a lysimeter test from Guidi et al. (2008). Hall et al. (1998) conducted sap flow measurements of willow and poplar plants planted in the same area and their conclusion was that transpiration rates for willow were, within measurement uncertainties, similar to those from poplar.

Aspects to be considered affecting SRC evapotranspiration

The reported vast variations in the transpiration rates from willow and poplar indicate that a number of aspects is probably affecting water use processes and water balances in SRC plantations in time and space. Based on the available literature, a short evaluation of most important aspects that should be accounted for when considering water balances of SRC is presented in the following.

Fast canopy development and high LAI in SRC during the vegetation period have been reported (Persson, 1995; Hall, 1998). For poplars grown in the UK for instance, it has been reported that LAI reaches 2 in mid-May and maximum around 4 after mid June. Willows in the same area developed leaves faster reaching LAI 3 in mid-April and maximum of 6.9 in mid-June, reducing to 2 by mid-October (Stephens et al., 2001). Consequently, the crop coefficient for the Penman equation is rather high throughout the year and so is the interception, which indicates that "water losses" will be high (high V_{ET}).

Interception of rainfall by foliage has been reported to be rather high in SRC, and therefore "water losses" due to interception are feared to be high with consequent effects on soil infiltration. Interception as a percentage of rainfall during the examined period was reported to be from as high as 31 % in willow SRC grown in Finland (Ettala, 1998), to 21 % in poplar SRC grown in the UK (Hall and Allen, 1997), to 18 % in poplar SRC in Germany (Bungart and Hüttl, 2004), down to 11 % in willow SRC in Sweden (Persson and Lindroth, 1994). Knur et al. (2007) calculated based on simulations an age-depended increase of interception losses from 20 % for three year poplar plantations to 29 % in nine year old plots under identical site conditions, referring to annual precipitation values. Thus variations depended mainly on stand ages, local conditions (higher leaf humidity results in higher interception) and the species examined, but are rather representative. Indicatively, the maximum rates for interception in food crop fields are ca. 15 % (Hall, 2003), therefore in most cases significantly lower than for SRC.

The degree of how much of the interception is a "water loss" or how well it substitutes transpiration (therefore not being a "loss") depends on the canopy height and structure. This is expressed in the radiation and aerodynamic terms of the Penman-Monteith equation. The decoupling factor (Ω) is the mathematical expression of this effect and has values from 0 up to 1, where 0 represents a vegetation fully coupled (higher plants having "evaporative demands" as a tall forest) and 1 a vegetation fully decoupled (short grasses and agricultural crops), respectively. There are indications that although willows and poplars grow high, their decoupling factor (Ω) is close to 1 when leaf area index is above 2 (Persson and Lindroth, 1994). As referred previously and in Persson and Lindroth (1994), LAI for willow and poplar for most of the vegetation period is above 2, and therefore interception should not entirely be considered as a "hydrological loss" since it seems to substitute transpiration to some extent.

The size and the formation in space of the field seem to be important for the expected V_{ET} of an SRC field. Allen et al. (1998) dealt with the relation between the width and height of SRC plantations and the crop coefficient in the Penman equation. The results show that the narrower the fields the higher the crop coefficient of plants of a certain height. It was also shown that this effect was accelerated when plants were higher. Therefore, if an SRC field is planted in narrow strips, which in reality results in edge effects with taller plants, the crop coefficient and consequently V_{ET} will be high. This is confirmed by Zhang et al. (1999) who reported low decoupling factor (Ω) for small-size poplar stands. When size increased, the decoupling increased indicating that evaporation would be lower (Hinckley et al., 1994). Therefore, small and narrow fields should be avoided for SRC in terms of water consumption. However, if SRC plantations are thought to be used as buffer zones for prevention of flooding in areas of high danger for such phenomena, then narrow strips parallel to the water course would have better results.

Root system development between willow and poplar seem to differ, since in general willow roots are more concentrated at the top compared to poplars; Rytter and Hansson (1996) suggested that most willow roots in a Swedish field were concentrated in the top 50 cm and Hall (2003) described that poplar roots were found down to 3 m depth. For Germany, Raissi et al. (2001) reported that for eight year old poplar stands the effective rooting depth ranged between 70 to 110 cm on sandy and loamy sites, thus nearer to the surface than Hall (2003) found. This does not necessarily mean that willows are not able to extract water from deeper soil depths in case of drought, but that probably poplars have another strategy for root development. However, when comparing willow SRC with other agricultural crops in the UK, Finch (2009) reported in a preliminary study that the "dense" willow root system does not go deeper than 30 to 40 cm and that "sparse" roots go to 1.5 m deep. In the same study, maize roots where at the same depth as willows, and winter wheat roots were only slightly less deep than these of willows indicating minor differences in the patterns for extracting groundwater between these species. It seems therefore rather unsure how differently the rooting development of SRC species would affect local water balances, since this will be depended on the groundwater level as well.

Water balances in SRC in relation to other crops

The impact of SRC on the water use and balances in a certain area must be judged in comparison with the crops that will be replaced in a potential shift to SRC. For most of the European countries future scenarios suggest that SRC will be established at productive agricultural land, therefore comparisons with arable crops would be relevant. In older literature, willow and poplar as tree species were usually compared to common forest species as well. The V_{ET} from SRC fields with willow and poplar is reported in most cases as significantly higher than arable crops and lower than other forest (Persson, 1995; Stephens, 2001; Hall, 2003; Knur et al., 2007). However, the estimated total evaporation from SRC in clay soils when adequately supplied with water was reported to be higher than this from mature coniferous forests (Persson and Lindroth, 1994). Hall (2003) reported that on a clay site with 700 mm rainfall SRC is expected to use 600 mm compared to 400 mm for barley and 450 mm for pine forest. In contrast to this, Hall et al. (1998) indicated that in case of dry summers when there is significant water deficit, the water use of poplar SRC will probably be considerably less than that of coniferous forest and closer to that of grassland. Sensitivity of willow SRC to dry summers have been also reported by Linderson et al. (2007), where transpiration rates varied between willow clones and were equal to 100 to 325 mm. Therefore, the levels of water consumption of SRC in relation to other crops grown in the same area seem to depend on site-specific factors as soil type, precipitation and others, and might vary from case to case, although SRC seem to have higher V_{ET} than arable crops in most cases.

In recent measurements however, preliminary published by Finch (2009), the cumulative transpiration difference between winter wheat and SRC willow for a 3-year period in the UK was only 50 mm (ca. 800 for winter wheat and 850 mm for SRC). Interception was on the other hand lower for SRC than winter wheat during 3 seasons (ca. 190 mm for SRC vs 220 mm for winter wheat), and therefore, the total difference in water use (or evaporation as the writer describes it) was very little between the two crops. This contradicts with results obtained in Germany stating that infiltration from poplar SRC fields was almost 3 times less than neighbouring arable fields (Knur et al., 2007). These results indicate that the water use by SRC in comparison to other crops largely depend on site-specific factors and probably on the methods chosen for calculation, and that probably the general perception that SRC "consume" significantly more water than other crops should not be generalised for all cases.

Implications for SRC effect on local hydrology

A range of studies indicated that the expected V_{ET} from SRC was elevated in comparison to this for other crops. Consequent implications for reduction of the water that reaches the soil and percolates in deeper layers resulted in negative assumptions concerning the effect of willow and poplar SRC on streams by Hall et al. (1998). The authors suggested that extensive establishment of SRC should be restricted to wetter parts of the UK, but they add that in case of warm summers, the water use of SRC will be similar to that of grassland. Perry et al. (2001) also predicted that poplar SRC in Northwest Minnesota may influence average peak flows in streams, after comparisons with much older hardwood stands and concluded that the same behaviour is to be expected. In this case however, the potential effect of SRC was claimed as positive reducing annual floods in the area. Allen et al. (1999) also assumed that introduction of poplar SRC in the UK will have adverse effects on water resources in the UK, and added that this impact will vary depending on rainfall and the alternative use as well.

Modelling exercises conducted by Stephens et al. (2001) indicated reductions of 10 to 15 % of the hydrologically effective rainfall in SRC fields compared to arable crops in the UK. Despite this, the authors claimed that the effect on hydrology to the catchment level would be minimal, after extrapolations based on the model results obtained and the assumption that 2 500 ha SRC will be planted in an area of 40 km radius from a biomass-driven power plant. This was due to the fact that the mean reduction in hydrologically effective rainfall for the catchment area would be ca. 0.5 % of the mean annual amount, which would be only a very minor portion, compared to the respective effect of cereals. Hall (2003) also suggested that even if SRC "consumes" more water than other arable crops, catchment scale effects of SRC on hydrology will be negligible, and that even when used as riparian buffers SRC will have little effect on river or streams due to low abstraction rates. He suggested however that in places where the potential biomass production from SRC exceeds 12 t DM/ha/yr, and precipitation happens to be lower than 550 mm, then these areas should be avoided for SRC planting since the consequences of reductions the hydrologically effective rainfall are much more serious in such areas.

As a conclusion, despite the predicted higher water "consumption" by willow and poplar SRC in the local (field) level, it seems that SRC poses no threats to the hydrology of a larger area e.g. having a distance from a biomass power plant up to 50 km, if the area is planted with 2 500 to 3 000 ha with SRC. Indicatively, and if we approximate, the referred area corresponds to ca. 30 % of the annual energy produced in a power plant in Sweden with capacity of 35 MW.

SRC effects on water quality

SRC is generally considered as a crop that improves the water quality in a certain area due to the management practices of this crop (EEA, 2008). Pest and weed control are conducted only during the establishment phase, and despite using chemical preparations, the total application rates considering the life-span of the crop (10 to 20 years) are minimal, especially comparing with the respective for arable crops. The appearance of fungus as leaf rust (*Melampsora sp.*) is rather common, especially in countries where the climate is humid for long periods, e.g. Northern Ireland, but even there application of fungicides is considered uneconomical and therefore not recommended (Dawson, 2007). Moreover, tillage is practiced only once before the establishment phase of the crop, and therefore soil disturbances that could enhance potential leaching to the groundwater of already existing hazardous chemical compounds (mainly N) are decreased. Due to all these facts, hardly any research on groundwater quality in SRC fields has been conducted aiming at determination of chemical compounds as pesticides and heavy metals in the groundwater.

To achieve high biomass production, recommendations for inorganic fertilization to SRC fields have been developed in different countries (for Sweden Ledin et al., 1994; for the UK DEFRA, 2002; for Mecklenburg-Vorpommern Germany Boelcke 2006; for Northern Ireland Dawson, 2007). Fertilization during the year of establishment is not recommended to avoid competition with weeds and potential leaching to the groundwater when the roots are not well-developed. For the next years, there are significant variations in the different recommendations for fertilization applications rates; for example, in the UK, ca. 70 kg N on average per hectare and year are proposed to be applied depending on the year of the cutting cycle. In Northern Ireland the respective amount is 120 to 150 kg N per hectare and year, and in Sweden the recommendations is ca. 100 kg N per hectare and year, but not after the second year of growth when no fertilization occurs. In general, SRC fertilization recommendations can be considered rather moderate compared to respective ones for arable crops (Hofmann, 2007). However, the fact that fertilization to SRC fields cannot be applied every year but usually only every year after harvest due to the nature of the crop (high stems that do not allow the available equipment to apply fertilizer every year) makes the applied amounts of nutrients relatively high. Therefore, to estimate any danger from N leaching in the groundwater, relevant information from intensively irrigated field with N should be collected, taking also into consideration potential critical management factors such as the time of N-application and methods selected (e.g. use of nitrification inhibitor).

Bergström and Johansson (1992) measured very low N concentrations (less than 1 mg/l N) in the groundwater of an intensively fertilized willow SRC field in south Sweden. Measurements of N in the surface groundwater at the same field for a period of eight years with average annual application rates of 112 kg N/ha showed that N concentrations remained below 1 mg/N for the whole period except during the year of establishment (Aronsson et al., 2000). These results came in agreement with Mortensen et al. (1998) that measured close to zero N concentrations in drainage water from Danish SRC fields, except for the establishment year. The maximum N concentrations in the drainage water that year was up to ca. 100 mg/l N for fertilized plots with 75 kg N, but were high even for control plots that did not receive any N (maximum ca. 60 mg/l N). This indicates that extensive mineralization probably occurs during the establishment phase. Goodlass et al. (2007) also reported high N concentrations in drainage water during the establishment phase, with reduction to low levels after the crop is established despite application of 200 kg N/ha in 3 years. In the same study, the authors studied the N concentrations in the drainage water after removing a poplar SRC field and found them equally high as for the establishment phase, and therefore concluded that SRC stands should be maintained for as long as possible. This conclusion was partly made after comparisons with the maximum N concentrations in the drainage water from arable crops in the area, which consistently exceeded 60 mg/l every year. Big differences in the amounts of N leached in the groundwater between SRC and a series of arable crops were reported in Denmark by Jørgensen and Hansen (1998). In a sandy soil, ca. 15 kg N/ha were leached on average from fertilized willow SRC, whereas the respective value for different cereals was between 70 to 120 kg N/ha.

Although differences in N leaching to groundwater from SRC compared to the reported "usual" N leaching figures from arable crops are rather striking, they could be attributed in some cases to the lower input of fertilizer applied to SRC compared to "normal" fertilization rates for arable crops. Moreover, in some cases some arable crops might be also irrigated in case of water shortage in summer, making comparisons difficult. To examine if SRC is equally good in N-leaching performance under circumstances with higher N amounts fertilized and when irrigation occurs, it is worth comparing results from related work of wastewater irrigation to SRC. This is a method used for treating and utilising nutrient-rich wastewaters (usually in N but also P) by irrigation to SRC fields, which has gained interest during the recent years in countries where SRC cultivation is rather common (Aronsson, 2000; Sugiura et al., 2008; Werner and McCracken, 2008). According to Aronsson (2000) after testing different irrigation regimes with wastewater under different soil conditions, wastewater application at least 150 kg N/ha yr should not pose any threat to extensive NO_3-N leaching in Sweden. Concentrations of N in the drainage water below 5 mg/l were recorded in an experimental willow SRC field in N. Ireland where ca. 200 kg N/ha/yr were applied (Werner and McCracken, 2008). Moreover, Sugiura et al. (2008) applied much higher amounts (ca. 300 kg N/ha/yr) and N concentrations in the drainage water at different depths was between 5 to 10 mg/l. This figure is rather low considering the high application rate and comparing with findings for other arable crops. The above findings suggest that in general leaching of N from SRC in comparison to arable crops is significantly lower and a shift from arable crops to SRC will probably imply an improvement of the groundwater quality and consequently of the surface water quality in a certain area, even when N fertilization exceeds the recommendations for good practice.

Application of municipal sewage sludge to SRC fields is a common practice in Sweden and in the UK that compensates P losses in newly harvested fields (Sagoo, 2004; Dimitriou and Aronsson, 2005). The application rates of N with sludge can vary depending on each country's legislation, but in most cases it is P that is considered potentially more dangerous than N; although in most cases legislation regulates the P rates, ca. 150 kg N/ha can be applied to SRC fields with a sludge application (Dimitriou, 2005). Leaching of NO_3-N to the drainage water has been reported during the application year of sewage sludge when applied to a willow SRC fields (Labrecque and Teodorescu, 2001; Sagoo, 2004). However, N-leaching was very low (below 10 mg/l) the subsequent years and results were similar for SRC fields treated with conventional inorganic fertilizers or wastewater, suggesting that N-leaching after sludge application should not be considered as a problem. Judging by the application rates of P when sewage sludge is applied to SRC (e.g. in Sweden 22 or 35 kg P/ha/yr depending on the P content existing in the soil, for a 7-year period), and from the fact that willows can accumulate in the stems ca.

8 kg P/ha/yr depending on the site, a surplus of P is usually applied with sewage sludge. Despite this, P is usually bound to soil particles and its leaching patterns differ that these of N which is in most cases related to water drainage. Therefore, P leaching when sewage sludge is applied to SRC should not be potentially considered a problem. Preliminary results by Dimitriou and Aronsson (submitted to Biomass and Bioenergy) from sewage sludge-amended lysimeters confirm the above speculation since P concentrations in the drainage water was close to zero throughout measurements during two vegetations periods. However, future P leaching cannot be excluded as a possible scenario when sewage sludge is applied for a number of years at high rates.

All the above about water quality and SRC indicate that when SRC replaces arable crops, an improvement of the groundwater quality is anticipated. In fact, several suggest the use of SRC in intensively to managed agricultural areas to improve the current water quality and meet EU obligations in terms of water quality expressed in the Water Framework Directive (Jørgensen and Mortensen, 2000; Eppler et al., 2007; EEA, 2008) and simultaneously use the land for agricultural production for biomass for energy that also fulfils other obligations concerning renewable energy.

Conclusions and future research aspects

The special features of SRC as a crop have implications to water balance and water quality. The main aim with SRC plantations is to produce high biomass, but this can imply potential reductions of water availability, especially in places where precipitation is low or even when dry summers occur in areas with otherwise adequate precipitation. This will consequently have an effect on groundwater enrichment, but this effect needs to be balanced with the biomass produced and compared with effects of alternatives to SRC crops in the area. Potential benefits on water quality must be also taken into account when judging SRC compared to other crops.

Reported V_{ET} of SRC vary markedly depending on the location (precipitation, soil type), selected species/clones, plant age and the climatic conditions during the estimation periods. Differentiations between the different estimation methods/models used and actual measurements occur as well. Willow and poplar V_{ET} seem not to vary much between each other, but have been in most cases calculated to be higher than other arable crops. There are indications that in areas receiving precipitation less than 600 mm, poplar and willow SRC should be avoided since effects on both biomass and on groundwater recharge would be probably negative. However, although in the local (field) scale effects on water balances might be evident due to SRC, in the large (catchment) scale these effects seem not to be transferable and the total effect on water balances in a larger area around e.g. a combined power station using biomass has been judged as negligible or very marginal. This, combined with the considerable improvements of groundwater quality in terms of leached nutrients (and as a consequence surface water quality) when SRC replaces intensively managed arable crops (a projected scenario for many SRC in Europe), leads to promising conclusions in terms of SRC impact on water issues.

However, potential negative effects attributed to SRC should be minimized and uncertainties that could hinder future decision-making about SRC should be clarified. Therefore, better and more extended comparisons of V_{ET} between SRC and arable crops, studying SRC fields grown for several years and are well-documented compared with respective arable crops in the same area, should be conducted. For this, certain measurements such as transpiration and rooting depth should be more precise and effects correlated to local conditions should be better examined (e.g. root function and depth in correlation to groundwater level). Also, an approach integrating effects of soil SRC issues on water (soil type or mycorrhizae abundance and their role) should be applied. Concerning water quality issues, and despite the very positive impact of SRC, the effects on groundwater quality of common management practices such as the application of sewage sludge and/or wood-ash to SRC fields should be studied in long-term perspective.

All these local effects on water issues must be extrapolated to the larger scale estimating potential effects of a hypothetical introduction of SRC in high percentage of a certain agricultural area. These results should constitute the base for recommendations and future decision-making processes about SRC establishment in comparison to other arable crops. Water related investigations in SRC can be helpful also considering climate change issues and the fact that trees can survive and adapt better than arable crops when drought conditions occur – preconditioned that rooting systems reach deeper soil layers. With regard to future scenarios and changes in agricultural policies and priorities, studies related to the use of marginal soils for poplar or willow SRC production should be also included as a future research aspect, despite the current situation pointing at replacement of productive agricultural soils.

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The impact of Short Rotation Coppice cultivation on groundwater recharge – a spatial (planning) perspective

Gerald Busch*

Abstract

The potential impact of willow/poplar SRC on groundwater recharge (GWR) of arable land is an important issue for spatial environmental assessments. The approach presented in this study aims at an impact assessment for regional planning purposes. Based on a linear regression of annual precipitation to actual evapotranspiration (V_{ET}), an upper L_h ($r^2 = 0.89$) and lower L_l ($r^2 = 0.74$) boundary of annual SRC water use was determined. The impact assessment made use of public available data on land cover (ATKIS) soil properties (BÜK50) and regional climate (precipitation grids). The methodology refers to well established rule-sets applied to the soil information system of Lower Saxony, Germany (NIBIS). Results are presented for two municipalities in the district of Uelzen, Northern Germany. The study area is characterized by sandy soils, a low groundwater table and precipitation ranging from 607 to 777 mm a^{-1}.

V_{ET} calculated for the study area ranges from 560 to 714 mm a^{-1} for (L_h) and from 488 to 544 mm a^{-1} for (L_l), respectively. A considerable water deficit calculated for (L_h) conditions bears a high risk of yield decline. Compared to current agricultural land use, GWR of (L_h) substantially decreases by about 70 %. In case of (L_l), GWR is reduced by about 5 to 10 % compared to current agricultural land use. Thus, from the perspective of ground water protection the focus should be on SRC systems representing (L_l) conditions.

The underlying data indicate that willow (poplar) SRC with 2 to 4 year rotation period is typical for (L_l) conditions. There is, however, considerable uncertainty and an urgent need to improve the data base. Representing (L_l) conditions, SRC could be an interesting land use option due to its potential positive impact on soil erosion groundwater quality and landscape structure.

Keywords: SRC, spatial planning, qualitative assessment, water use, ground water recharge, evapotranspiration, interception

Zusammenfassung

Potentielle Auswirkungen von Kurzumtriebsplantagen (KUP) auf die Grundwasserneubildung (GWN) – Eine Landschafts(planungs)perspektive

Die potentielle Auswirkung auf die Grundwasserneubildung durch den Anbau von Pappel und Weide auf Ackerstandorten ist ein wichtiges Kriterium in der Umweltbewertung von KUP. Es wird ein planungsrelevanter Ansatz präsentiert, der es erlaubt, diesen Einfluss quantitativ und qualitativ zu bewerten. Zwei lineare Regressionen ($r^2 = 0{,}88$; $r^2 = 0{,}74$) von Jahresniederschlag und aktueller Evapotranspiration (ET_a) determinieren den oberen (L_h) und unteren Grenzbereich (L_l) des annuellen Wasserverbrauchs von KUP. Die Berechnungen zum Einfluss auf die Grundwasserneubildung (GWN) nutzen allgemein verfügbare Daten zu Landnutzung (ATKIS), Boden (BÜK50) und Klima (Regionalisierte Niederschläge). Für die Berechnung kommen etablierte Methoden des NIBIS zum Einsatz. Am Beispiel von zwei Samtgemeinden (Suderburg und Rosche) des Landkreises Uelzen werden die Ergebnisse der Bewertung dargestellt. Das Untersuchungsgebiet weist vornehmlich sandige Böden und einen niedrigen Grundwasserspiegel auf. Bei Jahresniederschlägen von 607 bis 777 mm a^{-1} ergibt sich eine ET_a von 560 bis 714 mm a^{-1} für (L_h), bzw. von 488 bis 544 mm a^{-1} für (L_l). Das hohe Wasserdefizit während der Vegetationsperiode führt zu beträchtlichen Ertragsrisiken für (L_h). Im Vergleich zur aktuellen landwirtschaftlichen Nutzung reduziert sich für (L_h) die GWN im Mittel um 70 %. Für (L_l) beträgt die Verringerung lediglich 5 bis 10 %.

KUP, die den Fall (L_l) widerspiegeln, stellen in weiten Bereichen keinen bedenklichen Einfluss auf die GWN dar und sollten daher bevorzugt werden. Sie könnten zudem genutzt werden, um gezielt Erosionsschutz zu betreiben, die Landschaftsstruktur zu verbessern und die Grundwasserqualität zu sichern. Die vorhandene Datenbasis deutet an, dass Weide mit Rotationsperioden von 2 bis 4 Jahren die (L_l)-Variante repräsentieren – hier besteht aber erheblicher Untersuchungsbedarf.

Schlüsselwörter: Kurzumtriebsplantagen, Grundwasserneubildung, aktuelle Evapotranspiration, räumliche Bewertung, Wasserbedarf

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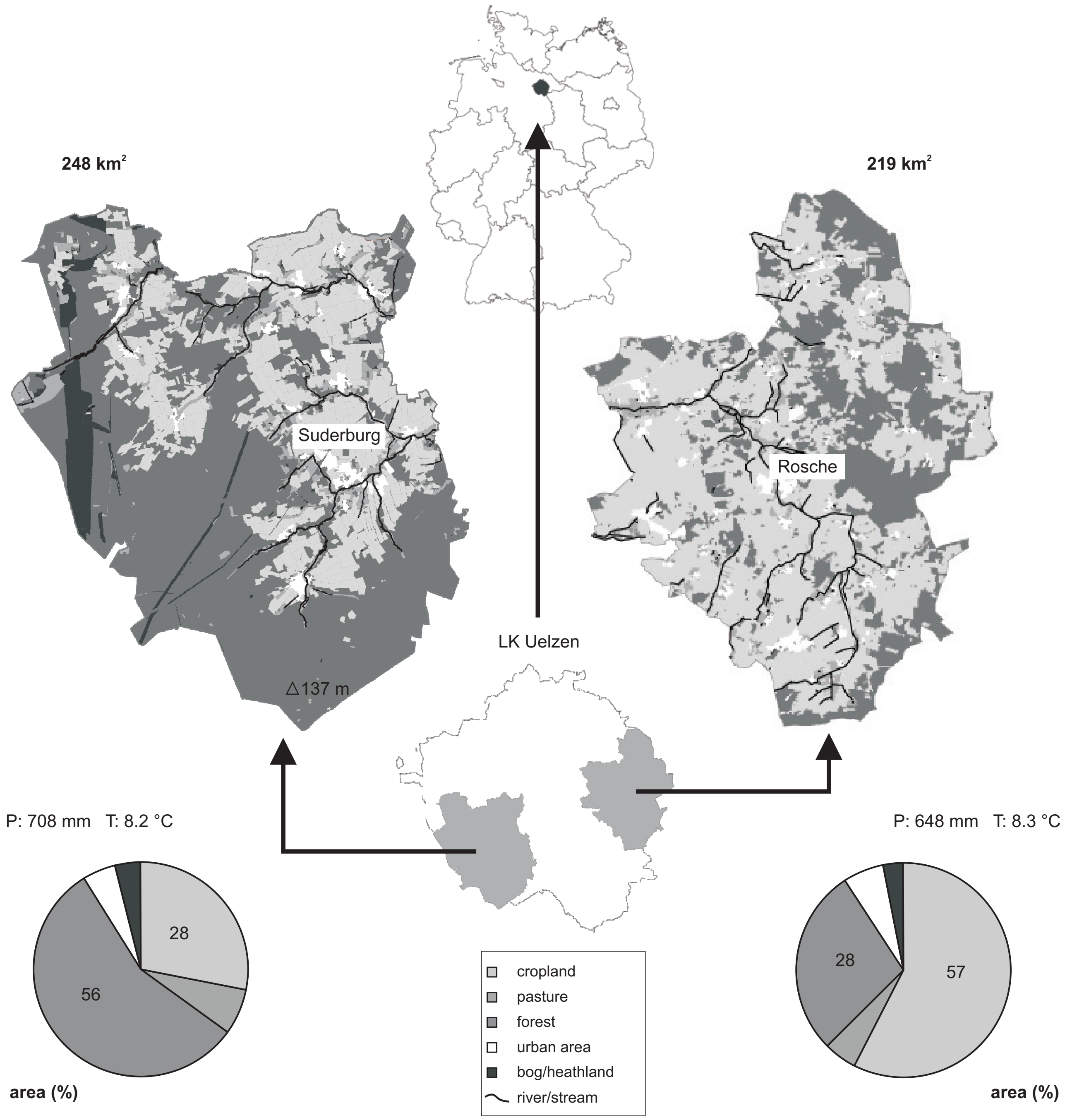
Introduction

Due to national (e.g. EEG, Biomasseverordnung) and international incentives (EU – new Renewable Energy directive) and increasing energy prices biomass crop cultivation spread to some 2 Mio ha in Germany until 2007. This area matches with 17 % of arable land in Germany (FNR, 2007). Various energy scenarios indicate options of up to 2 to 5 mio ha cropped additionally with biomass plants until 2030 (Simon et al., 2005; Fritsche und Wiegmann, 2005; Fritsche and Dehoust, 2004). Until now political incentives have focused on rape seed (biodiesel) and maize (biogas). Recently, supported by energy suppliers and industries of the energy sector (e.g. RWE = 10.000 ha, www.rwe.com; Schellinger KG = 5000 ha), the establishment of SRC fields has been promoted intensively. Estimating a proportion of



Source: ATKIS data - (AdV, 2003); BGR/NLFB, 2004

Figure 1:

Major Land cover types in the municipalities of Suderburg and Rosche

10 % of future potential biomass area in Germany would result in 200.000 to 500.000 ha of SRC fields. Since the established clones of willow and poplar are expected to show high annual water demand compared to conventional agricultural crops (e.g. Dimitriou et al., 2009; Knur et al., 2007; Hall, 2003; Stephens et al., 2001) the potential impact on groundwater recharge is of crucial importance.

The extensive dynamic of land use change from 2002 to 2007 revealed that ther are needs to be adapted to this process within the framework of existing ecological and spatial planning. To do so, tools are needed that allow for a rapid qualitative assessment even if the data material is sparse and the knowledge about potential impacts is limited. Both holds true for the potential impacts of SRC on groundwater recharge. Based on the statistical interpretation of currently available data on water use of SRC, this study presents a straight-forward concept to assess SRC impact on groundwater recharge compared to conventional cropping. Using public-available data on soil properties, precipitation patterns and land cover classification, this approach builds on existing methods of ecological landscape analysis, and spatial planning (Marks et al., 1992; Bastian und Schreiber, 1999, Müller, 2004; von Haaren, 2004). The approach was applied to two municipalities in the district of Uelzen, Northern Germany. As a result of the GIS analysis the potential impact of SRC could be spatially addressed and qualitatively assessed.

Material and methods

Study area

The climate of the study area is humid temperate oceanic with a mean annual temperature of 8.1 to 8.2° and mean annual precipitation ranging from 607 to 777 mm (BGR/NLFB, 2004). The existing hydrological system, morphological habit and soil association originate in pleistocene glacial processes (Saale glacial). Thus the landscape is characterized by an undulating terrain with some steeper slope in terminal moraine areas. An overall West-East-gradient of decreasing precipitation is modified by leeward effects of the terminal moraine margin. Soils are dominated by a sandy texture with increasing proportion of silt and clay in ground moraine regions. Apart from meadow areas the groundwater table is low. The very poor sandy soils were afforested with pine after World War II. Heath, associated bogs and fens still cover a small percentage of land and are of great nature conservation value. The agricultural area focuses on more productive soils situated in ground moraine regions. Maximum height of the study area is 137 m (South-West Suderburg). The cropland area, as a marginal part of the "Uelzen-basin", varies between 42 m and 75 m of height. In Suderburg, however, cropland covers only 28 % of the municipality (see Figure 1). Current agricultural land use is dominated by cropland with a crop mix of barley, potato, wheat, and sugar beet. In the district of Uelzen cropland irrigation is common practice. About 81 % of arable land is currently being irrigated with a mean annual groundwater use of 76 mm (Fricke, 2006; 2008).

Derivation of SRC actual evapotranspiration (V_{ET})

Based on an extensive literature review, available information on actual evapotranspiration (V_{ET}) of SRC was compiled. The data basis, however, is quite limited (see Dimitriou et al., 2009). The underlying data sources refer to various sites in Germany, Great Britain and Sweden, representing different climatic conditions and site qualities. The utmost data given in literature are modeling results stemming from different models, varying assumptions concerning the input parameters and a distinct model parametrization. For a detailed description regarding the review of SRC evapotranspiration see Dimitriou et al., 2009.

In total, 15 data sets were identified representing both non-irrigated and non-fertilized poplar and willow stands of different age (see Table 1 and Table 2). For some data sets, annual V_{ET} was derived from actual evapotranspiration during the vegetation period using a coefficient of 1.222 (Hörmann et al., 2007; Mayer, 2005; Nützmann, 2004) depending on the length of the vegetation period referred to. The data were separated into two groups:

Group 1 (Table 1) represents SRC with longer rotation periods, fully developed rooting extension or precipitation numbers that correspond to long-term average precipitation numbers.

Group 2 (Table 2) depicts the given data with newly established SRC fields or precipitation data that clearly deviate from the long-term average. In case of the Welzow site extraordinaray high precipitation numbers are associated with fairly high deep percolation rates (Bungart and Hüttl, 2004).

For both data groups, a linear regression was used to analyse the change of annual evapotranspiration (V_{ET}) against annual precipitation (P). The resulting linear functions were interpreted as boundary lines that cover a "corridor" of possible V_{ET} pathways (see Figure 2). The upper boundary line – which is representing "Group1-data – is addressed as"L_h variant" in the results section. The lower boundary line is referred to as "L_l variant".

Table 1:

Site characteristics of poplar and willow SRC – Group 1

stand/ shoot age	Site	Soil	Species	P (mm)	$V_E + V_T$ (veg)	$V_E + V_T$ (year)	V_I	V_{ET}	Source	Country
9/9	Neuruppin	loamy sand	Po	580 (c)		359	170	529	2	GER
9/9	Lindenberg	loamy sand	Po	634 (c)		393	171	564	2	GER
div	div	clay soil	Wi	700 (lta)		500	140	640	3	GB
6/3	Alyckan	sandy loam	Wi	641 (lta)	440	550 (e)	40	590	4	SE
x/2	Silsoe	sandy-loamy clay	Wi	574 (lta)		441	125	566	5	GB
x/2	Selby	sandy clay loam	Wi	643 (lta)		462	130	592	5	GB
x/2	Cirencester	sandy clay loam	Wi	776 (lta)		594	140	734	5	GB

[2] Knur et al., 2007,3 Hall, 2003,4 Persson, 1995,5 Stephens et al., 2001

x – subsequent rotation period
c - corrected
e – extrapolated
Po – poplar
Wi – willow
lta – longe-term average
veg – vegetation period
V_E evaporation
V_T transpiration
V_I interception
V_{ET} evapotranspiration

Table2:

Site characteristics of poplar and willow SRC – Group 2

stand/ shoot age	Site	Soil	Species	P (mm)	$V_E + V_T$ (veg)	$V_E + V_T$ (year)	V_I	V_{ET}	Country	Source
3/3	Neuruppin	loamy sand	Po	580 (c)		356	117	472	GER	2
3/3	Börringe	sandy loam	Wi	586	360	439 (e.)	30	469	SE	4
7/2	Brinkendahl	sandy loam	Wi	641 (lta)	374	456 (e.)	59	515	SE	4
8/8	Welzow	clayy sand	Po	749 (lta 570)		404	138	542	SE	6
8/8	Welzow	clayy sand	Po	749 (lta 570)		388	132	520	GER	6
3/3	Schönberg	sandy loam	Po	782				572	GER	7
3/3	Schönberg	sandy loam	Wi	782				500	GER	7

[2] Knur et al., 2007, [4] Persson, 1995, , [6] Bungart and Hüttl, 2004,[7] Kahle et al., 2005

x – subsequent rotation period
c - corrected
e – extrapolated
Po – poplar
Wi – willow
lta – longe-term average
veg – vegetation period
V_E evaporation
V_T transpiration
V_I interception
V_{ET} evapotranspiration

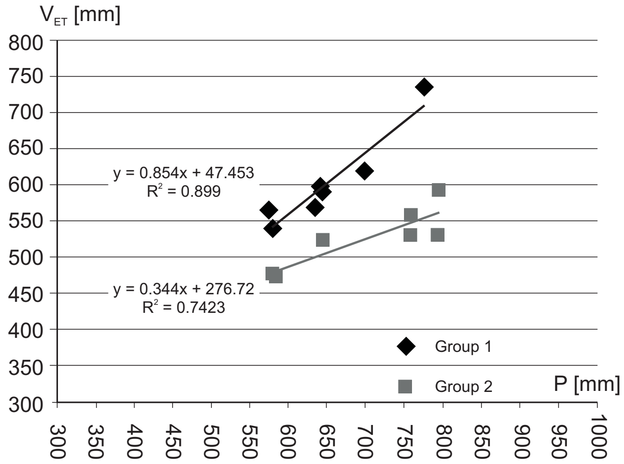


Figure 2:

Correlation of annual precipitation and actual evapotranspiration (V_{ET}) for different SRC sites

Calculation of groundwater change due to SRC

Precipitation patterns

The precipitation data for the study area were taken from NIBIS (BGR/NLfB, 2004; Müller, 2004). The data reflect a regionalized grid of monthly precipitation patterns with a grid size of 50 m² and are based on long-term measurements (1961 to 1990) provided by "Deutscher Wetterdienst". The data were summed up to annual precipitation numbers and to values referring to the vegetation period (1.4. to 31.10.) The vegetation period for SRC plants is clone-specific and depends on climatic boundary conditions. However, this vegetation period was chosen to cover the range of values taken from the international literature (e.g. Persson, 1995; Stephens et al., 2001; Persson and Lindroth, 1994). In a second step, precipitation patterns were intersected with current cropland area to cover potential sites of SRC. Current cropland area was derived from ATKIS (Amtlich-Topographisches-Kartographisches Informationssystem) provided by LGN (Landesvermessung+Geobasisinformation Niedersachsen).

Plant available water (PAW) and total plant available water (TPAW)

Plant available water was calculated by multiplying the available soil water capacity (AWC) by the plant's rooting depth. AWC was determined by using the database of soil maps (scale of 1:50.000) provided by the Federal Institute of Geosciences and Natural Resources (BGR). The multi-step calculation of PAW followed a well established rule-set supplied by the State Authority for Mining, Energy and Geology (Müller, 2004). TPAW comprises both, available soil water capacity and capillary rise from the water table. Again, the calculation followed the rule-set of Müller (2004). Effective rooting depth was calculated for 2 variants:

(1) A rooting depth ranging between 90 to 150 cm – this reflects poplar stands with longer rotation periods and different soil texture according to Raissi et al. (2001).
(2) A rooting depth ranging between 55 cm and 100 cm – referring to data obtained from willow sites by Persson (1994), and Rytter and Hansson (1996).

Soil water demand

By balancing actual evapotranspiration (V_{ETveg}) and precipitation (P_{veg}) for the vegetation period, the soil water demand (SW_{dem}) was calculated for both, the (L_h) and the (L_l) variant (Step 1 – Figure 3). According to Hörmann et al. (2007), Mayer (2005) and Nützmann (2004), a ratio of V_{ETveg}/V_{ET} of 0.85 was applied. The resulting balance was set off against total plant available water (for 2 root extensions) to calculate soil water demand of SRC (SW_{dem} – see Step 2 - Figure 3).

Soil water deficit and potential risk of yield decline

In general, plants could only use 50 to 70 % of the TPAW without reducing transpiration and growth (Wohlrab et al., 1992; Frede, 2006). Further, deep percolation of precipitation water during the vegetation period reduces the available precipitation water for SRC. To cover this aspect for an evaluation of potential soil water deficit, a 10 % percolation rate of precipitation during the vegetation period was assumed to take effect. This is in accordance with data provided by Knur et al., (2007); Persson, (1995); and Bungart and Hüttl (2004). Thus, a potential soil water deficit occurs, if soil water demand plus percolation rate exceeds 70 % of TPAW.

STEP	Group I (rooting depth 55 to 90 cm) Group II (rooting depth 70 to150 cm)
I	$P_{Veg} - V_{ETveg} = SW_{dem}$
II	$SW_{dem} + TPAW = SW_{use}$
III	$P_{nVeg} - SW_{use} - V_{ETnveg} = GWR_{pot}$
IV	$GWR_{pot}/GWR_{crops} \cdot 100 = GWR_{change}$ %

Figure 3:

Analysis steps for the calculation of change in GWR

Ground water recharge (GWR) of current land use

Again, the calculation followed a rule-set developed by Müller (2004) that is officially used for planning purposes. Major input parameters for this calculation are: total plant available water, precipitation, potential evapotranspiration (V_{Etp}). The calculation of V_{Etp} refers to a typical crop-mix based on municipal agricultural statistics. As an input for the qualitative assessment of the environmental impact analysis, the results were subdivided into 5 classes ranging from very low to very high according to Marks et al. (1992).

Ground water recharge (GWR) of potential SRC sites

To calculate the amount of groundwater recharge from SRC fields (GWR_{pot}), used TPAW (SW_{use}) during the vegetation period was balanced against both, precipitation and actual evapotranspiration of the non-vegetation period (V_{ETnveg}). In a last step (see Figure 3), the overlay of groundwater recharge from SRC and current agricultural land use (GWRcrops) produced relative change of groundwater recharge due to SRC on arable land.

Ecological impact analysis

The ecological impact analysis is an established tool for qualitative assessments in regional planning exercises (von Haaren, 2006; Jessel and Tobias, 2002). In this study, the approach was used to evaluate the potential impact of SRC on groundwater recharge. The ecological impact analysis is based on a matrix approach (see Figure 4) that builds on sensitivity of a landscape function (GWR on the y-axis) versus impact intensity (relative decline in GWR due to SRC on the x-axis). As a result it is possible to evaluate qualitatively the impairment of an ecological function (groundwater recharge). Thus, the qualitative deduction always reflects a mental model and/or ecological goals which could be region-specific. The rationale behind the qualitative assessment in this study was to distinguish between 5 classes ranging from very low to very high potential damage (Figure 4). Very low to low impact on GWR was assessed to be tolerable (see Figure 4).

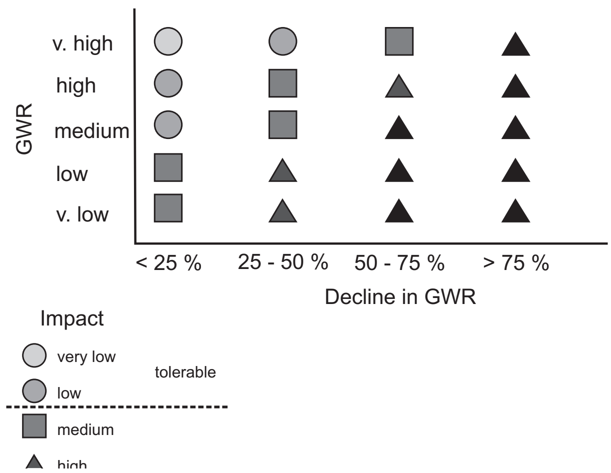


Figure 4:
Assessment matrix of the ecological impact analysis

Results and discussion

V_{ET}

Annual V_{ET} of the (L_h) variant ranges between 610 and 714 mm in Suderburg and between 560 and 638 mm in Rosche, respectively. Mean V_{ET} in Suderburg is about 57 mm a^{-1} higher compared to Rosche. Figure 5a depicts the SW-NE gradient of V_{ET} in Suderburg. In Rosche, the central and southern part show lowest values of annual V_{ET}. Due to the lower gradient of the statistical regression in the (L_l) variant, V_{ET} values do not spread as much as shown for the (L_h) variant. Mean annual V_{ET} in Suderburg is only about 20 mm higher than in Rosche. Annual V_{ET} ranges between 503 and 544 mm in Suderburg and between 485 and 514 mm a^{-1} in Rosche, respectively (see Figure 5b). The difference in mean annual V_{ET} for the two variants (L_h) and (L_l) is 130 mm in Suderburg and 93 mm in Rosche.

Soil water demand

Precipitation was balanced with V_{ET} for the vegetation period to calculate for the soil water demand.Figure 6 depicts that there is a considerable difference in water deficit for the two SRC variants. Mean V_{ET}veg (L_h) was calculated to be 550 mm in Suderburg and 502 mm for Rosche. The resulting demand for plant available water ranges from 107 to 153 mm – with man values of 131 mm and 112 mm, respectively (see Figure 6a). An average TPAW of 169 mm in Suderburg and 185 mm in Rosche, respectively covers the soil water demand during the vetation period on 98 % of the cropland area. For the (L_l) variant, mean V_{ETveg} only slightly varies between 423 mm in Suderburg and 420 mm in Rosche. Soil water demand is con-

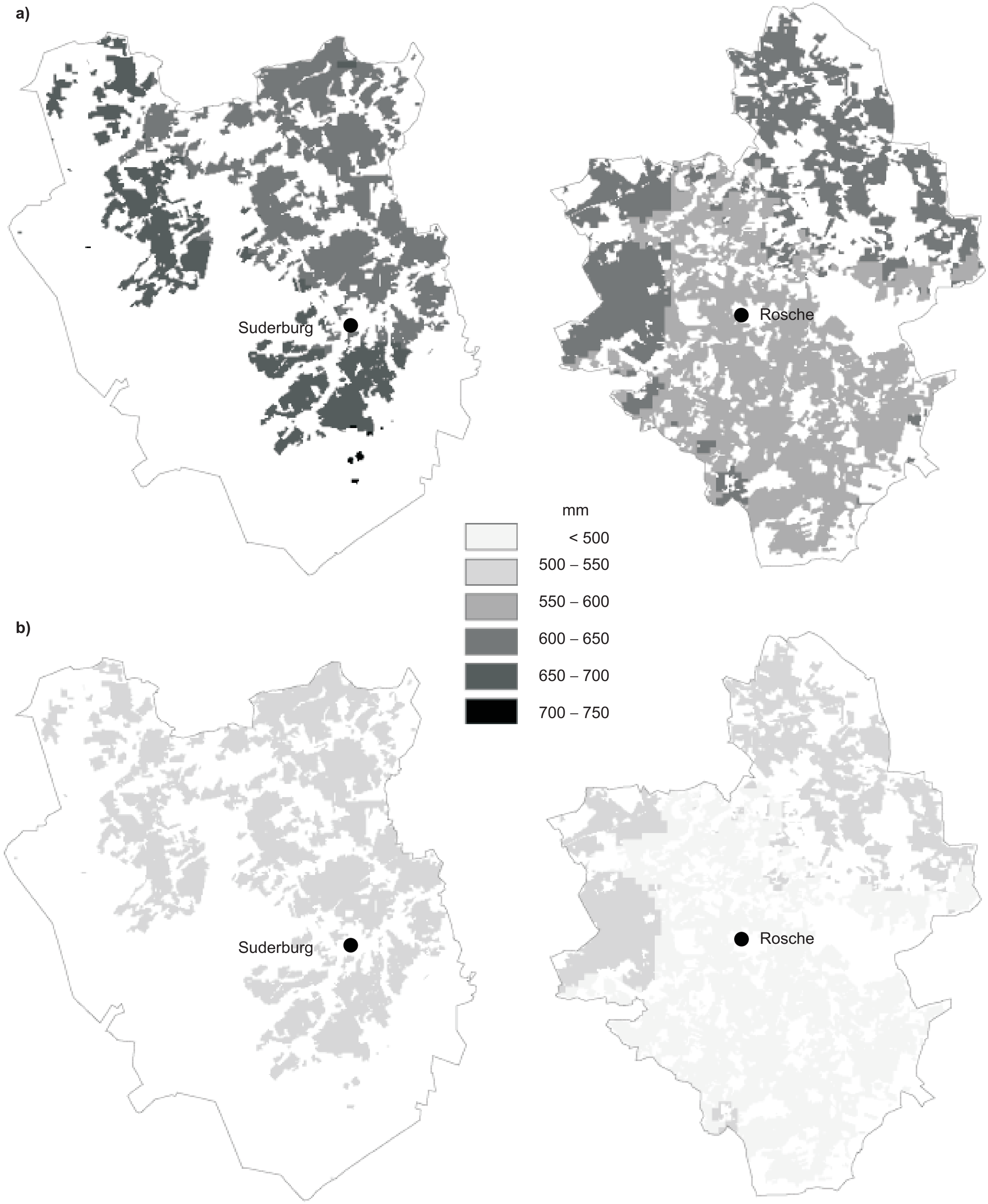


Figure 5:

Annual actual evapotranspiration of potential SRC sites on cropland in the municipalities of Suderburg and Rosche. a) L_h variant b) L_l variant siderably lower – with mean values of 8 mm and 30 mm, respectively. In Suderburg about 30 % of cropland area even shows a small surplus in the balance between V_{ET} and precipitation during the vegetation period (see Figure 6b). This demand is balanced by a mean TPAW of 108 to 114 mm in both municipalities. Note, that only 2 % in Suderburg, respectively 6 % of the cropland area in Rosche gains a profit from the groundwater table.

Soil water deficit and potential risk of yield decline

At first sight, SRC demand for soil water is fully balanced for both variants. However, to answer the question if the the amount of TPAW is sufficient to sustain stable yields, both, losses of percolation water and plant physiological restrictions have to be considered. On average, a 10 % loss of precipitation (P_{veg}) due to deep percolation accounts for 38 mm in Suderburg and 34 mm in Rosche, respectively. In Suderburg, 131 mm of soil water demand plus 38 mm of percolation water need to be balanced by at least 241 mm of TPAW (L_h variant). Thus, 169 mm of TPAW (area-weighted average) results in a potential soil water deficit of about 70 mm in Suderburg. In Rosche, the sum of soil water demand (112 mm) and percolation water (34 mm) accounts for 146 mm. With a mean TPAW of 185 mm these sites are not capable of compensating for the demand of about 208 mm.

Hence, 95 to 98 % of the cropland area in both municipalities shows insufficient plant available water to sustain stable yields and is therefore not suited for SRC under (L_h) conditions (see Figure 7a). The remaining 2 to 5 % of cropland area reflects soils with capillary rise from the water table. ((L_h) variant – see Figure 7a).

For the (L_l) variant, average soil water demand in Suderburg only accounts for about 8 mm while it is balanced by an average TPAW of 108 mm. Hence, the sum of soil water demand and percolation loss by about 46 mm is fully balanced on all potential SRC sites (see Figure 7b) without using water due to capillary rise from the groundwater table.

In Rosche, due to lower precipitation, SRC soil water demand during the vegetation period is higher and accounts for 30 mm on average. As a result, about 64 mm have to be balanced by 70 % of TPAW to maintain stable yields. Despite an average TPAW of 114 mm, about 32 % of the cropland sites do not meet this criterion (see Figure 7b). However, about 68 % of the cropland area is suited for SRC under (L_l) conditions.

To sustain stable yields, the following rule of thumb could be formulated:

$$V_{ETveg} <= \Sigma(0.55\ P_{year} + 0.7*PAW).$$

If this equation is true, no groundwater is needed to supply SRC water demand. With annual precipitation of 610 to 770 mm and V_{ETveg} (L_h) ranging from 475 to 600 mm at least 200 to 260 mm of PAW is needed to balance the water demand without using groundwater and to maintain stable yields. For (L_l) conditions (V_{ETveg} = 400 to 450 mm) PAW only has to account for 80 to 100 mm.

Ground water recharge (GWR)

For cropland areas, the average proportion of GWR to annual precipitation is 38 % in Suderburg and 35 % in Rosche. In absolute numbers, GWR from cropland ranges between 52 and 360 mm a^{-1}. According to the classification by Marks et al. (1992), high GWR is predominant on cropland areas in Suderburg. In Rosche, only northeastern parts show high GWR while medium GWR is typical for the major cropland area (see Figure 8).

For (L_h), SRC groundwater recharge ranges from 42 to 95 mm a^{-1} which is equivalent to 7.5 to 13 % of annual precipitation. Thus, the average decline of GWR due to SRC use is about 74 %. In both municipalities more than 90 % of cropland area shows a decline in GWR which is higher than 70 % compared to current cropland use (see Figure 9).

The numbers given for GWR of current cropland use do not take into account that about 85 % of arable land in the district of Uelzen is irrigated (Fricke, 2006). On average, 76 mm of groundwater is used as irrigation water (Fricke, 2008). Considering a loss of 70 % due to evapotranspiration of irrigation water (according to Frede, 2006), a comparison between current land use and SRC of the (L_h) variant still reveals an average decrease by about 70 %. Thus, according to the classification of Marks et al. (1992), GWR of SRC has to be classified as "very low".

For (L_l), SRC groundwater recharge ranges from 122 to 233 mm a^{-1} (low to medium according to Marks et al., 1992). This is equivalent to 20 to 30 % of annual precipitation, and indicates an average decline by about 30 % compared to current cropland use. Taking into account the irrigation loss of current cropland use, the relative decline reduces to only 5 to 10 % on average.

Qualitative ecological impact assessment

The ecological impact analysis of SRC on groundwater recharge builds on the sensitivity of GWR versus its impact intensity caused by changes of GWR. The impairment of the groundwater recharge as an ecological function was assessed to be tolerable if the decline in GWR does not exceed 25 % and GWR of current land use is higher than 180 mm a^{-1}. For very high GWR numbers (> 320 mm a^{-1}) a tolerable decline could even rise to 50 % (see Figure 4). Due to the very high decline in GWR under (L_h) conditions,

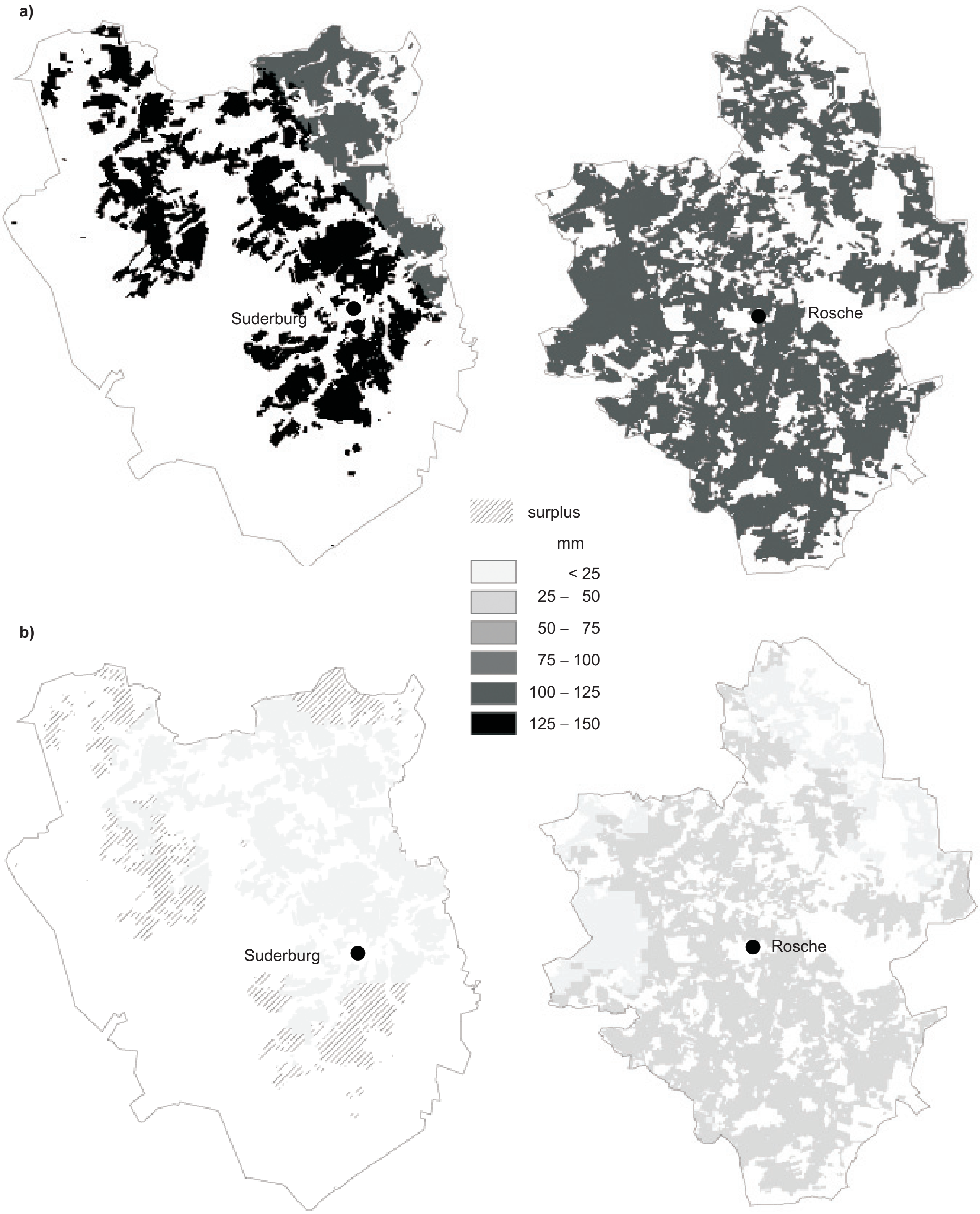


Figure 6:

Soil water demand (during the vegetation period) of potential SRC sites on cropland in the municipalities of Suderburg and Rosche. a) L_h variant b) L_l variant

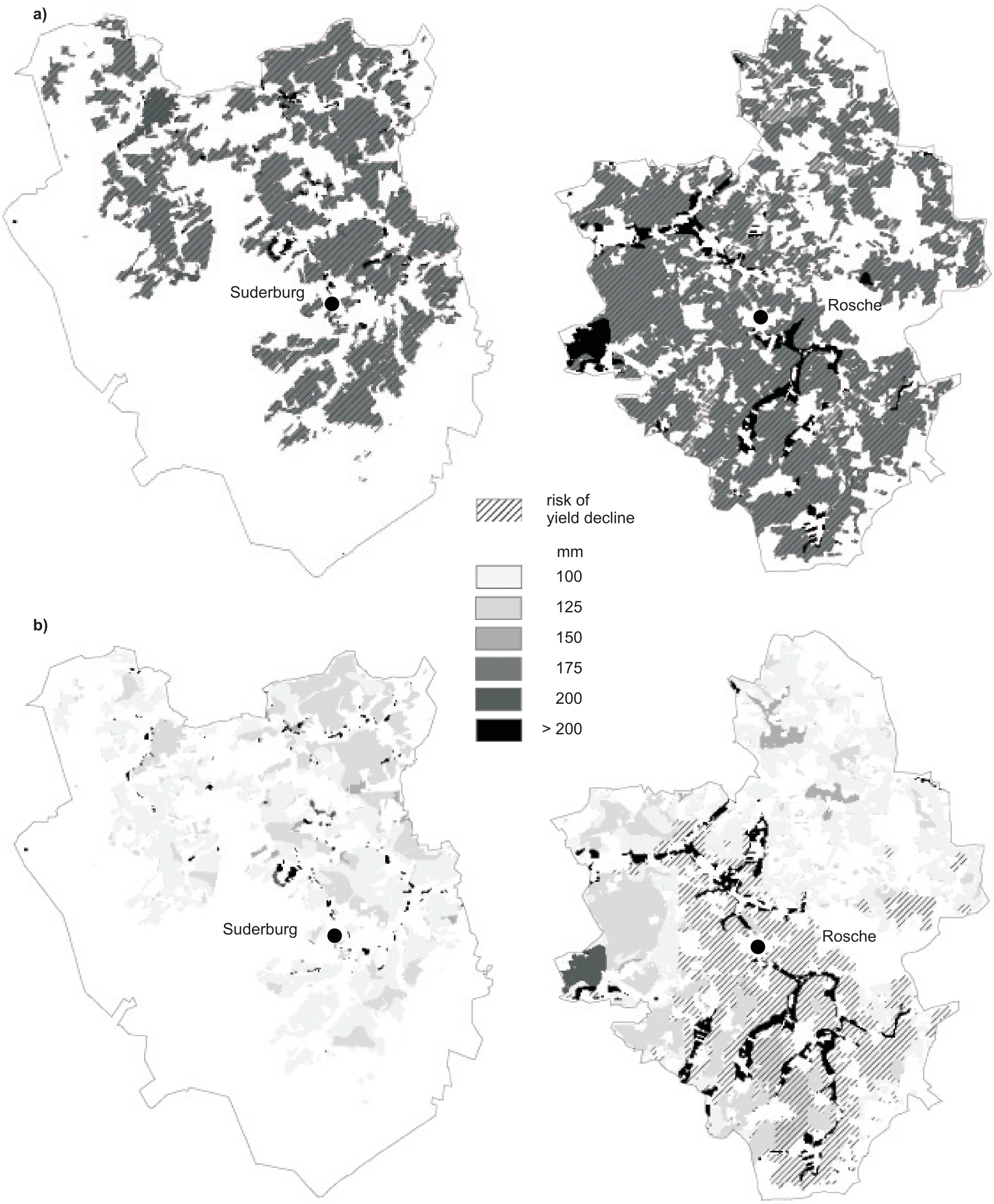


Figure 7:

Total plant available water (during the vegetation period) from cropland in the municipalities of Suderburg and Rosche. a) L_h variant b) L_l variant. The L_h variant was calculated with an effective rooting depth of 90 to 150 cm. The L_l variant refers to an effective rooting depth between 55 and 100 cm.

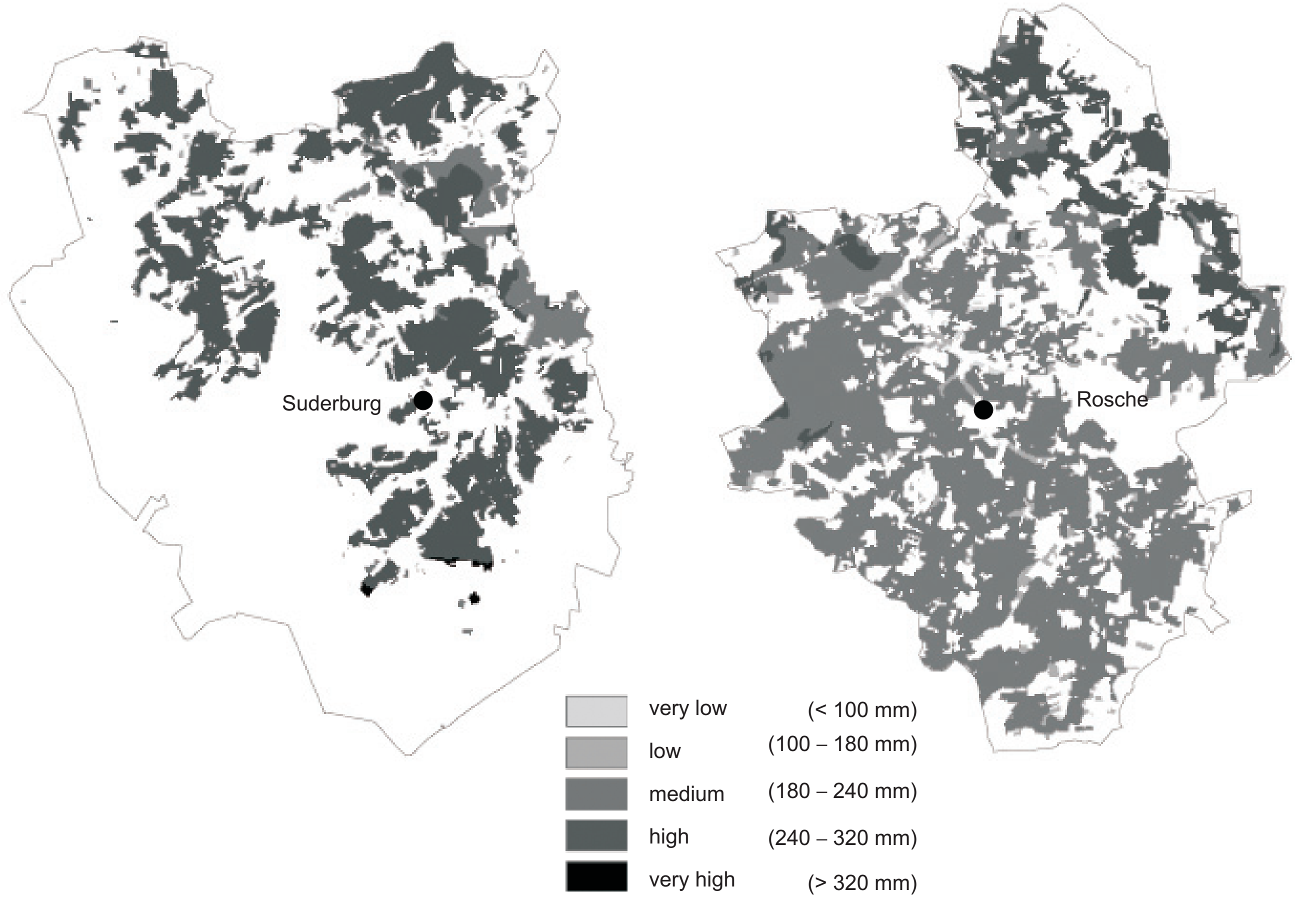


Figure 8:
Ground water recharge (current land use) from cropland areas in the municipalities of Suderburg and Rosche

this environmental goal could not be met in the municipalities of Suderburg and Rosche (Figure 10). On the contrary, about 90 % of the potential SRC sites were assessed to have a very high impact on ground water recharge. With respect to groundwater protection, SRC systems representing (L_h) conditions should not be established in the study area.

For (L_l) conditions, large areas (about 86 % in Suderburg and 92 % in Rosche, respectively) show a medium impact, i.e., the decline in GWR ranges from 25 to 40 % while GWR is classified as medium to high. These areas do not meet the goals of the qualitative assessment. Note however, that the average reduction by about 80 mm a^{-1} is comparable to the current loss of irrigation water for annual cropping. Approximately 9 to 14 % of the crop area in both municipalities does meet the assessment goal and should be given priority when implementing SRC.

Conclusions

There is an urgent need to assess potential impacts of SRC on landscape functions (e.g. for spatial planning). Despite sparse data and, thus, high uncertainty of interpretation, the approach presented in this study could be a starting point since its statistical relation of annual precipitation against actual evapotranspiration allows to assess SRC water use based on public available data. The two variants, (L_h) and (L_l) represent frontier areas of a fairly broad corridor of potential SRC water use. It remains an open question if a transition from (L_l) to (L_h) conditions during a SRC lifecycle is a plausible pathway.

Results for the two boundary conditions show that impacts on GWR could be completely different depending on site-specific conditions and the SRC variant chosen. In the study area, the combination of sandy soils with high rates of GWR but quite low amounts of PAW leads by itself to a potentially high impact of SRC on GWR. The results of an ecological impact assessment on GWR would be quite

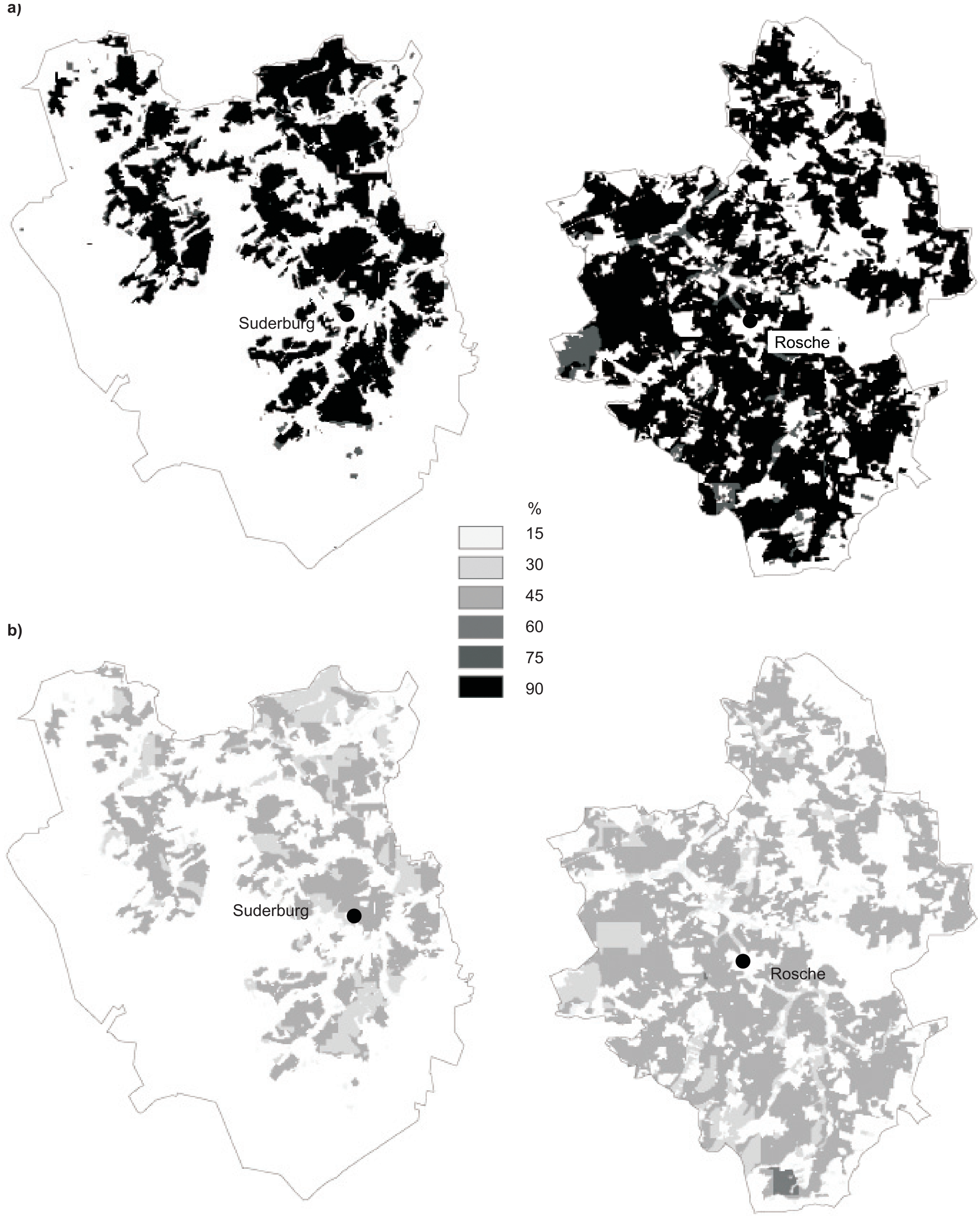


Figure 9:

Relative decline in ground water recharge from SRC compared to current land use on cropland areas in the municipalities of Suderburg and Rosche. a) L_h variant b) L_l variant

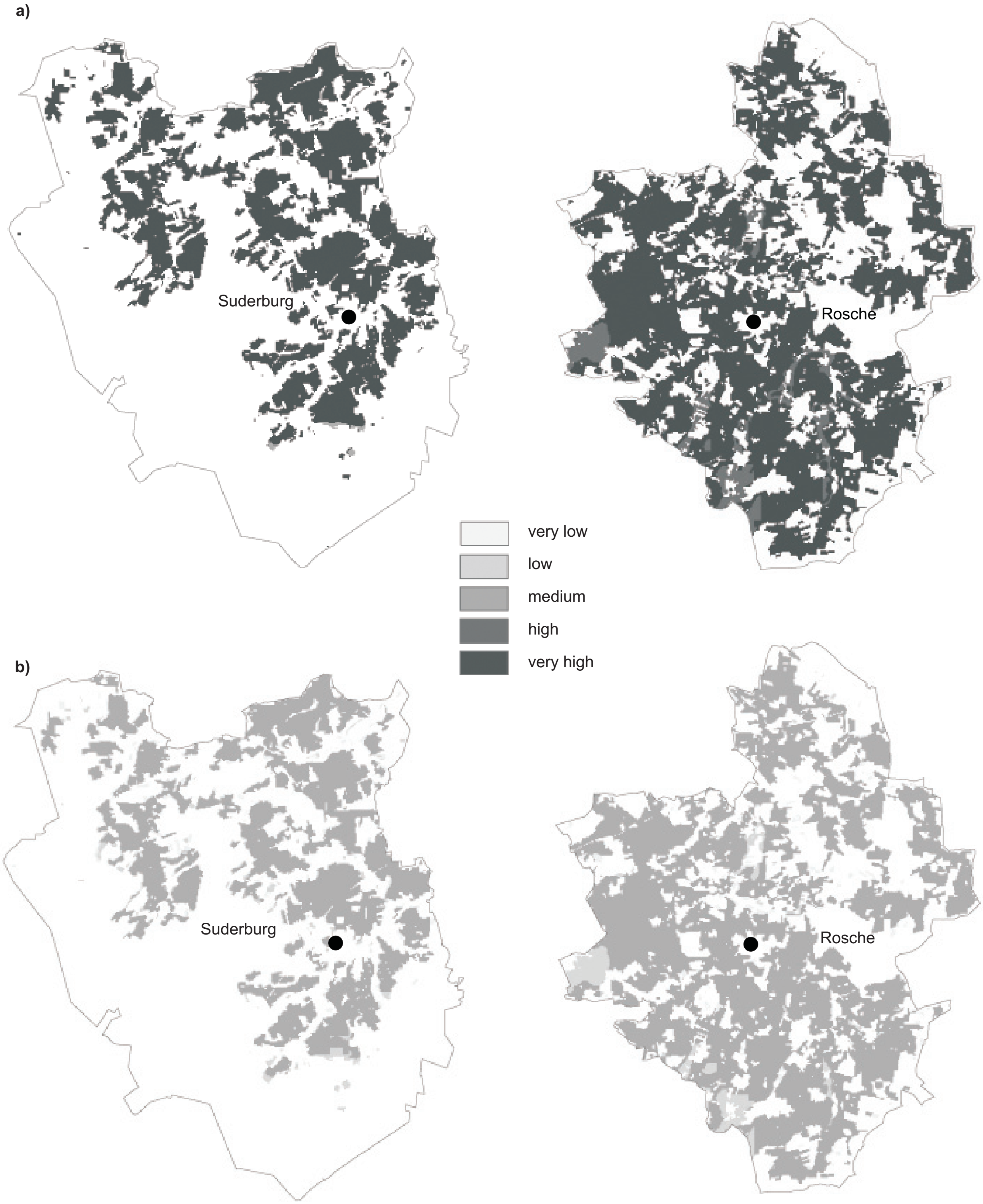


Figure 10:

SRC and its ecological impact on ground water recharge in the municipalities of Suderburg and Rosche. a) L_h variant b) L_l variant different when carried out on loamy soils with considerably lower rates of GWR but higher AWC.

As a consequence, an ecological evaluation of landscape functions has to consider the impact of site-specific conditions on environmental goals. The following conclusions can be drawn from the results of this paper:

- In general, existing data imply that SRC water use increases with longer rotation periods. Thus, these types of SRC (e.g. poplar SRC) should be restricted to sites with high water supply. In regions with a potential deficit in one of the water pools (precipitation, available soil water, and capillary rise from the water table) it is a risk-minimizing strategy to focus on SRC with rotation periods of 2 to 4 years.
- SRC representing (L_h) conditions should preferably be established on sites with high AWC (loamy and silty texture) to minimize a potential negative impact on groundwater recharge and to sustain stable yields
- For (L_l) conditions, sandy soils could be appropriate sites for SRC with a rotation period of 2 to 4 years.

Representing (L_l) conditions, SRC could be an interesting land use option due to its potential positive impact on soil erosion groundwater quality and landscape structure (e.g. Baum et al, 2009; Lamersorf et al., 2008). In regions with low summer precipitation, it could be more appropriate to establish SRC with species such as black locust. Thus, it would be important to collect data for additional species and for precipitation numbers that are at the lower end of this investigation. An improved database is needed to narrow down the potential corridor of water use and to derive rule-sets for a semi-quantitative to qualitative assessment of landscape functions.

The following items are judged to be most important:

- to record sites with different (physical) soil properties
- to consider different climatic types with a focus on precipitation patterns
- to take into account distinct rotation periods
- to distinguish between willow, poplar and certain clones
- to broaden the approach by other species, e.g. black locust
- to carry out long-term measurements
- to capture transpiration, soil evaporation and interception
- to identify effective rooting depth

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Socio-economics in SRC – a review on concepts and the need for transdisciplinary research

Jörg Köhn*

Abstract

The production of woodchips in short rotation coppices (SRC) presents a new market opportunity, which is strengthened by both the EU agricultural policy and the European emission trading system. This market will grow substantially in the next few decades. Woodchips can substitute for fossil fuel to generate heat and power as well as provide car fuels from biomass. The SRC market will interfere with both the food and feed markets. Farmers and other stakeholders in the woodchip market have to decide whether to go for the SRC option or not. Examples from Sweden and Germany show how different stakeholders along the production line including end-users and financial institutions value chances and risks in the SRC market. The assumption that SRC is an environmentally friendly and economically promising option that can contribute to climate change mitigation (for example Pieprzyk, 2009) is lacking scientific evidence and reliable data. An assessment on the total impact(s) to the environment and society is thus needed. In this paper research concepts are reviewed for a comprehensive analysis outlining a research agenda to evaluate SRC markets. One result of the review is that socio-economic research in SRC requires transdisciplinary work integrating the fields of ecology, technology, business studies, and economics with innovation, stakeholders´ involvement, and societal impacts on regional economies.

Keywords: short rotation coppices, carbon emission trading, socio-economic studies, environmental costs

Zusammenfassung

Sozio-ökonomische Studien zu Kurzumtriebsplantagen (KUP) – Ein Überblick über Konzepte und die Notwendigkeit transdisziplinärer Forschung

Die Produktion von Holzhackschnitzeln in Kurzumtriebsplantagen (KUP) ist ein relativ junger Markt. Die EU Agrarpolitik und das Europäische Emissionshandelssystem schieben diesen KUP (SRC) Markt an. Dieser Markt wird in den nächsten Dekaden stark anwachsen. Holzhackschnitzel erlauben es, fossile Energieträger zur Erzeugung von Strom und Wärme, aber auch zur Herstellung von Kraftstoffe für Automobile zu ersetzen. Der Markt für KUP greift in den Markt zur Erzeugung von Nahrungsmitteln und Futter ein. Landwirte und andere Stakeholder müssen entscheiden, ob sie in den KUP Markt investieren wollen oder nicht. Beispiele aus Schweden und Deutschland zeigen, wie unterschiedlich verschiedene Stakeholder ihre Chancen und Risiken im KUP Markt bewerten. Der Annahme, dass KUP eine umweltfreundliche und wirtschaftlich tragfähige Option sind, um den Klimawandel zu beeinflussen, fehlt der wissenschaftliche Beweis und eine Datenbasis, die eine verlässliche Abschätzung der Gesamteffekte auf die Umwelt und die Gesellschaft zulässt. In diesem Beitrag wurden Forschungskonzepte für diese Analyse geprüft und der Forschungsbedarf für die Bewertung von KUP Märkten dargestellt. Ein Ergebnis dieser Analyse ist, dass die sozioökonomische Forschung in KUP Märkten Transdisziplinarität erfordert, die sowohl die Forschungsfelder Ökologie, Technologie, Betriebswirtschaft, Volkswirtschaft, Innovationen, Stakeholderbeteiligung und soziale Effekte im regionalen, nationalen und globalen Maßstab integriert.

Schlüsselworte: Kurzumtriebsplantagen, Emissionshandel, sozio-ökonomische Studien, Umweltkosten

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1. Introduction

The consumption of energy as both heat and power is an important driver of society. The World Resources Institute used carbon emissions as an indicator of measuring the level of industrialization in national economies (Hammond, 1994). The increasing demand of energy has induced climate change due to the continuously increasing discharge of climate relevant gases into the atmosphere (Deutscher Bundestag, 1990). The mitigation of carbon dioxide emissions ranks high in the political agenda of the creators of climate protection policies today. The European Union has introduced an emission trading system that should allow cutting carbon emissions in an artificially created market (EU, 2003). Carbon dioxide emission has been selected the indicator to assess the emission status of national economies. Carbon emissions cannot be measured directly so a complex accounting system, along with national registers, deals with this problem in every EU country (for example DHESt, 2008). The basic idea behind the emission trading system is that pollution of the atmosphere with carbon will no longer be without cost or penalty. The emission trading system will encourage industry to invent technologies to reduce their emissions. If industry has to pay for air pollution with climate gases, the price for the permissions is part of production costs, and industry always attempts to cut those costs to a minimum (Dales, 1968).

Industry may choose from several options to reduce production costs related to carbon emissions:

1. Increasing the technological efficiency of energy conversion in existing power stations.
2. Replacing fossil fuels with fuel from renewable sources, for example co-incineration technology. The substitution of fossil fuels in existing plants has some technological limits due to the lower heating value of renewable fuels.
3. Expanding the market share of renewable energy, for example in new installations.
4. Cutting energy demand in industry and households, for example with new appliances or a tariff system that reacts to supply and demand disproportions (Factor 10 Institute, 2008).

Options 2 and 3 are related to the SRC topic. Whereas option 2 calls for co-combustion technologies, option 3 is part of a market that also includes solar and wind energy. It is widely accepted by economists that the emission trading system will increase the total costs for energy produced from coal, oil or natural gas and thus improve the competitiveness of energy produced from renewable sources (Helynen, 2007).

Blue-green bacteria, algae and green plants are able to convert solar energy into biomass using photosynthesis. Carbon dioxide and other waste gases, for example nitrogen, may serve as inputs to the process of photosynthesis. The conversion of waste gases from industry into nutrients may cause unwanted effects, for example algal blooms in coastal ecosystems (Gren et al., 1996). Biological systems are able to collect and convert carbon dioxide using sunlight (along with other waste gases or even hazardous substances, Baum et al., 2009). Carbon is stored in biomass and is available for another cycle of energy production (and carbon dioxide release). Unfortunately this cycle is not forming a closed loop. Each and any conversion process is characterised by a "loss" of energy (for example heat that can not be used) or creation of unusable matter (for example ash). As the technological loop cannot be closed in full the economic cycle cannot either (Georgescu-Roegen, 1971).

SRC plants, mainly willow or poplar, use the biological mechanism of photosynthesis and reintroduce carbon into the economic cycle. The main advantage of using trees to produce biomass is that energy can be stored in the fibre structure of the wood. Energy is stored during growth and even after harvest as long as the fibre structure is intact. The technological processes of incineration or gasification extract the stored energy for heat and power (Dimitriou et al., 2009) or synthetic fuel production (ECJRC et al., 2007; DENA, 2006).

The main product of SRC plantings is chopped wood (woodchips). The interest in forest industry is the production of logs. Logs can be used in almost all sectors of the timber industry whereas the market for woodchips is more restricted. The market value of woodchips is low compared to logs. Woodchips can be made of forest by-products such as roots, branches and treetops, or of waste wood, of timber from landscape management or of SRC crops. It is assumed that the market will run short for the lack of resources, for example in the Baltic Sea region in the next years (VTT, 2007). The production of woodchips in SRC plantings may help to overcome shortness in the timber market with forest products.

The history of planting SRC is closely linked to the recultivation of open mining fields (Hüttl, 2001). One may argue on the one hand that SRC may enrich landscapes by adding mosaic structures in the meaning of the Natura 2000 concepts by forming keystone habitats (European Commission, 1992). On the other hand it is argued that SRC can have negative impacts on environmental goods and services for example the ground water table because of the demand of water for SRC. Both arguments need to be evaluated. There are few databases on this issue (see for example Dimitriou et al., 2009).

This paper will review concepts and conceptualise questions that need to be answered in order to allow a socio-economic evaluation of the SRC market's potential. The socio-economics involved in the SRC market face many challenges:

1. The SRC market is still in an early state of development.
2. The allocation of arable land to SRC plantings and the institutionalization of SRC woodchip production and markets are evolving.
3. The assessment of environmental impacts using cost and benefit analysis (CBA) needs data from environmental impact assessments.
4. The application and introduction of new technologies continuously make the market subject to change.
5. The regional impact on the society with SRC covering large areas and changing landscapes is widely unknown.

2. Material and methods

This paper uses literature, national or European Union regulations, project reports and interviews with stakeholders for the analysis. Most of the project reports were funded by the EU and addressed the local, the regional, the national, or the EU levels. All these studies focussed on the market potential of woodchips produced in SRC plantings. Data on environmental and societal impacts that the production of woodchips in SRC may induce directly or indirectly (Leontieff, 1986) are rare or even missing. There are some preliminary studies only assuming that SRC plantings will increase the demand on labour in the rural areas (Tack et al., 2002; Baltic Sea Agro-Industrial Network, 2004). VTT (2007) assumes that the SRC market will be pushed by the European emission trading system since forestry cannot deliver the expected increasing demand on timber and invite new stakeholders in this emerging market.

This paper is part of an on-going research project using study sites for ecological analyses and impact assessments in Sweden and Germany (RATING-SRC, 2009). Along with the ecological research the socio-economic studies are conducted in the same regions.

This paper represents research questions the socio-economic part of the project intends to answer. The paper is divided into sub-sections. Each sub-section describes the research question of the specific sub-section and the tools used to analyse and evaluate data and individual or social aspects a SRC market will have on the environment and the social system.

This paper bases on literature and interviews mainly.

3. An emerging market

This sub-section tries to analyse

– Why are woodchips produced in SRC seen as an economic and political option to support a market that is historically based on forests only?
– Why does this market attract new stakeholders?

The history of the use of timber as a fuel dates back almost 790,000 years (Goren-Inbar et al., 2004). Weeber (1990, 11-12) analysed historical documents from the ancient world and could show that devastation of landscapes and changes in the regional climate due to clear cutting of forests had already occurred BC. Fischer (1996, 22, 76, 124) observed that the availability of timber (firewood) had a great impact on human history in Europe. He analysed the evolution of prices of agricultural products including firewood over a period of 800 years. According to his studies firewood prices indicated the emergence of economic and social crises in Europe much earlier than the increase of prices for food. Interpreting these results one may conclude that the allocation of land to produce timber or food was already an economic problem in the Middle Ages.

Fossil fuels have replaced timber as the main source of fuel only since the 19th century. This is due to (1) the rapidly increasing demand for electricity, (2) the supply of fuels for transportation, and (3) the hunger of the chemical industry for resources such as coal, oil, and natural gas. The extensive use of fossil fuels caused a steadily increasing release of carbon into the atmosphere. The amount of carbon a society emits served as an indicator of the level of a nations´ industrialization (Hammond, 1994, various tables).

IPCC addressed carbon dioxide as a main driver for climate change (IPCC, 1995). IPCC (1996) assessed the economic and social dimensions of climate change, developed response strategies, evaluated costs for response options, and demanded integrated research to assist the development of policy instruments for combating climate change. IPCC (1995, 13) demands "stabilizing carbon dioxide concentrations in the range of 350 ppmv (near current levels) to 750 ppmv." IPCC (1996, 6) calls for cost-effective measures to reduce carbon emissions in line with applicable international agreements. Economic measures mentioned in the IPCC report include "encouraging forms of international cooperation to limit greenhouse gas emissions, such as implementing coordinated carbon/energy taxes, activities implemented jointly, and tradable quotas." The emission trading system introduced in Europe in 2003 (EU, 2003) is based on the tradable quota principle recommended by IPCC in 1996.

The member states in the European Union had to introduce national registers for carbon emissions because of the 2003 directive. The total sum of those registered emissions formed the initial European market for carbon emissions. In a grandfathering process the shares in this market were distributed to the involved industries for free. In next step the policy will cut the market volume. This procedure will be repeated by policy until an environmental sustainable level is achieved (Goodland, 1995). The reduction of the market volume will increase prices for the individual pollution rights traded in this market. National agencies record and supervise the national emission trades (for example in Germany, DEHSt, 2008). The increase in prices will steadily make the use of fossil fuels more expensive, thus will improve the competitiveness of renewable fuels. Stakeholders in the biofuel market expect an increasing demand on biomass and particularly SRC woodchips (VTT, 2007; Pieprzyk, 2009). One result of the introduction of the European emission trading system is that new stakeholder groups feel invited into this market. We learned from interviews and direct request for consultancy that companies which usually trade oil (for example GEE, 2008) or provide financial services (for example WestLB, 2007) are investing in SRC especially in Eastern Europe.

The idea of SRC plantings is based on the experience of recultivation processes. The mining of coal released large, ecologically devastated areas to society. One option for recultivation is planting trees able to cope with those harsh environments (Hüttl, 2001). Special breeds of poplar and willow have been developed and adapted to those conditions. Both species grow fast, use sunlight and carbon dioxide as inputs, enrich carbon in the soil and store energy in biomass. The crops of SRC plantings are chipped and these woodchips return to the economic cycle biomass.

EU agricultural policy (BMVEL, 2006) supports the use of arable land to plant SRC in order to substitute as much fossil fuel as possible with renewable ones (Dimitriou et al., 2009).

4. The allocation of arable land, opportunity costs and the institutionalization of the market

The planting of SRC requires assigning land for this specific use. One option EU policy supports is using arable agricultural land. This land can be used either for the production of human or animal food or for SRC. SRC compete with those uses. Since agricultural land is a limited resource costs will rise. The first part of this section deals with those aspects.

The second part of this section is related to another economic question, that of market organization or institutionalization. The economic issue behind this question is transaction costs.

Both aspects are related to each other and to decision making processes in SRC and will be explored in detail during this research project (RATING-SRC, 2009).

The assignment of arable land to SRC plantings is clearly an economic issue. The supply of arable land is short and the demand on arable land for food and feed production will compete with SRC plantings. Any allocation of scarce resources always relates to opportunity costs.

The concept of opportunity costs is an artificial construct that allows comparing expected returns on investment for alternative uses of the same resource, i.e. soil in this example. The calculation of future earnings may include risk or uncertainty factors (Stiglitz, 1997, 41). It is possible to calculate risks but not uncertainties. Risk is an expectation on a probability of outcomes in the analysed process. Uncertainty, to the contrary, also includes unforeseeable impacts and thus cannot be quantified. For example, fourth measure impacts are seen as an uncertainty (Raiffa, 1968).

We have learned from interviews in Germany that farmers compare outcomes from SRC with those from annual crops, for example wheat (Tack et al., 2001). Experiences in Mecklenburg – Western Pomerania show the SRC market does develop as expected. For example, only three farmers in an area of about 400.000 ha grow SRC on approximately 50 ha of arable land. That result was picked from an official database of EU-subsidised farmers for planting SRC on arable land this year (AFL Bützow, 2009, interview August 10, 2009). One of the three farmers grows 20 different clones of willow on a 30 hectare plot. Choren Industries has contracted him to grow willows in the SRC mode (Bauernzeitung, 2009). Farmer did not succeed since the Choren BTL plant was not built, this farmer could not sell the produced woodchips. Thus the farmer´s investment did not provide the expected returns.

EU subsidies have compensated the losses to a certain extent. EU pays, for example, about 300 € per hectare per year for the use of arable land. In addition, EU also pays a premium of about 56 € per hectare for growing energy crops. This premium however will be paid for the last time in 2009 (AFL Bützow, 2009, interview August 10, 2009).

The farmer can reduce opportunity costs using marginal soils for SRC plantings, growing cereals or other annual crops in better quality soil. For example, the soil quality at the particular willow spot reported here was too low to grow wheat. The contract with Choren Industries to grow willow allows the farmer keeping the plot within the register of arable land. So the EU payments for arable land compensated the losses in part in this example (Schaack, 2008, interview in 2008). If spent costs are not covered by earnings the real losses turn in to sunk costs (Stiglitz, 1997, 42f).

In an earlier project entrepreneurs (Michel-Kim et al., 1998) initiated a ten hectare SRC planting with a farmer growing willow, poplar and alder to produce seedlings. No seedlings and no woodchips could be placed in the market. Consequently, the farmer cut the plantings clear and converted the plot back into a wheat field. These examples are well known to the farmers in the region and made farmers cautious with regard to growing SRC in North-East Germany.

The German example shows that the opportunity costs early birds face in new markets (Slater et al., 2006, 32) may form a barrier for investments since the risk is too high. The EU subsidies buffer the risk of total losses. Since EU pays the subsidies on a per hectare calculation the resistance of farmers against the new crop still is relatively high.

If one compares the situation in Sweden with the one in Germany the picture is almost completely different. The arable land covered with SRC plantings is substantially larger (Dimitriou et al., 2009). One reason for this is that the market is institutionalised and mediated by a farmers´ organization in Sweden. This organization mediates the marketing and offers (full) services to farmers and landowners along the whole SRC production chain (Agrobränsle, 2009). The growing of SRC plantings is based on contracts with end-users to avoid sunk costs.

Furthermore, Swedish law permits the use of ash and sludge mixtures as fertilizer, and it permits watering the SRC plantings with treated wastewater. The farmers may avoid additional costs for irrigation and fertilization of the SRC fields because of this regulation (ENA, 2009). The use of ash from CHP plants fired with woodchips from non-polluted sources is not permitted in Germany whereas it is common practice in Sweden. Ash from the combustion of non-polluted woodchips is seen as a waste according to German law (AVV, 2007, 100103).

The institutionalization of the Swedish market includes contracting of farmland, planting, weeding, harvesting SRC and trading SRC woodchips. Farmers are used to the institutionalised and mediated market since it is somewhat to markets with annual crops. Opportunity and transaction costs are reduced in an institutionalised market (North, 1984).

The institutionalization of the market has, according to interviews with stakeholders, the following advantages:

– Costs for purchasing adapted technology for planting, weeding and harvesting SRC are avoided due to the "full service offer" of the farmers´ organization.
– The market for SRC woodchips can grow fast and deliver large quantities of woodchips to end-users.
– The end-user industry can trust in a market serving their demand.
– The end-user industry has to deal with large suppliers only. They can be sure that the suppliers will deliver a product on contract at a negotiated (accountable) price and in large quantities meeting the demand for example of a CHP plant.
– A relatively fast substitution of fossil fuels with SRC woodchips is technologically possible and economically feasible.
– The risks and costs of planting SRC to the individual farmer can be reduced.
– The opportunity and transaction costs are relatively low.
– The main players in the SRC market can push the market lobbying for the products using climate protection arguments.
– The market attains public awareness.

The disadvantages are:

– The market lacks competition. A few suppliers may dominate an individual farmer's interest and may squeeze farmers´ income from SRC planting.
– Few suppliers of seedlings, planting, weeding or harvesting technology dominate the market. The innovation rate may slow down.
– The market "invites" new players, for example financial institutions or other investors. The formerly farmers´ business may shift into an investment business.

5. The assessment of environmental impacts, costs and benefits

Ecological research needs to be translated into economic terms when a policy intends to interfere in markets with cost-effective measures as IPCC (1996) demands. This section names and reviews the underlying basic concepts of environmental and ecological economics that should be applied one way or the other in this research project.

It is necessary to quantify and to evaluate environmental impacts related to SRC plantings in order to increase acceptance (Dimitriou et al., 2009). The ecological data need to be translated into an economic language using prices and quantities. As long as those data are missing an evaluation using the methodological frameworks of environmental (see for example Bromley, 1995) or ecological economics (see for example Costanza, 1991) is impossible. This paper reviews the concepts of environmental and ecological economics and proofs their applicability for an economic evaluation that will fit into a comprehensive socio-economic evaluation of SRC.

Every human activity on Earth impacts the environment. Pigou (1920) highlighted the fact that human wealth depends on a healthy environment in economic literature for the first time. The entrepreneur who cares for the environment with his business may face higher costs than competitors in the same market who do not. Pigou calls for a mechanism that ensures that the entrepreneur who cares for the environment can survive in the market. In principle, there are two options solving this problem: First, making the polluter compensate the cleaner business, second, regulating the total load of pollution by political intervention in the market.

It took almost fifty years until economic theory proposed instruments that can cope with the environmental issues. The book of Rachel Carson (Silent Spring, 1962) served as a wake-up call for an environmental policy. Economists proposed three main strategies, the concept with standards and prices (Baumol et al., 1971), the property rights approach (Dales, 1968) and the environmental liability tool (IPCC, 1996; EU, 2004).

A policy sets taxes or subsidies (prices) and standards permitting a limited pollution using safe minimum standards or critical loads (quantities, Baumol et al., 1971). If an industry cannot cope with those limits, it has to pay a fine (tax). Private or public environmental management systems, for example a wastewater treatment plant, have to cope with the additional pollution. They get paid from the polluter but may also be compensated by using the fine paid by the polluter to improve the treatment system. The main problem, however, is setting the "right" standards and prices.

The application of Dales´ proposal will indirectly push the SRC business. Dales (1968) proposed to regulate environmental markets by issuing pollution rights. The basic concept to protect the common good, "clean air", (Hardin, 1978, 1244) is to assign property rights for a limited amount of pollutants emitted, for example, to the air. Policy regulates or reduces the total amount of pollution until a sustainable level of pollution can be approved. This environmental management should induce at least two effects: First, industry reacts in developing more efficient technologies to avoid payments due to the emission trading system and second, the trade in the emission market will generate a pricing system internally within the rules of a market economy. The carbon emission trading scheme applies Dales´ approach in Europe (EU, 2003). VTT (2007) sees in the emission trading system a good chance for the bioenergy business to grow.

Environmental liability has been regulated in an EU directive of 2004. The economic concept on environmental liability is based on insurance concepts.

All three concepts are to some extent related to SRC planting:

1. Setting environmental standards affected technological changes in combustion processes. The co-incineration of timber in coal-fired power stations was introduced to reduce sulphur emissions due to the fact that timber does not emit sulphur. Co-incineration is seen as a technology that allows substituting fossil fuels with timber (woodchips) within certain technological limits today.
2. Setting climate protection targets using carbon dioxide as the only indicator for climate gas emission. The emission trading system increases the demand on renewable fuels since they are accounted "neutral" to carbon emissions. The emission trading system will also push the development and application of new technologies that convert and use energy more efficiently.
3. Environmental liability affects the overall planning process for any process generating heat and power. The costs related to environmental liability cases are accounted in the internal cost schemes of the enterprises.

All three economic concepts make environmental costs and benefits part of the business accounting scheme. Life cycle analysis (LCA) may report on all impacts that a certain product or service generates during its whole life (cradle to grave cycle). Various concepts have been developed to assess environmental costs and benefits of products (for example Rees et al., 1994; Bishop et al., 1995; Bockstael, 1995; Freeman III, 1995; Tietenberg, 1995; Pulm, 2001).

This project will analyse environmental impacts of SRC plantings along the production chain, trading and timber industry, the results on environmental impact analyses (other papers) and market data on woodchip handling and industry. The socio-economic analysis tries to translate those ecological and technological data into economic terms. The main focus is on an economised LCA methodology along with concepts of environmental valuation of costs and benefits (Bishop et al., 1995).

It can be expected that the Dales´ concept and its application in the European emission trading system will have the strongest impact on the development of the SRC market. However, the results of the analysis and the economic evaluation of the impacts will show "how green energy production with SRC really is".

6. The application and introduction of new technologies

Technological development is a driver in markets since it influences substitution of products, production modes and the main market indicator price. One can expect that technological change and changes in the management scheme will influence the SRC market. Böhlcke et al. (2006) report that the annual growth rate may increase by a factor of three or four if the rotation period (management scheme) is changed from three to six years.

In this project we will distinguish two main clusters of technological aspects related to SRC:

1. New or adapted technology to optimise SRC woodchip production.
2. New technologies end-users will apply.

The breeding of new clones:
- Growing faster and allowing larger yields and returns on investments.
- Growing in environments that did not allow any crop to grow. The new clones may "re-cultivate" harsh environments.
- Improving changes to / changing environments – less rain than before, salty soils etc.
- Resistance to pests or long droughts.

The planting of seedlings:
- Making planting less costly.
- Lighter machinery decreases pressure on soil.

The protection of SRC plantings against pests and weeds:
- Applying less harmful pesticides to the environment.
- Supporting soil symbiosis factors.
- Making clones resistant to various kinds of pests.

The stimulation of plant growth:
- Using various irrigation technologies to water SRC fields depending on the regional climate conditions or sources of water supply.
- Applying the "right" mixture of fertilisers.
- Avoiding soil erosion.

The adaptation of harvesting technology:
- Allowing cutting on market demand or extending rotation periods, if necessary, for environmental or marketing reasons.
- If market demand drops down SRC should be allowed to collect carbon and sun for another year. The diameters of the stems will increase at this time, therefore the harvesting technology has to be adapted or the harvesting needs to be done by hand.

The drying and storing technologies:
- SRC are harvested in autumn or winter. For storage the woodchips need to be dried to an extent less than 15 per cent moisture content.

Adapted technologies to reduce environmental impacts:
- Avoiding soil pressure.
- Supporting groundwater regeneration.
- Increasing bio-diversity.

Box 1:

Examples for technological change that may influence economy of SRC plantations

The first cluster of technologies deals with the development of machinery and innovations in management schemes along the production chain of SRC, whereas the second cluster focuses on the end-user industry, for example a shift from using woodchips in a CHP plant to the production of synthetic fuels.

Many of those technologies may be adapted from agricultural or forest technology and experience. All the processes invite stakeholders from almost all branches of research and business, for example scientists in basic and applied research, financial institutions, land owners, owners of workshops and farm industry.

The second cluster includes stakeholders that are interested in using woodchips as inputs to industries. Woodchips are used, for example, to produce fibreboards, compound products, pulp and paper, or energy. The various industries will compete for woodchips from forest or agricultural production.

This research project deals mainly with the energy market. One may divide the energy market into three submarkets that may increase the demand on woodchips:

1 The main market for woodchips from SRC plantings is the energy production in CHP plants (Dimitriou et al., 2009). CHP plants use woodchips as a co-fuel together with bark and sawdust from timber industry. Small and medium sized CHP plants produce power for the grid and heat for a district heating system. They operate seasonally or during the whole year. Pricing in the market for power effects the operational mode of the CHP plants. In Sweden the main focus of CHP industry is on heat production for district heating (ENA, 2009) whereas the focus in Germany is an all year long operation initiated by feed-in tariffs for electricity (BMU, 2009).
2. Large industrial power stations co-fire woodchips with fossil fuels to comply with (mainly sulphur) emission standards or to fulfil critical load regulations or to reduce obligations from carbon emission trading. The CHP as well as the coal fired large power stations use condensing power technology to heat water and convert steam into power. Heat is a by-product of this process. Some power companies operating large coal fired power stations are also involved in the coal mining business, for example Vattenfall in the Niederlausitz region. They grow SRC re-cultivating soils after open mining (Hüttl, 2001).
3. A new technological option inducing an important increase in demand for timber is the biomass-to-liquid (BTL) technology. The conversion demands 4 kg of woodchips to produce 1l of synthetic fuel. The energy

content of 1 kg of timber is almost half the energy content of 1l of synthetic fuel, i.e. the energy conversion rate is about 50 per cent (Köhn et al., 2009). Various pre-market studies see BTL technology as an economically feasible option to produce motor vehicle fuels (EJRC et al., 2007; DENA, 2006; BMU et al., 2007).

An analysis of the Agentur für erneuerbare Energien shows that planting SRC is the better option to produce biomass than growing annual energy crops (Pieprzyk, 2009).

7. Impacts of SRC on regional society

One aspect of SRC plantations is to produce as many goods locally or regionally as possible. The production of local energy – heat and power – in CHP plants or synthetic fuels may serve as such an option. If SRC are an environmentally sound and economically feasible option it may contribute to regional wealth. This section in the research project will analyse and evaluate those aspects.

European Joint Research Center (EJRC et al., 2007) analysed two scenarios for the BTL market. They compared global versus regional markets for biomass. Biomass contains less energy per ton than coal or oil, therefore transportation costs for biomass are substantially higher than for coal or oil. The consortium proposes to use biomass on a regional scale (EJRC et al., 2007). CHP plants connected to district heating systems fuelled with woodchips from SRC plantings are in line with those proposals.

Large BTL plants need approximately 1 million tons of woodchips to produce about 200.000 tons of synthetic fuels. Choren Industries (2009) argues that those plants are the smallest feasible size based on technological and economic models. The production of 1 million tons of woodchips will cover an area of approximately 150.000 to 200.000 hectares. This calculation is based on the documentation of yields in poplar or willow SRC plantings (Boelcke, 2006; Gienapp, 2006; Agrobränsle, 2009). If 20 per cent of a region of one million hectares is planted with SRC only one BTL plant can be supported by fuel. Köhn et al (2009) show that small BTL plants are feasible both from the technological and business concept.

The market for woodchips is not as transparent as required for a complex economic analysis at present. It is impossible to apply Leontieff´s input-output tables (1986) using the databases available today. Leontieff´s methodology, however, allows calculating direct and indirect effects a business causes on the regional or national economy.

The research project aims to generate a database to support such calculations. We assume, however, if SRC plantings will substitute for the production of annual crops on arable land only the overall impact on the agricultural and the agro-industrial market may tend to zero. This assumption bases on the idea that the earnings from annual crops and SRC woodchips in the long run will not differ substantially from the costs for labour, caring for the crop and harvesting.

However, the impact on the national economy will differ since the emission trading system as well as the generation of products from biomass instead of coal or oil will cause economic and environmental effects that need to be calculated in input-output tables.

The following impacts on the global economic and the global societal systems can be expected (examples only):

- In the fossil fuel and energy business. Climate change demands concerted action to mitigate carbon emissions. The emission trading system earmarks fossil fuels with an extra price making it more expensive in relation to renewable energy sources. This makes renewable energy more competitive in relation to fossil fuels. (EU) Policy intends to decrease the market share of fossil fuels. The shrinking market will, of course, effect the global market, for example, in total sales of fossil fuels, in need for global transportation, in substitution of technology (i.e. coal fired power stations), and in installation of better technologies to generate power and store energy. The impacts are expected to be seen in the annual gross domestic product reports on a national scale.
- In the forest and agricultural business. The "value" of forests and forest products is assumed to be a critical indicator. In addition, the creation of mosaic structures in landscapes with SRC may support coping with climate change, for example protecting or regenerating of groundwater in Central European regions like Brandenburg (INKA BB, 2009). The share of land planted with SRC is expected to grow substantially (Pieprzyk, 2009).
- In greening desert strategies. Several stakeholders (sciences and business, interviews 2007 – 2009) intend to grow jatropha trees using SRC experience to avoid further desertification in Africa. Another approach is to green deserts using sewer water to water those plantings (IfaS, 2009, interview). There is hope that such plantings will create new areas for the agricultural business, agro-forestry, and induce social wealth.
- In "terra preta" strategies. SRC may serve as an intermediate stage of cultivation enriching carbon in the soil and improving soil productivity in the long-term. Those attempts are related to the "terra preta strategy" (Glaser et al., 2001; Heck, 2009) and to the demand of EU agricultural policy. EU demands returning carbon to arable land in a three year period depending on the crops planted (BMVEL, 2006).

– In people's awareness to climate change and in people's knowledge and acceptance of sustainability issues (Hardin, 1968).

8. Summary

The planting of SRC in order to produce woodchips is an emerging market. The reform of the EU agricultural policy and the introduction of the European emission trading system are seen as instruments to push the SRC woodchip market. It is expected that the market will substantially grow in the next few decades. Woodchips, as one source of renewable energy, may substitute for fossil fuels to generate heat, and power as well as motor vehicle fuels from biomass (EJRC et al., 2007).

The SRC market will interfere in the food and feed market because these markets compete for the same resource – arable land. Any competition on a scarce resource demands rethinking the modes of production and the cost benefit schemes of the stakeholders involved in those markets. Stakeholders have to decide whether to go for the new market or not. Examples from Sweden and Germany show how different stakeholders take chances and risks in an evolving market. Whereas the market has been institutionalised and mediated in Sweden, farmers in Germany experienced some pitfalls.

Biomass produced in SRC is assumed to be the better option to grow biomass for energy purposes compared with growing annual crops, for example corn or cereal (Pieprzyk, 2009). This assumption, however, lacks scientific evidence. The environmental impacts SRC plantings may have on the environment need to be analysed and evaluated in a regional context. Those data may allow for an economic evaluation of costs and benefits in an environmental accounting system or in Life Cycle Analyses (LCA). LCAs, however, also have to take into account technologies that use and convert the energy stored in the biomass, which is produced in SRC. For this reason LCA compares concepts and technologies of businesses using woodchips from SRC, as well. The overall societal performance depends on the efficient use of resources and can be calculated on a regional, national or global scale using input-output tables (Leontieff, 1986).

In conclusion, socio-economics in SRC require transdisciplinary research integrating the fields of ecology, technology, business studies, and market economy with innovation, stakeholders´ involvement and societal impacts (Funtowicz et al., 1993).

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Improved national calculation procedures to assess energy requirements, nitrogen and VS excretions of dairy cows in the German emission model GAS-EM

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Summary

The calculation module for the assessment of feed intake and excretion rates of dairy cows in the German agricultural emission model GAS-EM is described in detail. The module includes the description of methane emissions from enteric fermentation as well as the assessment of volatile solids and (renal and faecal) nitrogen excretions responsible for carbon and nitrogen species emissions from manure management. Input parameters are milk yield and composition, weight and weight gain as well as feed properties.

The model is based on the derivation of energy requirements and the limitation on dry matter intake. The results agree well with those obtained from regression models and respective experiments.

The model is able to reflect national and regional peculiarities in dairy cow husbandry. It is an adequate tool for the establishment of emission inventories and for the construction of scenarios for policy advice.

Keywords: Dairy cows, model, energy balance, mass balance, methane, nitrogen, emission, inventory

Zusammenfassung

Verbesserte Rechenverfahren zur Bestimmung von Energie-Bedarf, Stickstoff- und VS-Ausscheidungen von Milchkühen im deutschen Emissionsmodell GAS-EM

Das Rechenmodul zur Bestimmung der Futteraufnahme und der Ausscheidungsraten von Milchkühen im deutschen landwirtschaftlichen Emissionsmodell GAS-EM wird ausführlich beschrieben. Es ist in der Lage, die Methan-Emissionen aus der Verdauung sowie die zur Berechnung der Emissionen von Methan und den Stickstoff-Verbindungen benötigten Ausscheidungen von Kohlenstoff („volatile solids“) und Stickstoff (renal und fäkal) aus dem Wirtschaftsdünger-Management zu berechnen. Als Eingangsgrößen werden Milchmenge und Milchzusammensetzung, Gewicht und Gewichtszunahme sowie Futtereigenschaften benötigt.

Das Modell basiert auf der Ableitung des Energiebedarfs und der Limitierung der Trockenmasseaufnahme. Die Ergebnisse stimmen mit denen aus Regressionsmodellen und den ihnen zugrunde liegenden Messungen überein.

Das Modell erlaubt eine Berücksichtigung der nationalen und regionalen Besonderheiten in der Milchkuh-Haltung. Es ist daher als Instrument zur Berechnung von Emissionsinventaren sowie zur Erstellung von Szenarien zur Politikberatung geeignet.

Schlüsselwörter: Milchkühe, Modell, Energiehaushalt, Stoffhaushalt, Methan, Stickstoff, Emission, Inventar

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1 Introduction

In Northwest and Central Europe, emissions of gaseous pollutants from agriculture are dominated by emissions from animal husbandry. The most important key source is dairy cattle, so any measures to reduce emissions have to consider dairy cattle as a priority. Emission reduction strategies which consider both the necessity of agricultural production and its particular features have to rely on models that describe the emitting processes adequately. With respect to dairy cattle, the processes that emit methane (CH_4) from enteric fermentation as well as CH_4, ammonia (NH_3), nitrous and nitric oxides (N_2O and NO) from manure management have to be reproduced. Due to geographic and historical reasons, dairy cattle husbandry in Northwest and Central Europe exhibits regional variations in the characteristics of production that should be reflected in emission modelling. However, the models provided in the guidance documents of the Intergovernmental Panel on Climate Change (IPCC) and the UNECE Convention on Long-Range Transboundary Air Pollution do not provide methods that can adequately reflect those variations. The assessment of the emissions of greenhouse gases as described in the IPCC (2006) methodology reflects animal performance to some extent; it does not consider feed properties. The modelling of emissions of nitrogen (N) species from manure management presupposes an adequate assessment of N excretion in both faeces and urine. This cannot be achieved by the IPCC (2006) methodology (IPCC, 2006, pg. 10.57 ff) as it presupposes knowledge of the N intake in feed. The UNECE methodology (EMEP, 2006, pg. B1090-16) does not provide a procedure to quantify N excretion rates.

The module currently describing dairy cattle in the German agricultural emission model GAS-EM is inconsistent as it describes the emissions of CH_4 from enteric fermentation and manure management using a modified IPCC approach (IPCC, 2006, see Dämmgen et al., 2009a). However, the description of the emission of N species relies on N excretions assessed by the DIAS model (Kristensen et al., 1998).

Both models quantify the energies involved, but the results differ in principle. The IPCC model aims at a description of animals *fed as required* with regard to energy, whereas the DIAS model is based on *actual feeding* data obtained from surveys. As the improved GAS-EM dairy cow module CDC09 is to be coupled with land use data, it has to adopt the current German units describing feed properties, in particular the net energy for lactation (NEL). In addition, it has to be consistent with the feeding recommendations provided by GfE (2001).

Furthermore, the assessment of CH_4 emissions from enteric fermentation as well as VS excretion makes use of IPCC default factors to derive gross energy (GE) intakes from metabolizable energy (ME) intakes. Again, a national (NEL based) approach would be desirable.

Thus, the intention of this paper is to provide a harmonized methodology for the assessment of the excretion of both carbon (C) and N in order to calculate the emissions of C and N species in dairy cattle husbandry using the same data set. At the same time, an improvement of the quality of input parameters such as milk yield, animal weight and grazing times is to be achieved. This will be dealt with in a companion paper.

2 Methodological changes and improvements

2.1 Inconsistencies and lack of detail

German emission reporting for dairy cattle relied on the IPCC (2006) methodology for the description of CH_4 emissions and on N excretion data obtained from the DIAS model as described by Kristensen et al. (1998).

A major prerequisite for assessing excretions is the adequate description of energy requirements and restricting entities such as dry matter (DM) intake. Here, the two models used different approaches. Both models could not reflect German feeding practices to the degree desirable.

In some respects, the IPCC model to derive emissions from enteric fermentation and VS excretion rates does not appear to reflect physiological requirements, i.e. in deriving the energy requirements for pregnancy from the animal weight rather than the development of the conception product and the additional energy requirement for maintenance of the mother.

When incorporated into the German emission model GAS-EM in 2005, the DIAS model differed from the other model descriptions available in principle. Whereas the other models used simple methods to derive total N excretions (i.e. the sum of faecal and renal N) from linear regressions using one or few parameters, the DIAS model considered energies needed as well as feed intake during grazing and allowed for the assessment of faecal N, from which the TAN (total ammoniacal nitrogen) contents of excreta could be obtained (Dämmgen and Lüttich, 2005). The knowledge of TAN is a prerequisite to quantify emissions of N species using a mass flow approach (Dämmgen and Hutchings, 2008; Reidy et al., 2008).

The DIAS model was developed to describe the Danish situation. In contrast to other modules within GAS-EM, it does not reflect feeding to requirements (as postulated within IPCC) but rather the actual Danish feeding practice. The model uses Scandinavian Feed Units throughout. These cannot be "translated" into SI units directly. In the DIAS model, input variables for feed differentiate between grazing, roughage fed indoors and concentrates, using

Danish standard values for each of them. As the future module is to be part of integrated assessment studies which include regionally specific feed supply with varying diet components, etc., a module fully compatible with IPCC and UNECE standards was to be developed that used SI units throughout. The latter is a practical prerequisite as the respective German data bases describing feed and animals (DLG, 1997; KTBL, 2006; Beyer et al., undated) strictly use SI units.

The DIAS model as used in GAS-EM was unable to reflect the different feeding practices in typical grassland regions and the other regions, where a mixed diet (containing maize silage as a major component) is fed.

2.2 Preliminary remarks and definitions

2.2.1 Overview

As animal performance presupposes an adequate energy intake, the first step towards an improved model is the assessment of the net energy requirements, the second the calculation of the resulting feed intake and the third the derivation of VS and N excretions. The methods derived will be applied to standard cows and standard feeds to illustrate congruencies and differences.

A companion paper will deal with the different feeding regimes in Germany and their identification (Dämmgen et al., 2009b).

2.2.2 Notation

It is necessary to differentiate between annual, actual daily and mean daily entities. Therefore, it was decided to characterize them with different symbols as follows:

Annual data will be written in upper case letters, daily data in lower case letters. Actual daily data will be identified by an asterisk, mean daily data by plain symbols, e.g.

DM annual amount of DM intake (kg cow^{-1} a^{-1} DM)
dm^* actual daily amount of DM intake (kg cow^{-1} d^{-1} DM)
dm mean daily amount of DM intake (kg cow^{-1} d^{-1} DM)

Symbols for entities used frequently are

m mass of nitrogen
NEL amount of NEL
t time
w animal weight
x, X fraction

Constants and coefficients are normally used in an alphabetical order. A full caption is provided for each equation.

3 Energy requirements and DM intake

IPCC (2006) as well as the relevant German and US bodies (GfE, 2001; BNAR, 2001) deduce energy requirements from the net energy requirements. These are themselves obtained from the partial net energy requirements for maintenance, for obtaining feed, for lactation, pregnancy, growth and work (draft power) that are then summed to yield the total net energy requirements.

3.1 The NEL approach

In order to analyze feed requirements and to characterize feed properties, Austria, Germany and Switzerland (Gruber et al., 2004), Belgium, the Czech Republic, France, Hungary, Italy, the Netherlands (Smink et al., 2005), Slovakia and the USA (BNAR, 2001, pg. 13 ff) use the NEL approach where NEL means net energy for lactation. Irrespective of the literal meaning of the words, the net energy for lactation concept expresses the net energies for all processes in this unit.

3.2 NEL requirements

In analogy with the IPCC (2006) methodology, the overall NEL requirements are expressed by Equation (1):

$$NEL_{\mathrm{tot}} = \alpha \cdot \begin{pmatrix} nel_{\mathrm{m}} + nel_{\mathrm{f}} + nel_{\mathrm{lc}} \\ + nel_{\mathrm{d}} + nel_{\mathrm{p}} + nel_{\mathrm{g}} \end{pmatrix} \qquad (1)$$

where

NEL_{tot} annual NEL required (MJ cow^{-1} a^{-1} NEL)
α time units conversion factor (α = 365 d a^{-1})
nel_{m} NEL required for maintenance (MJ cow^{-1} d^{-1} NEL)
nel_{f} NEL needed to obtain food (MJ cow^{-1} d^{-1} NEL)
nel_{lc} NEL for lactation (MJ cow^{-1} d^{-1} NEL)
nel_{d} NEL required for draft power (MJ cow^{-1} d^{-1} NEL)
nel_{p} NEL required for pregnancy (MJ cow^{-1} d^{-1} NEL)
nel_{g} NEL consumed for growth (MJ cow^{-1} d^{-1} NEL)

3.2.1 NEL requirement for maintenance

NEL requirements for maintenance are obtained using Equation (2) (GfE, 2001, pg. 20):

$$nel_{\mathrm{m}} = a \cdot w_{\mathrm{unit}} \cdot \left(\frac{w}{w_{\mathrm{unit}}} \right)^{0.75} \qquad (2)$$

where

nel_m	net energy required for maintenance (MJ cow^{-1} d^{-1} NEL)
a	constant (a = 0.364 MJ kg^{-1} d^{-1} NEL)
w_{unit}	animal weight unit (w_{unit} = 1 kg cow^{-1})
w	animal weight (kg cow^{-1})

In GfE, Equation (2) uses a coefficient α_{GfE} = 0.293 MJ kg^{-1} d^{-1} NEL to derive nel_m from animal weights. However, it could be shown that this is inadequate and underestimates nel_m considerably. Agnew et al. (2003) and Kebreab et al. (2003) suggest that the factor in the respective equation for ME_m be 0.62 MJ kg^{-1} d^{-1} ME. This can be translated into a coefficient for nel_m of 0.364 MJ kg^{-1} d^{-1} NEL. This factor is used in CDC09: α_{CDC09} = 0.364 MJ kg^{-1} d^{-1} NEL.(1)

3.2.2 NEL requirement to obtain feed

IPCC (2006), pg. 10.16 provides an approach to estimate the NE requirements for activity (i. e. to obtain feed) as proportional to the NE requirements for maintenance. This approach can immediately be rewritten in NEL terms to be used in CDC09. Taking formally into account separate contributions by housing and grazing one arrives at

$$nel_f = \left(c_{house} \cdot \left(1 - \frac{\tau_{pasture}}{\alpha} \right) + c_{pasture} \cdot \frac{\tau_{pasture}}{\alpha} \right) \cdot nel_m \quad (3)$$

where

nel_f	NEL needed to obtain feed (in analogy to IPCC) (MJ cow^{-1} d^{-1} NEL)
c_{house}	coefficient for housing (c_{house} = 0.00; IPCC(2006)-10.17, Table 10.5)
$\tau_{pasture}$	duration of grazing time (d a^{-1})
α	time units conversion factor (α = 365 d a^{-1})
$c_{pasture}$	coefficient for pasture ($c_{pasture}$ = 0.17; IPCC(2006)-10.17, Table 10.5)
nel_m	NEL required for maintenance (in analogy to IPCC) (MJ cow^{-1} d^{-1} NEL)

3.2.3 NEL requirements for lactation

In contrast to IPCC (2006), the approach proposed includes the energy requirements for the synthesis of milk protein (GfE, 2001, pg. 21f):

$$nel_{lc} = y_m \cdot \left(\begin{array}{l} c_{lact\,1,\,GfE} + c_{lact\,2,\,GfE} \cdot x_{fat} \\ + c_{lact\,3,\,GfE} \cdot x_{MP} + d \end{array} \right) \cdot a \quad (4)$$

where

nel_{lc}	net energy requirements for lactation (MJ cow^{-1} d^{-1} NEL)
y_m	milk yield (kg cow^{-1} d^{-1})
$c_{lact\,1,\,GfE}$	constant ($c_{lact\,1,\,GfE}$ = 0.95 MJ kg^{-1})
$c_{lact\,2,\,GfE}$	coefficient ($c_{lact\,2,\,GfE}$ = 38 MJ kg^{-1})
x_{fat}	mass fraction of fat (kg kg^{-1})
$c_{lact\,3,\,GfE}$	coefficient ($c_{lact\,3,\,GfE}$ = 21 MJ kg^{-1})
x_{MP}	mass fraction of milk protein (kg kg^{-1})
d	constant (d = 0.1 MJ kg^{-1} NEL)
a	correction factor (a = 1 MJ MJ^{-1} NEL for daily data and 1.04 MJ MJ^{-1} NEL for annual mean data)

In GfE (2001), the application of Equation (4) presupposes the knowledge of daily milk yields and milk constituents. However, these vary with time (see Figure 1).

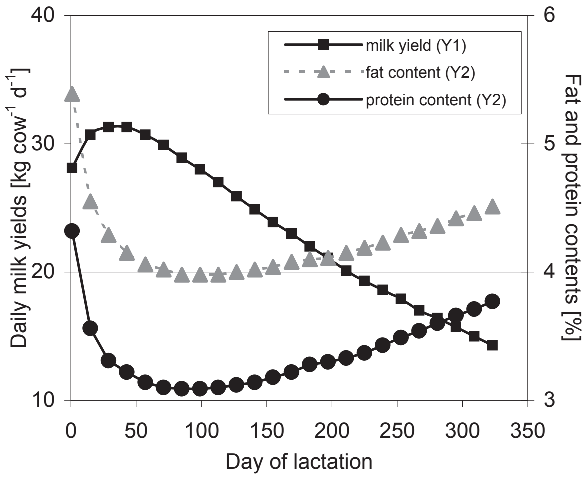

Figure 1:

Typical time series of daily milk yields and milk fat and protein contents (redrawn after Greimel and Steinwidder, 1998)

Exemplary model calculations indicate that the use of mean daily milk yields and constituent concentrations instead of actual data requires a correction factor $a \neq 1$.

(1) In principle, the NEL system does not demand that metabolizable energies, ME and NEL be interconverted. However, the NEL approach documented in GfE (2001) does not include the NEL requirements for grazing. Since the extent to which cattle obtain feed by grazing varies regionally, it is desirable to account for the consequent energy demand. As this is a minor constituent of the total NEL requirements, the conversion from NEL to ME proposed by van Es (1975) (see GfE, 2001, pg. 19) is used:

$$ME = \frac{NEL}{a + b \cdot X_{ME}}$$

where

NEL	net energy for lactation (MJ NEL)
ME	metabolizable energy (MJ ME)
α	constant (α = 0.4632)
b	constant (b = 0.24)
X_{ME}	metabolizability (assumed: X_{ME} = 0.60 MJ MJ^{-1})

3.2.4 NEL requirements for pregnancy

The GfE (2001), pg. 23, methodology provides absolute figures derived from the energy used for the development of the conception products and the udder:

$$NEL_{\mathrm{p}}^{*} = NEL_{\mathrm{cp}}^{*} + NEL_{\mathrm{u}}^{*} \tag{5}$$

where

- NEL_{p}* actual net energy required for pregnancy (MJ calf^{-1})
- NEL_{cp}* actual net energy required for the development of the uterus including the conception product (MJ calf^{-1})
- NEL_{u}* actual net energy required for the development of the udder (MJ calf^{-1}) NEL_{u}* = 31 MJ calf^{-1} d^{-1} (GfE, 2001, pg.23)

The net energy lactation for the development of the conception product is a function of the daily growth rate:

$$NEL_{\mathrm{cp}}^{*} = \sum_{\tau_{\mathrm{p}}=1}^{\tau_{\mathrm{p,\,fin}}} a \cdot w_{\mathrm{calf}} \cdot e^{\mathrm{b} \cdot \tau_{\mathrm{p}}} \tag{6}$$

where

- $NEL*_{\mathrm{cp}}$ actual NEL required for the development of the conception product (MJ calf^{-1})
- a constant (a = 0.000122 MJ kg^{-1} d^{-1})
- w_{calf} birth weight of calf (kg calf^{-1})
- b constant (b = 0.0165)
- τ_{p} day after conception
- $\tau_{\mathrm{p,\,fin}}$ day of birth ($\tau_{\mathrm{p,\,fin}}$ = 279)

As calf weights are not available on a national scale in Germany, a standard calf weight of 36 kg calf^{-1} is assumed. In this case, NEL_{cp}* and NEL_{u}* add up to overall NEL_{p}* requirements of 917 MJ calf^{-1}, independent of the calf's or the mother's weight.[2]

The GfE approach is used in CDC09.

Again, the daily NEL requirement for the development of the conception product, nel_{p}, is then calculated from NEL_{p}* as a function of the interval between calvings according to

$$nel_{\mathrm{p}} = \frac{NEL_{\mathrm{p}}^{*}}{t_{\mathrm{ibc}}^{*}} \tag{7}$$

where

- nel_{p} NEL required for pregnancy (MJ cow^{-1} d^{-1} NEL)
- NEL_{p}* NEL required for pregnancy (NEL_{p}* = 917 MJ calf^{-1} NEL)
- t_{ibc}* absolute duration of interval between calvings (d calf^{-1})

3.2.5 NEL requirements for growth

GfE (2001), pg. 22, relate the NEL requirements for growth to the weight gain per year. This approach is also used in CDC09.

$$nel_{\mathrm{g,\,GfE}} = \frac{a \cdot \Delta w}{\alpha} \tag{8}$$

where

- $nel_{\mathrm{g,\,GfE}}$ NEL required for growth (MJ cow^{-1} d^{-1})
- a constant (a = 25.5 MJ kg^{-1} NEL)
- Δw weight gain (kg cow^{-1} a^{-1})
- α time units conversion factor (α = 365 d a^{-1})

3.3 Linking feed intake with energy requirements

If an animal is fed according to requirements, the net energy requirements (NEL_{tot}) have to be met by the net energy for lactation provided in feed (NEL_{feed}):

$$NEL_{\mathrm{feed}} = NEL_{\mathrm{tot}} \tag{9}$$

Feed will be supplied in concentrates and roughage inside the animal house and during grazing. The respective shares are also governed by the DM intake of the animals, as DM intake is limited. Once the DM intake and the feed properties are known, the intakes of GE, DE, ME, NEL and N can be deduced.

3.3.1 Treatment of DM intake

DLG (1986) provides a model of dry matter intake. DLG (2006) provides an alternative and much more detailed approach, but demands input data which are not available at the spatial and temporal scales used for inventory construction.

The methodology adopted for CDC09 differentiates between the DM intakes during lactation and during the dry period:

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[2] The mean duration of pregnancy is 279 d. Thus an amount equal to 266 MJ calf^{-1} NEL results for the development of calf and uterus. GfE (2001), pg. 23, recommend for the NEL required for the udder during the last weeks of pregnancy 13 MJ calf^{-1} d^{-1} NEL for weeks 6 to 4 before calving, as well as 18 MJ calf^{-1} d^{-1} for the final three weeks before calving which results in 651 MJ calf^{-1} NEL. Thus, the energy requirements for conception products and the development of the udder add up to 917 MJ calf^{-1} NEL .

$$DM = DM_{\text{lact}} + DM_{\text{dry}} \tag{10}$$

where

DM	total intake of DM (kg cow^{-1} a^{-1} DM)
DM_{lact}	intake of DM during the lactation period (kg cow^{-1} a^{-1} DM)
DM_{dry}	intake of DM during the dry period (kg cow^{-1} a^{-1} DM)

and

$$\begin{aligned} DM_{\text{lact}} &= DM_{\text{conc, lact}} + DM_{\text{rough, lact}} \\ &= \left(dm_{\text{conc, lact}} + dm_{\text{rough, lact}}\right) \cdot t_{\text{lact}} \end{aligned} \tag{11}$$

$$\begin{aligned} DM_{\text{dry}} &= DM_{\text{conc, dry}} + DM_{\text{rough, dry}} \\ &= \left(dm_{\text{conc, dry}} + dm_{\text{rough, dry}}\right) \cdot t_{\text{dry}} \end{aligned} \tag{12}$$

where

DM_{lact}	annual intake of DM during the lactation period (kg cow^{-1} a^{-1} DM)
$DM_{\text{conc, lact}}$	annual intake of DM during the lactation period with concentrates (kg cow^{-1} a^{-1} DM)
$DM_{\text{rough, lact}}$	annual intake of DM during the lactation period with roughage (kg cow^{-1} a^{-1} DM)
$dm_{\text{conc, lact}}$	daily intake of DM during the lactation period with concentrates (kg cow^{-1} d^{-1} DM)
$dm_{\text{rough, lact}}$	daily intake of DM during the lactation period with roughage (kg cow^{-1} d^{-1} DM)
t_{lact}	duration of the lactation period (d a^{-1})
DM_{dry}	annual intake of DM during the dry period (kg cow^{-1} a^{-1} DM)
$DM_{\text{conc, dry}}$	annual intake of DM during the dry period with concentrates (kg cow^{-1} a^{-1} DM)
$DM_{\text{rough, dry}}$	annual intake of DM during the dry period with roughage (kg cow^{-1} a^{-1} DM)
$dm_{\text{conc, dry}}$	daily intake of DM during the dry period with concentrates (kg cow^{-1} d^{-1} DM)
$dm_{\text{rough, dry}}$	daily intake of DM during the dry period with roughage (kg cow^{-1} d^{-1} DM)
t_{dry}	duration of the dry period (d a^{-1})

The assessment of the respective entities is described in the following chapters. The duration of the lactation and dry periods is calculated in Chapter 6.1.1.3.

3.3.1.1 DM intake during lactation

The daily amount of DM intake during the lactation period can be described according to the procedure proposed by DLG (1986) and subsequently modified (see Spiekers et al., 2006):

$$\begin{aligned} dm_{\text{rough, lact}} &= a \cdot w + b \cdot \left(\frac{X_{\text{NEL, rough, lact}}}{X_{\text{NEL, rough, ref}}}\right)^{c} \\ &\quad - d \cdot dm^{2}_{\text{conc, lact}} + f \cdot \max\left(y_{\text{ECM}} - e; 0\right) \end{aligned} \tag{13}$$

where

$dm_{\text{rough, lact}}$	daily DM intake in roughage during lactation (kg cow^{-1} d^{-1} DM)
a	constant (a = 0.006 d^{-1})
w	animal weight (kg cow^{-1})
b	constant (b = 0.19 kg cow^{-1} d^{-1})
$X_{\text{NEL rough, lact}}$	NEL content of roughage (MJ kg^{-1} NEL)
$X_{\text{NEL, rough, ref}}$	reference NEL content of roughage ($X_{\text{NEL, rough, ref}}$ = 1 MJ kg^{-1} NEL)
c	exponent (c = 2.16)
d	constant (d = 0.026 kg^{-1} cow a)
$dm_{\text{conc, lact}}$	daily DM intake in concentrates during lactation (in kg cow^{-1} d^{-1} DM)
y_{ECM}	milk yield (energy corrected) (kg cow^{-1} d^{-1})
e	constant (e = 25 kg cow^{-1} d^{-1})
f	constant (f = 0.1)

$X_{\text{NEL, rough, lact}}$ is the weighted mean of the NEL contents of the roughage fed.

It should be noted that the results obtained with this equation are still consistent with DLG (2006) (see Gruber et al., 2006).

3.3.1.2 Energy intake and net energy requirements during lactation

The intake of net energy for lactation during the lactation period can be described as in Equation (14):

$$\begin{aligned} & dm_{\text{rough, lact}} \cdot X_{\text{NEL, rough, lact}} \\ & \quad + dm_{\text{conc, lact}} \cdot X_{\text{NEL, conc, lact}} = \frac{NEL_{\text{lact}}}{t_{\text{lact}}} \end{aligned} \tag{14}$$

where

$dm_{\text{rough, lact}}$	daily DM intake in roughage during lactation (kg cow^{-1} d^{-1} DM)
$X_{\text{NEL rough, lact}}$	NEL content of roughage during lactation (MJ kg^{-1} NEL)
$dm_{\text{conc, lact}}$	daily DM intake in concentrates during lactation (kg cow^{-1} d^{-1} DM)

$X_{\text{NEL conc, lact}}$	NEL content of concentrates during lactation (MJ kg^{-1} NEL)
NEL_{lact}	net energy input required during the lactation period (MJ cow^{-1} a^{-1} NEL)
t_{lact}	duration of the lactation period (d a^{-1})

3.3.1.3 Combining DM intake and energy requirements during lactation

Equations (13) and (14) are two relations for the two unknowns $dm_{\text{rough, lact}}$ and $dm_{\text{conc, lact}}$. The solution of this set of equations requires a rearrangement of Equation (14):

$$dm_{\text{rough, lact}} = \frac{\frac{NEL_{\text{lact}}}{t_{\text{lact}}} - dm_{\text{conc, lact}} \cdot X_{\text{NEL, conc, lact}}}{X_{\text{NEL, rough, lact}}} \qquad (15)$$

Combining Equations (13) and (15) and rearranging into a standard form for second order equations yields

$$\frac{NEL_{\text{lact}}}{t_{\text{lac}} \cdot X_{\text{NEL, rough, lact}}} - dm_{\text{conc, lact}} \cdot \frac{X_{\text{NEL, conc, lact}}}{X_{\text{NEL, rough, lact}}}$$
$$- a \cdot w - b \cdot \left(\frac{X_{\text{NEL, rough, lact}}}{X_{\text{NEL, rough, ref}}} \right)^{c} + d \cdot dm^{2}_{\text{conc, lact}} \qquad (16)$$
$$- f \cdot \max\left(y_{\text{ECM}} - e; 0\right) = 0$$

which is equivalent to

$$dm^{2}_{\text{conc, lact}} + B \cdot dm_{\text{conc, lact}} + C = 0 \qquad (17)$$

with

$$B = -\frac{X_{\text{NEL, conc, lact}}}{X_{\text{NEL, rough, lact}}} \cdot \frac{1}{d} \qquad (18)$$

and

$$C = \left(\begin{array}{l} \frac{NEL_{\text{lact}}}{t_{\text{lac}} \cdot X_{\text{NEL, rough, lact}}} - a \cdot w - \\ b \cdot \left(\frac{X_{\text{NEL, rough, lact}}}{X_{\text{NEL, rough, ref}}} \right)^{c} - f \cdot \max\left(y_{\text{ECM}} - e; 0\right) \end{array} \right) \cdot \frac{1}{d} \qquad (19)$$

The meaningfull solution of this equation is

$$dm_{\text{conc, lact}} = -\frac{B}{2} - \sqrt{\left(\frac{B}{2}\right)^{2} - C} \qquad (20)$$

Inserting $dm_{\text{conc, lact}}$ into Equation (15) yields $dm_{\text{rough, lact}}$.

3.3.1.4 NEL requirements during the dry period

In analogy to the DM intake, the NEL intake is considered separately for the lactation and dry periods:

$$NEL_{\text{tot}} = NEL_{\text{lact}} + NEL_{\text{dry}} \qquad (21)$$

where

NEL_{tot}	annual total net energy requirements (MJ cow^{-1} a^{-1} NEL)
NEL_{lact}	annual net energy requirements during the lactation period (MJ cow^{-1} a^{-1} NEL)
NEL_{dry}	annual net energy requirements during the dry period (MJ cow^{-1} a^{-1} NEL)

At present, the information available from GfE (2001) does not allow for the variation in animal weight and the length of the dry period. However, from Table 1.4.3 in GfE (2001) an estimate of the actual amount of NEL^*_{dry} can be derived. According to this data set, mean NEL requirements for weeks 6 to 4 before birth are 50.6 MJ cow^{-1} d^{-1} NEL. For the final three weeks of pregnancy, NEL requirements are 55.6 MJ cow^{-1} d^{-1} NEL.

Thus, the overall actual requirement can be deduced as:

$$NEL^{*}_{\text{dry}} = t^{*}_{\text{dry}} \cdot \frac{a+b}{2} \qquad (22)$$

where

$NEL_{\text{dry}}{}^*$	actual NEL requirements during the dry period (MJ cow^{-1} NEL)
$t_{\text{dry}}{}^*$	actual duration of the dry period (d)
a	constant (a = 50.6 MJ cow^{-1} d^{-1})
b	constant (b = 55.6 MJ cow^{-1} d^{-1})

Hence, annual and daily requirements are, respectively,

$$NEL_{\text{dry}} = t_{\text{dry}} \cdot \frac{a+b}{2} \qquad (23)$$

$$nel_{\text{dry}} = \frac{NEL_{\text{dry}}}{t_{\text{dry}}} \qquad (24)$$

where

NEL_{dry}	annual NEL requirements during the dry period (MJ cow^{-1} a^{-1} NEL)
t_{dry}	annual duration of the dry period (d)
a	constant (a = 50.6 MJ cow^{-1} d^{-1} NEL)
b	constant (b = 55.6 MJ cow^{-1} d^{-1} NEL)
nel_{dry}	daily NEL requirements during the dry period (MJ cow^{-1} d^{-1} NEL)

3.3.1.5 DM intake during the dry period

DM intake during the dry period is defined by Equation (12).

In CDC09, the daily DM intake with concentrates, $dm_{\text{conc, dry}}$, is assumed to be 1 kg cow^{-1} d^{-1} fresh matter; the DM content is constant and 0.88 kg kg^{-1}.

Given $dm_{\text{conc, dry}}$, the balance of daily NEL requirements and NEL contents in the feed can be rearranged to yield the daily DM intake with roughage:

$$dm_{\text{rough, dry}} = \frac{nel_{\text{dry}} - dm_{\text{conc, dry}} \cdot X_{\text{NEL, conc}}}{X_{\text{NEL, rough}}} \qquad (25)$$

where

$dm_{\text{rough, dry}}$	daily intake of DM during the dry period with roughage (kg cow^{-1} d^{-1} DM)
nel_{dry}	mean daily NEL intake feed during the dry period (MJ cow^{-1} d^{-1} NEL)
$dm_{\text{conc, dry}}$	daily intake of DM during the dry period with roughage (kg cow^{-1} a^{-1} DM)
$X_{\text{NEL, conc}}$	NEL content of concentrates during the dry period (MJ kg^{-1} NEL)
$X_{\text{NEL, rough}}$	NEL content of roughage during the dry period (MJ kg^{-1} NEL)

The assessment of the annual intake of DM with concentrates and roughage presupposes the knowledge of the annual duration of the dry period:

$$DM_{\text{conc, dry}} = dm_{\text{conc, dry}} \cdot t_{\text{dry}} \qquad (26)$$

likewise

$$DM_{\text{rough, dry}} = dm_{\text{rough, dry}} \cdot t_{\text{dry}} \qquad (27)$$

where

$DM_{\text{conc, dry}}$	annual intake of DM during the dry period in concentrates (kg cow^{-1} a^{-1} DM)
$dm_{\text{conc, dry}}$	daily intake of DM during the dry period in concentrates (kg cow^{-1} d^{-1} DM)
t_{dry}	duration of the dry period (d a^{-1})
$DM_{\text{conc, dry}}$	annual intake of DM during the dry period in concentrates (kg cow^{-1} a^{-1} DM)
$dm_{\text{rough, dry}}$	daily intake of DM during the dry period in roughage (kg cow^{-1} d^{-1} DM)

3.3.2 Intake during grazing and in housing

The shares of DM intake in roughage consumed indoors and during grazing are assumed to be proportional to the respective shares in time spent indoors and grazing. Time spent in the dairy parlour is considered as time without feeding roughage.

$$DM_{\text{rough}} = DM_{\text{rough, house}} + DM_{\text{rough, graz}} \qquad (28)$$

$$\frac{DM_{\text{rough, house}}}{DM_{\text{rough, graz}}} = \frac{t_{\text{house}}}{t_{\text{graz}}} \qquad (29)$$

where

DM_{rough}	annual intake of DM with roughage (kg cow^{-1} a^{-1} DM)
$DM_{\text{rough ,house}}$	annual intake of DM with roughage consumed indoors (kg cow^{-1} a^{-1} DM)
$DM_{\text{rough ,graz}}$	annual intake of DM with roughage during grazing (kg cow^{-1} a^{-1} DM)
t_{house}	time spent indoors (a a^{-1})
t_{graz}	time spent grazing (a a^{-1})

hence

$$DM_{\text{rough, graz}} = DM_{\text{rough}} \cdot \frac{t_{\text{graz}}}{t_{\text{house}} + t_{\text{graz}}} \qquad (30)$$

Note that the time spent indoors excludes the time needed for milking. Grazing influences the mean NEL content of roughage needed in Equation (13) and the subsequent discussion.

3.4 Gross energy intake and methane emissions from enteric fermentation

In the IPCC methodology, the amount of CH_4 originating from enteric fermentation is a function of gross energy intake (GE).

3.4.1 Gross energy inputs

If the GE contents of the diet components are known, GE can be determined from the respective amounts of the constituents and their GE contents:

$$GE = \sum_{i=1}^{n} m_i \cdot X_{\text{GE,i}} \qquad (31)$$

where

GE	gross energy intake (MJ cow^{-1} a^{-1} GE)
m_i	amount of feed taken in with component i (kg a^{-1})

$X_{GE,\,i}$ GE content of feed component i (MJ kg^{-1} GE)

3.4.2 Methane conversion factor

The CH_4 emission from enteric fermentation is derived from the gross energy intake and the methane conversion factor as follows:

$$EF_{CH4,\,ent} = GE \cdot \frac{x_{CH4} \cdot \alpha}{\eta_{CH4}} \tag{32}$$

where

$EF_{CH4,\,ent}$	emission factor for CH_4 from enteric fermentation (kg cow^{-1} a^{-1} CH_4)
GE	gross energy intake (MJ cow^{-1} d^{-1})
x_{CH4}	methane conversion factor (MJ MJ^{-1})
α	time units conversion factor (α = 365 d a^{-1})
η_{CH4}	energy content of methane (η_{CH4} = 55.65 MJ (kg CH_4)$^{-1}$)

Ellis et al. (2007) investigated 10 regression approaches relating $x_{CH4,\,GE}$ to feed. The equation using entities provided in the German inventory yielding the smallest root mean square prediction error uses the DM intake as in the following equation:

$$x_{CH4,GE} = \frac{E_{CH4}}{GE} = \frac{1}{GE} \cdot \left(a + b \cdot DM\right) \tag{33}$$

where

$x_{CH4,\,GE}$	methane conversion factor related to GE (MJ MJ^{-1})
E_{CH4}	methane emitted daily (MJ cow^{-1} d^{-1})
GE	gross energy intake (MJ cow^{-1} d^{-1})
a	constant (a = 3.23 MJ cow^{-1} d^{-1})
b	coefficient (b = 0.809 MJ MJ^{-1})
DM	DM intake (kg cow^{-1} d^{-1})

Equation (33) is used in CDC09.

4 Nitrogen intake and excretion

4.1 Nitrogen intake

The N intake can be calculated using the following equation:

$$m_{feed} = x_N \cdot \sum_{1}^{i} DM_i \cdot x_{N,\,XP,\,i} \tag{34}$$

where

m_{feed}	amount of nitrogen in feed (kg cow^{-1} a^{-1} N)
x_N	nitrogen content of crude protein (x_N = 1/6.25 kg kg^{-1} N)
DM_i	amount of DM consumed with feed constituent i (kg cow^{-1} a^{-1} ME)
$x_{N,\,XP,\,i}$	crude protein content of feed constituent i (kg kg^{-1} XP)

A feed constituent can either be fed in animal housing or consumed by grazing.

However, the applicability of this equation is restricted to those cases where both the amounts fed and the diet composition are known. For emission inventories, the amount of DM intake can be modelled adequately. For the protein content of the feed assumptions have to be made with regard to typical regional feeds.

4.2 Nitrogen excretion rates

The accuracy of the modelling of the emission of N species depends significantly on the accuracy of the assessment of N excretion rates (Webb et al., 2005). Both overall and renal excretion rates are needed (Dämmgen and Hutchings, 2008).

4.2.1 Total nitrogen excretion

The nitrogen balance offers a direct assessment of the amount of N excreted:

$$\begin{aligned} m_{excr} &= m_{faeces} + m_{urine} \\ &= m_{feed} - \left(m_l + m_g + m_p + m_s\right) \end{aligned} \tag{35}$$

where

m_{excr}	amount of N excreted in faeces and urine (kg cow^{-1} a^{-1} N)
m_{faeces}	amount of N excreted in faeces (kg cow^{-1} a^{-1} N)
m_{urine}	amount of N excreted in urine (kg cow^{-1} a^{-1} N)
m_{feed}	amount of N taken in in feed (kg cow^{-1} a^{-1} N)
m_l	amount of N exported in milk (kg cow^{-1} a^{-1} N)
m_g	amount of N retained in weight gained (kg cow^{-1} a^{-1} N)
m_p	amount of N excreted in conception products (kg cow^{-1} a^{-1} N)
m_s	amount of N lost in skin and hair (kg cow^{-1} a^{-1} N)

4.2.2 Composition of excreted nitrogen

Emissions of N species are usually related to the reactive nitrogen excreted. This is almost entirely excreted with urine and consists of urea and allanthoin (see e.g. Dämmgen and Erisman, 2005). As the enzyme urease is ubiquitous, these compounds decompose readily to yield ammonium (NH_4). NH_4 and NH_3 in excreta are summarized as total ammoniacal nitrogen (TAN).

It would be advantageous to assess renal N excretion directly. However, no simple modelling procedure is available. Complex models such as DAFOSYM (Rotz et al., 1992), the INRA model (Martin and Sauvant, 2007) or Molly (e.g. Johnson and Baldwin, 2008) require input information that is not normally available. Renal N excretion cannot be related to the N intake with feed satisfactorily (e.g. Rohr, 1992; Kebreab et al., 2001, 2002; Gehman et al., 2008). For the purpose of inventory making, measurements of milk urea N are likely to provide an adequate tool to estimate renal N excretion (e.g. Lebzien et al., 2008). These data is not available yet, so an indirect approach of quantifying faecal N excretion is used:

$$m_{\text{urine}} = m_{\text{excr}} - m_{\text{faeces}} \tag{36}$$

where

m_{urine}	amount of N excreted in urine (kg cow^{-1} a^{-1} N)
m_{excr}	amount of N excreted in faeces und urine (kg cow^{-1} a^{-1} N)
m_{faeces}	amount of N excreted in faeces (kg cow^{-1} a^{-1} N)

Faecal N excretion is almost independent of the N intake with feed. It is rather dominated by microbial XP synthesis in the rumen. The calculation procedure used in DIAS and CDC09 (Equation (37), is based on Danish experimental results, and takes N input into account. However, the effect of N in feed is comparatively small:

$$m_{\text{faeces}} = \alpha \cdot \beta \cdot \left(a \cdot \frac{m_{\text{feed}}}{\alpha} + \left(b \cdot \frac{DM}{\alpha} + c \cdot \left(\frac{DM}{\alpha} \right)^2 \right) \cdot x_{\text{N}} \right) \tag{37}$$

where

m_{faeces}	N excreted in faeces (kg cow^{-1} a^{-1} N)
α	time units conversion factor (α = 365 d a^{-1})
β	mass units conversion factor (β = 0.001 kg g^{-1})
m_{feed}	N intake in feed (kg cow^{-1} a^{-1} N)
a	constant (a = 0.04 g kg^{-1})
DM	DM intake (kg cow^{-1} a^{-1})
b	constant (b = 20 g kg^{-1})
c	constant (c = 1.8 g kg^{-2} cow d)
x_{N}	nitrogen content of crude protein (x_{N} = 1/6.25 kg kg^{-1} N)

The TAN content of excreta X_{TAN} is then calculated as percentage related to the total amount of N excreted.

$$X_{\text{TAN}} = \frac{m_{\text{urine}}}{m_{\text{excr}}} \tag{38}$$

4.2.3 Nitrogen intake with feed

CDC09 calculates the N intake with feed as follows:

$$m_{\text{feed}} = \left(dm_{\text{rough}} \cdot X_{\text{XP, rough}} + dm_{\text{conc}} \cdot X_{\text{XP, conc}} \right) \cdot x_{\text{N}} \tag{39}$$

where

m_{feed}	amount of N intake in feed (kg cow^{-1} a^{-1} N)
dm_{rough}	DM intake with roughage (kg cow^{-1} a^{-1} DM)
$X_{\text{XP, rough}}$	mean XP content of roughage (kg kg^{-1} XP)
dm_{conc}	DM intake with concentrates (kg cow^{-1} a^{-1} DM)
$X_{\text{XP, conc}}$	mean XP content of concentrates (kg kg^{-1} XP)
x_{N}	N content of XP (x_{N} = 1/6.25 kg kg^{-1} N)

4.2.4 Nitrogen exported in milk

In CDC09, m_{l} is related to milk yield and milk protein content

$$m_{\text{l}} = Y_{\text{m}} \cdot X_{\text{MP, milk}} \cdot x_{\text{N, milk}} \tag{40}$$

where

m_{l}	amount of N exported in milk (kg cow^{-1} a^{-1} N)
Y_{m}	annual milk yield (kg cow^{-1} a^{-1})
$X_{\text{MP, milk}}$	protein content of milk (kg kg^{-1} MP)
$x_{\text{N, milk}}$	N content of milk protein ($x_{\text{N, milk}}$ = 1/6.38 kg kg^{-1} N)

4.2.5 Nitrogen in weight gain and conception products

A similar approach is used to quantify m_{g} and m_{p}:

$$m_{\text{g}} = \Delta w \cdot x_{\text{N, cow}} \tag{41}$$

where

m_{g}	amount of N retained in weight gained (kg cow^{-1} a^{-1} N)
Δw	weight gain (kg cow^{-1} a^{-1})
$x_{\text{N, cow}}$	mean N content of whole cow body ($x_{\text{N, cow}}$ = 0.0256 kg kg^{-1} N) (DLG, 2005)

$$m_{\mathrm{p}} = n \cdot w_{\mathrm{calf}} \cdot x_{\mathrm{N,\,calf}} \tag{42}$$

where

m_p amount of N retained in conception product (kg cow^{-1} a^{-1} N)
n number of calves per cow and year
w_{calf} weight of calf (kg calf^{-1})
$x_{N,\,cow}$ mean N content of whole calf body ($x_{N,\,calf}$ = 0.0296 kg kg^{-1} N) (DLG, 2005)

4.2.6 Nitrogen in skin and hair

The amount of N lost in hair and skin can be obtained from a simple approach:

$$m_s = \alpha \cdot \beta \cdot d \cdot \left(\frac{w}{w_{\mathrm{unit}}} \right)^e \tag{43}$$

where

m_s N losses in skin and hair N (kg cow^{-1} a^{-1} N)
α time units conversion factor (α = 365 d a^{-1})
β mass units conversion factor (β = 0.001 kg g^{-1})
d coefficient (d = 0.018 g cow^{-1} d^{-1} N)
w animal weight (kg cow^{-1})
w_{unit} animal weight unit (w_{unit} = 1 kg cow^{-1})
e exponent (e = 0.75)

4.2.7 Nitrogen entering pasture and the manure management system

The overall amount of nitrogen considered for the assessment of the pasture and manure management mass flow is:

$$m_{\mathrm{MM}} = m_{\mathrm{faeces}} + m_{\mathrm{urine}} + m_{\mathrm{s}} \tag{44}$$

where

m_{MM} amount of N entering pasture and the manure management system (kg cow^{-1} a^{-1} N)
m_{faeces} amount of N excreted with faeces (kg cow^{-1} a^{-1} N)
m_{urine} amount of N excreted with urine (kg cow^{-1} a^{-1} N)
m_s amount of N lost in skin and hair (kg cow^{-1} a^{-1} N)

The amounts of organic N, $m_{org,\,MM}$, and TAN_{MM} are:

$$m_{\mathrm{org,\,MM}} = m_{\mathrm{faeces}} + m_{\mathrm{s}} \tag{45}$$

$$TAN_{\mathrm{MM}} = m_{\mathrm{urine}} \tag{46}$$

5 Excretion of volatile solids (VS)

In its Tier 2 approach, IPCC (2006), pg. 10.42, relates the emissions of CH_4 from manure management to the rate of volatile solids excreted. The equation may be re-written as follows:

$$VS_{\mathrm{excr}} = \frac{1}{b} \cdot (FE + UE) \cdot (1 - X_{\mathrm{ash}}) \tag{47}$$

where

VS_{excr} amount of VS excreted daily (kg cow^{-1} d^{-1} VS)
b conversion factor for dietary GE per kg of DM (b = 18.45 MJ kg^{-1})
FE faecal energy excreted (MJ cow^{-1} d^{-1})
UE urine energy excreted (MJ cow^{-1} d^{-1})
X_{ash} ash content of faeces (kg kg^{-1})

IPCC (2006), vol. 3, pg. 4.23, explains b as being "the energy density of feed is about 18.45 MJ per kg of DM. This value is relatively constant across a wide range of forage and grain-based feeds commonly consumed by livestock."

The **faecal energy** can be expressed as

$$FE = GE \cdot (1 - X_{\mathrm{DE}}) \tag{48}$$

where

GE gross energy intake (MJ cow^{-1} d^{-1})
X_{DE} digestibility of organic matter (kg kg^{-1})

For simplicity, X_{DE} is taken to be constant for a given diet, even though in reality, it will vary between breeds and physiological states.

The **urine energy** is related to the gross energy

$$UE = a \cdot GE \tag{49}$$

where a is generally assumed to be a more or less constant proportion 0.03 < a < 0.07. IPCC (2006), pg. 10.42, suggests

a constant (a = 0.04 MJ MJ^{-1})

The **ash content** X_{ash} varies with time and feed. For the purpose of emission reporting, IPCC (2006) suggests a constant ash content of 0.08 kg kg^{-1}. In CDC09 an ash content (national value) of 0.133 kg kg^{-1} as proposed in Hennig and Poppe (1975), pp 172, is used.

In principle, CH_4 emissions from manure management are related to the amount of **degradable volatile solids**. The relation between the excretion rates of total VS (VS_{excr}), degradable VS (VS_d) and non-degradable VS (VS_{nd}) is as follows:

$$VS_{excr} = VS_d + VS_{nd} \quad (50)$$

In a first approach, VS_d is proportional to VS_{excr}. The factor relating the two is depending on the composition of the excreta. The methodology proposed by IPCC (2006), pg. 10.41, considers this in a "maximum methane producing capacity for manure", B_0, which is listed for each animal category. IPCC (2000), pg. 4.32, recommends that countries should establish national B_0 values. So far, CDC09 relies on the IPCC default value listed in IPCC (2006), Tables 10A-4.

6 Comparison between results obtained with CDC09 and the IPCC (2006) methodologies and the application of the DLG (2005) dataset

It is obvious that the results achieved with the CDC09 approach are likely to differ from those obtained with the IPCC methodology. The different approaches are described and the respective results compared (Chapter 6.1).

It is also important to investigate to what extent modelling of the excretion of animals fed according to requirements deviates from empirical data. Hence, the modelled data are compared to a data set provided in DLG (2005). This data set describes feed intake and N excretion as a function of energy corrected milk (ECM) yield (6000, 8000 and 10000 kg cow^{-1} a^{-1} ECM) and feed composition. For each feeding regime, feed composition varies with performance. (Chapter 6.2)

6.1 Comparison with IPCC

CDC09 was compared with IPCC (2006) for a standard German dairy cow for a range of milk yields and for two contrasting diets using typical milk fat and protein contents.

The national data used for the comparison was obtained partly from the literature. In some cases, it had to be derived from national data sets (e.g. duration of the lactation period). In other cases, assumptions have to be made (e.g. (e.g. weight gain and duration of dry period).

6.1.1 Data set used for comparison

6.1.1.1 Animal performance data

The comparison makes use of the following performance and feed data:

- mean animal weight: 630 kg cow^{-1}, weight gain 80 kg cow^{-1} in 3 a;
- milk fat content 40 g kg^{-1}, milk protein content 34 g kg^{-1}

Milk yields between 4500 and 10000 kg cow^{-1} a^{-1} are taken into consideration.

6.1.1.2 Diets

The exemplary calculations made for this paper differentiate between two common diets. Diet 1 is based on a mix of maize and grass silages combined with barley straw and high protein concentrates (Table 1). Diet 2 reflects the situation in the northwest German grassland region and is based on grass silage and straw combined with protein poor concentrates (Table 2). The feed properties are listed in Table 3.

Table 1:
Composition of diet 1 ("mixed") used for intercomparison

Feed components	Share
Roughage	kg kg^{-1} DM
grass silage	0.46
maize silage	0.46
barley straw	0.08
Concentrates	kg kg^{-1} DM
standard concentrates MLF 18/3	1.00

Table 2:
Composition of diet 2 ("grass") used for intercomparison

Feed components	Share
Roughage	kg kg^{-1} DM
grass silage	0.97
barley straw	0.03
Concentrates	kg kg^{-1} DM
barley	0.88
sugar beet shreds	0.12

Table 3:

Standard data used to establish the comparison data

Feed constituent	DM content	ME in DM	NEL in DM	DE in DM	GE in DM	XP in DM
	kg kg-1	MJ kg-1	MJ kg-1	MJ kg-1	MJ kg-1	kg kg-1
grass (pasture)	0.19	10.6	6.35	14.1	18.45	0.19
grass silage	0.35	10.2	6.15	12.55	17.94	0.16
grass silage DLG 1	0.35	10.2	6.0	12.9	18.2	0.16
grass silage DLG 2	0.35	10.4	6.3	13.4	18.5	0.16
maize silage	0.27	10.95	6.6	12.45	18.00	0.08
maize silage DLG 1	0.27	10.8	6.5	13.4	18.5	0.08
maize silage DLG 2	0.27	11.0	6.7	13.7	18.5	0.08
straw (barley)	0.86	6.4	3.5	8,62	18.20	0.04
barley (grain)	0.88	12.9	8.2	15.5	18.6	0.119
wheat (grain)	0.88	11.6	7.5	16.36	18.52	0.121
rape seed expeller	0.90	12.5	7.5	15.2	20.3	0.396
soya expeller	0.91	12.1	7.6	16.2	22.5	0.440
sugar beet shreds	0.90	11.9	7.4	13.8	18.2	0.099
concentrate MLF 18/3	0.88	10.8	6.7	15.57	18.86	0.180

6.1.1.3 Annual durations of the lactation and dry periods

The calculation procedures make use of annual milk yields and annual energy requirements which are related to the durations of the lactation and dry periods per year. These data are obtained from the respective absolute durations as shares of the year:

$$t_{\mathrm{lact}} = \frac{t^*_{\mathrm{lact}}}{t^*_{\mathrm{lact}} + t^*_{\mathrm{dry}}} \cdot \alpha \qquad (51)$$

$$t_{\mathrm{dry}} = \frac{t^*_{\mathrm{dry}}}{t^*_{\mathrm{lact}} + t^*_{\mathrm{dry}}} \cdot \alpha \qquad (52)$$

where

- t_{lact} duration of the lactation period (d a^{-1})
- t_{lact}* absolute duration of the lactation period (d)
- t_{dry}* absolute duration of the dry period (d)
- α time units conversion factor (α = 365 d a^{-1})
- t_{dry} duration of the dry period (d a^{-1})

The absolute duration of the lactation period is the difference of the absolute durations of the interval between calvings and the dry period. The duration between calvings can be treated as a function of the annual milk yield (see e.g. Ferguson, 1996; Seeland and Henze, 2003; and literature cited therein). The German statistical data available illustrate the trend (Figure 2), from which Equation (53) is derived:

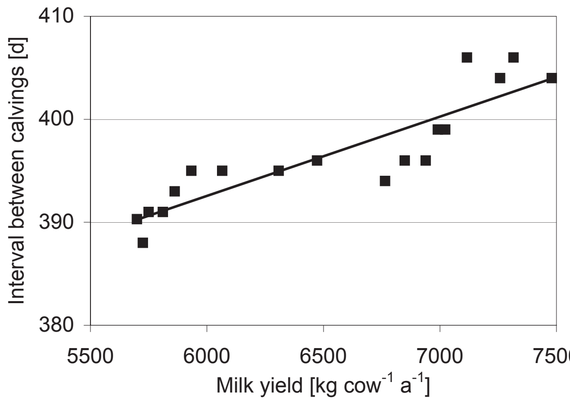

Figure 2:

Relation between mean intervals between calvings (data extracted from ADR 1992 ff) and mean annual milk yield (all herdbook cows). R^2 = 0.74

$$t^*_{\mathrm{lact}} = t^*_{\mathrm{ibc}} - t^*_{\mathrm{dry}} = \left(a + b \cdot Y_{\mathrm{M}}\right) - t^*_{\mathrm{dry}} \qquad (53)$$

where

- t_{lact}* absolute duration of the lactation period (d)
- t_{ibc}* absolute duration of interval between calvings (d)
- t_{dry}* absolute duration of the dry period (d)
- a constant (a = 346.4 d)
- b coefficient (b = 0.00769 d kg^{-1} cow a)
- Y_{M} annual milk yield (kg cow^{-1} a^{-1}) as reported in official statistics

Values for a and b were obtained for the regression line in Figure 2.

Durations of the dry period normally range from 40 d to 60 d. High performance cows are likely not to conceive as easily as medium performance cows. However, no relation could yet be established to relate this parameter to animal performance. The actual duration of the dry period t_{dry}*is assumed here to be 42 d (reflecting the data provided in GfE, 2001, pg. 23). The annual duration of the dry period t_{dry} is calculated as in Equation (52).

6.1.2 Comparison of energies

6.1.2.1 Energy requirements for maintenance

A direct comparison between CDC09 and IPCC or DIAS is impossible as the units used are not compatible.

6.1.2.2 Total energy requirements

The IPCC methodology is based on NE intakes throughout. If the feeding occurs according to the energy requirements, the amount of net energy required (NE_{tot}) has to equal the net energy intake with feed (NE_{feed}).

$$NE_{feed} = NE_{tot} \tag{54}$$

$$NE_{tot} = \alpha \cdot \left(ne_m + ne_f + ne_{lc} + ne_d + ne_p + ne_g \right) \tag{55}$$

where

NE_{tot} net energy required (MJ cow^{-1} a^{-1} NE)
α time units conversion factor (α = 365 d a^{-1})
ne_m net energy required for maintenance (MJ cow^{-1} d^{-1} NE)
ne_f net energy needed to obtain food (MJ cow^{-1} d^{-1} NE)
ne_{lc} net energy for lactation (MJ cow^{-1} d^{-1} NE)
ne_d net energy required for draft power (MJ cow^{-1} d^{-1} NE)
ne_p net energy required for pregnancy (MJ cow^{-1} d^{-1} NE)
ne_g net energy consumed for growth (MJ cow^{-1} d^{-1} NE)

6.1.2.3 Net energy for maintenance

In principle, IPCC (2006), pg. 10.15, uses the same approach as GfE (2001) to quantify NE_m:

$$ne_{m,\,IPCC} = a_{IPCC} \cdot w_{unit} \cdot \left(\frac{w}{w_{unit}} \right)^{0.75} \tag{56}$$

where

ne_m net energy required for maintenance (MJ cow^{-1} d^{-1} ME)
a_{IPCC} constant (b_{IPCC} = 0.386 MJ kg^{-1} d^{-1} ME)
w_{unit} animal weight unit (w_{unit} = 1 kg cow^{-1})
w animal weight (kg cow^{-1})

However, the coefficients vary. The GfE methodology results in an ne_m that is about 25 % lower than the results obtained with the IPCC approach. The application of the coefficient a_{CDC09} (see Chapter 3.2.1) leads to values larger than predicted by IPCC (2006).

6.1.2.4 Net energy to obtain feed

As the IPCC (2006) approach (see Equation (3)) is used in the CDC09 methodology as well, no differences occur.

6.1.2.5 Net energy for lactation

The only net energy entity that can be compared between the CDC09 (i.e. GfE) and IPCC (2006) methodologies is the net energy for lactation.

IPCC (2006), pg. 10.18, relates the energy used for lactation to milk yield and milk fat contents:

$$ne_{l,\,IPCC} = y_m \cdot \left(c_{lact\,1,\,IPCC} + c_{lact\,2,\,IPCC} \cdot x_{fat} \right) \tag{57}$$

where

$ne_{l,\,IPCC}$ net energy for lactation (MJ cow^{-1} d^{-1} NE)
y_m milk yield (kg cow^{-1} d^{-1})
$c_{lact\,1,\,IPCC}$ constant ($c_{lact\,1,\,IPCC}$ = 1.47 MJ kg^{-1})
$c_{lact\,2,\,IPCC}$ coefficient ($c_{lact\,2,\,IPCC}$ = 40 MJ kg^{-1})
x_{fat} mass fraction of fat (kg kg^{-1})

In principle, the results obtained from both approaches are similar (Figure 3). The national approach yields larger energy requirements. Differences between the two procedures are increasing with time.

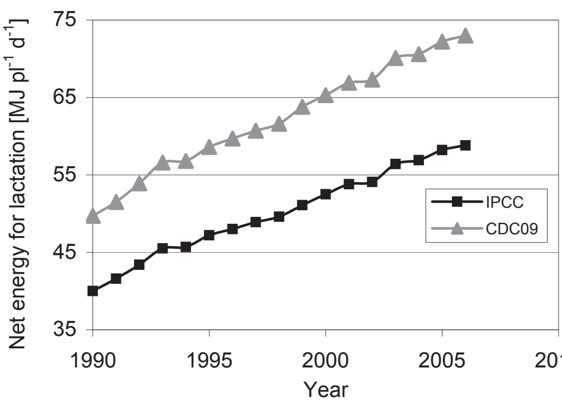

Figure 3:

Time series of the net energy requirements for lactation as calculated using the IPCC (2006) and the CDC09 methodologies

However, CDC09 allows to investigate the option of modified fat and increased protein contents on trace gas emissions (e.g. Brade et al., 2008).

6.1.2.6 Net energy for pregnancy

The IPCC methodology recommends to calculate the net energy for pregnancy proportional to the maintenance energy (IPCC, 2006, pg. 10.20).

$$ne_{\mathrm{p,\,IPCC}} = c_{\mathrm{preg}} \cdot ne_{\mathrm{m}} \qquad (58)$$

where

$ne_{\mathrm{p,\,IPCC}}$ net energy required for pregnancy (MJ $\mathrm{cow^{-1}\,d^{-1}}$)
c_{preg} coefficient for pregnancy (c_{preg} = 0.10; IPCC(2006)-10.20, Table 10.7)
ne_{m} net energy required for maintenance (MJ $\mathrm{cow^{-1}\,d^{-1}}$)

The IPCC and the CDC09 methodologies differ in principle, as they relate the energy requirements for pregnancy to different entities. The results obtained are quite different (see Figure 4). This is because with increasing mean live weights of cows in Germany, the IPCC methodology suggests increasing energy requirements for pregnancy. However, in the CDC09 method, the simultaneously increasing intervals between calvings indicate a decrease in energy requirements per cow and year.

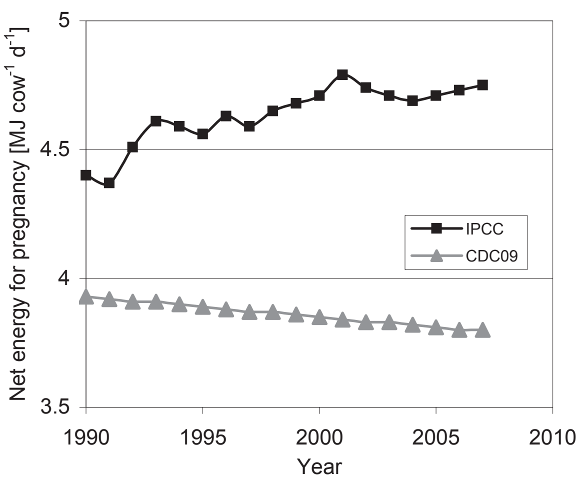

Figure 4:

Net energy requirements for pregnancy as calculated using the IPCC (2006) and the CDC09 methodologies. (IPCC based on mean German cow weights; CDC09 relying on numbers of births per cow and year)

The time series of calf weights used by ADR (1992 to 2007) differentiates between various breeds, but does not exhibit any changes between 1992 and 2007.

Neither IPCC (2006) nor CDC09 relate $NE_{\mathrm{p,\,IPCC}}$ or $NEL_{\mathrm{p,\,Ger}}$ to a calf weight. IPCC (2006) does not consider the number of births per year; it also fails to give a reference cow weight, for which the relation was designed.

6.1.2.7 Net energy for growth

The IPCC methodology relates the net energy for growth to a modified metabolic weight and the weight gained (IPCC(2006)-10.17):

$$ne_{\mathrm{g,\,IPCC}} = a \cdot w_{\mathrm{unit}} \cdot \left(\frac{w}{b \cdot w_{\mathrm{unit}}}\right)^{\mathrm{c}} \cdot \left(\frac{\Delta w}{\alpha \cdot w_{\mathrm{unit}}}\right)^{\mathrm{d}} \qquad (59)$$

where

$ne_{\mathrm{g,\,IPCC}}$ NE required for growth (MJ $\mathrm{cow^{-1}\,d^{-1}}$ NE)
a constant (a = 22.02 MJ $\mathrm{kg^{-1}}$ NE)
w mean animal weight of the population ($a$kg $\mathrm{cow^{-1}}$)
b constant (b = 0.8)
w_{unit} animal weight unit (w_{unit} = 1 kg $\mathrm{cow^{-1}}$)
c exponent (c = 0.75)
Δw weight gain (kg $\mathrm{cow^{-1}\,a^{-1}}$)
α time units conversion factor (α = 365 d $\mathrm{a^{-1}}$)
d exponent (d = 1.097)

The explanations given in IPCC (2006), pg. 10.17 suggest that this procedure is to be applied to growing animals rather than to dairy cattle with a moderate weight gain rate.

Thus, CDC09 uses the GfE (2001) approach instead as described in Equation (8).

6.1.3 Dry matter intake and related entities

CDC09 relates the intakes of GE, DE, ME, ash and nitrogen to the amount of DM taken in with the feed components using the following equations:

$$GE = \sum_{\mathrm{i=1}}^{\mathrm{n}} DM_{\mathrm{i}} \cdot X_{\mathrm{GE,\,i}} \qquad (60)$$

$$DE = \sum_{\mathrm{i=1}}^{\mathrm{n}} DM_{\mathrm{i}} \cdot X_{\mathrm{DE,\,i}} \qquad (61)$$

$$ME = \sum_{\mathrm{i=1}}^{\mathrm{n}} DM_{\mathrm{i}} \cdot X_{\mathrm{ME,\,i}} \qquad (62)$$

$$m_{\mathrm{feed}} = \sum_{\mathrm{i=1}}^{\mathrm{n}} DM_{\mathrm{i}} \cdot X_{\mathrm{XP,\,i}} \cdot X_{\mathrm{N}} \qquad (63)$$

where

GE	gross energy intake (MJ cow^{-1} a^{-1} GE)
DM_i	amount of DM taken in with component i (kg a^{-1})
$X_{GE, i}$	GE content of feed component i (MJ kg^{-1} GE)
$X_{DE, i}$	DE content of feed component i (MJ kg^{-1} DE)
$X_{ME, i}$	ME content of feed component i (MJ kg^{-1} ME)
m_{feed}	amount of nitrogen intake (kg cow^{-1} a^{-1} N)
$X_{XP, i}$	crude protein content of feed component i (kg kg^{-1} XP)
X_N	nitrogen content of crude protein (X_N = 1/6.25 kg kg^{-1} N)

The IPCC (2006) methodology does not provide an approach to calculate DM intakes for the high quality forages used in Germany (IPCC, 2006, pg. 10.22). Thus, a direct comparison is impossible.

6.1.4 Total gross energy requirements

According to IPCC (2006), pg 10.21, GE is assessed as follows:

$$ge_{IPCC} = \left(\frac{ne_m + ne_f + ne_{lc} + ne_d + ne_p}{\left\{\frac{ne}{de}\right\}} + \frac{ne_g}{\left\{\frac{ne_g}{de}\right\}} \right) \cdot \frac{1}{X_{DE}} \qquad (64)$$

where

ge_{IPCC}	gross energy intake (MJ cow^{-1} d^{-1} GE)
ne_m	net energy required for maintenance (MJ cow^{-1} d^{-1})
ne_f	net energy needed to obtain food (MJ cow^{-1} d^{-1} NE)
ne_{lc}	net energy for lactation (MJ cow^{-1} d^{-1})
ne_d	net energy required for draft power (MJ cow^{-1} d^{-1} NE)
ne_p	net energy required for pregnancy (MJ cow^{-1} d^{-1} NE)
ne	net energy consumed for maintenance, lactation, work and pregnancy (MJ cow^{-1} d^{-1} NE)
de	digestible energy (MJ cow^{-1} d^{-1} DE)
ne_g	net energy consumed for growth (MJ cow^{-1} d^{-1} NE)
X_{DE}	mean digestible energy as fraction of gross energy (MJ MJ^{-1})

The fractions in braces ({}) are calculated according to Equations (65) and (66) (IPCC (2006) pg 10.20, eq. 10.14;. IPCC (2006) pg 10.21, eq. 10.15).

$$\left\{\frac{ne}{de}\right\} = 1.123 - 0.4092 \cdot X_{DE} + 0.1125 \cdot X_{DE}^2 - \frac{0.254}{X_{DE}} \qquad (65)$$

where

ne	net energy consumed for maintenance, lactation, work and pregnancy (MJ cow^{-1} d^{-1})
de	digestible energy (MJ cow^{-1} d^{-1})
X_{DE}	digestible energy expressed as fraction of GE (MJ MJ^{-1})

$$\left\{\frac{ne_g}{de}\right\} = 1.164 + 0.516 \cdot X_{DE} + 0.1308 \cdot X_{DE}^2 - \frac{0.374}{X_{DE}} \qquad (66)$$

where

ne_g	net energy required for weight gain (MJ cow^{-1} d^{-1})
de	digestible energy (MJ cow^{-1} d^{-1})
X_{DE}	digestible energy expressed as fraction of GE (MJ MJ^{-1})

As illustrated in Figure 5, CDC09 calculates GE intakes that exceed those modelled in the IPCC approach for milk yields for all milk yields due to increased nel_m and nel_{lc}. The two diets yield very similar amounts of GE.

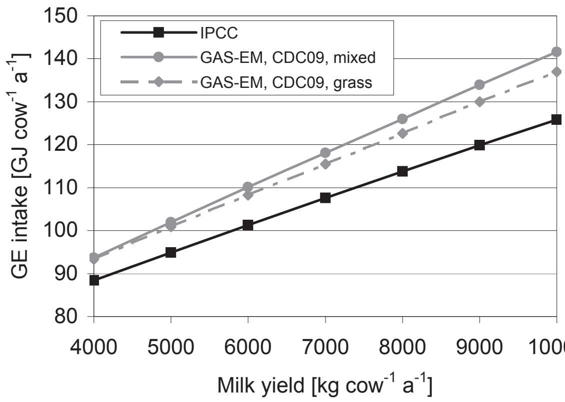

Figure 5:

Gross energy intake as calculated with the IPCC and the CDC09 modules for different annual milk yields

6.1.5 Methane emissions from enteric fermentation

The CH_4 emission from enteric fermentation is derived from the gross energy intake and the methane conversion factor as follows:

$$EF_{\mathrm{CH4,\,ent}} = ge \cdot \frac{x_{\mathrm{CH4}} \cdot \alpha}{\eta_{\mathrm{CH4}}} \qquad (67)$$

where

$EF_{\mathrm{CH4,\,ent}}$	emission factor for CH_4 from enteric fermentation (kg cow^{-1} a^{-1} CH_4)
ge	gross energy intake (MJ cow^{-1} d^{-1})
x_{CH4}	methane conversion factor (MJ MJ^{-1})
α	time units conversion factor (α = 365 d a^{-1})
η_{CH4}	energy content of methane (η_{CH4} = 55.65 MJ (kg CH_4)$^{-1}$)

Whereas x_{CH4} is a constant in the IPCC approach, it varies with feed composition in CDC09. As shown in Figure 6, methane conversion factors used in CDC09 fall below the IPCC default values. This was also observed in detailed Dutch and US studies (Smink et al., 2005; Kebreab et al., 2008). However, CDC09 methane conversion factors are in the range proposed in IPCC (2006) pg.10.30 for countries with high quality feed.

Conversion factors for mixed and grass based diets are almost identical.

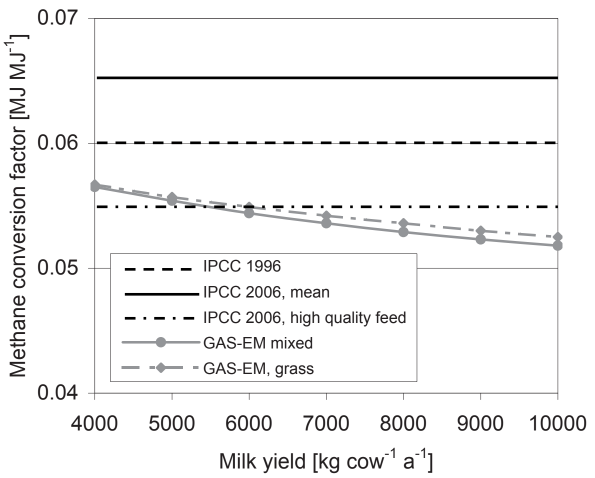

Figure 6:

Methane conversion factors as calculated with the IPCC and the CDC09 modules for different annual milk yields. The CDC09 lines for mixed and grass are almost identical.

As a consequence, the results obtained for CH_4 emission factors for enteric fermentation differ considerably (Figure 7).

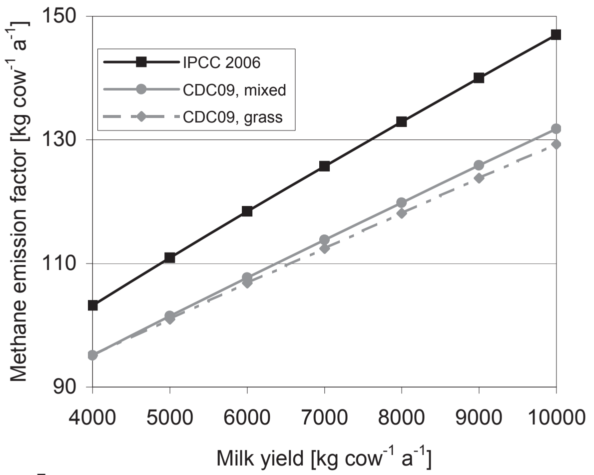

Figure 7:

Methane emission factors as calculated with the IPCC and the CDC09 modules for different annual milk yields. The CDC09 lines for mixed and grass are almost identical.

The difference observed is in the order of magnitude of 10 %. Emission factors of the mixed and grass silage based diets are almost identical.

6.1.6 Excretion of volatile solids

IPCC and the CDC09 approaches are identical in principle. However, CDC09 uses different constants.

As discussed in Chapter 6.1.4, **GE intakes** calculated with the IPCC and CDC09 approaches deviate.

IPCC (2006), pg. 10.14, recommends a value for the **digestibility** X_{DE} in the range of 0.55 to 0.75. Figure 8 illustrates that the feed quality in German diets is close to the upper limit suggested in IPCC (2006).

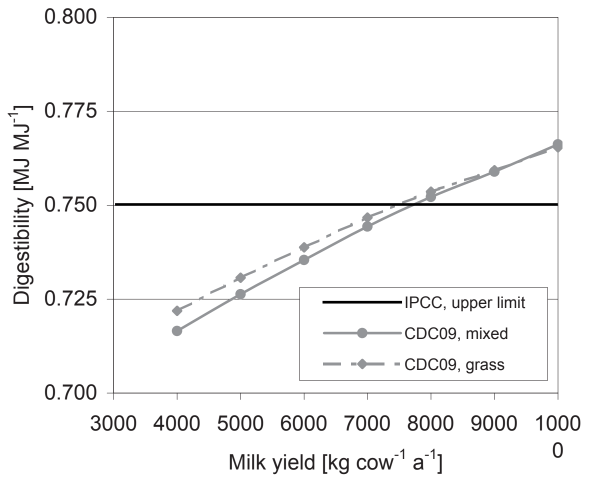

Figure 8:

Mean digestibilities as proposed by IPCC (2006) and as calculated with the CDC09 module for different annual milk yields

In Figure 9, VS excretions rates calculated according to IPCC use a mean digestibility of 0.65 MJ MJ^{-1}.

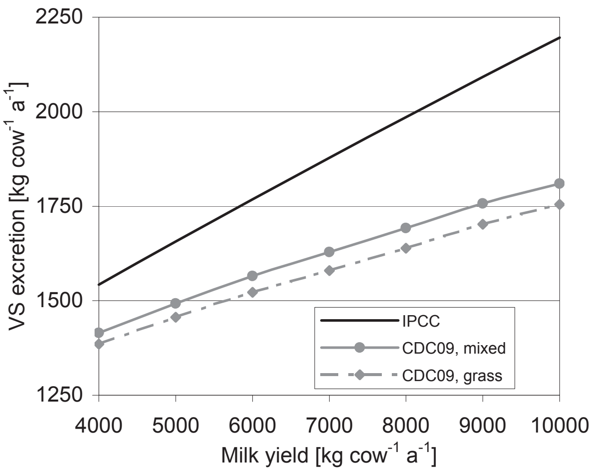

Figure 9:

VS excretion rates as calculated according to IPCC (2006) and the CDC09 module for different annual milk yields

For all milk yields, CDC09 VS excretion rates deviate significantly from IPCC.

In principle, CH_4 emissions from manure management should not be related to VS as in Equation (47), but to its degradable portion (see Sommer et al., 2004). In practice, it is assumed that this fraction is constant. At present there is no proposal to model its amount.

6.2 Comparison with the DLG (2005) dataset

DLG (2005) describes a consistent data set of feed, feed composition, properties of feed constituents and N excretion rates for cows with annual milk yields of 6000, 8000 and 10000 kg cow^{-1} a^{-1} ECM. Animal weight and weight gain are not mentioned. Hence, this comparison assumes a mean weight of 630 kg cow^{-1} and a weight gain of 20 kg cow^{-1} a^{-1}.

The authors of DLG (2005) communicated the feed constituent properties (Obermaier and Spiekers, 2008). They are listed in Table 3. For milk yields of 6000 and 8000 kg cow^{-1} a^{-1} ECM grass silage DLG 1 and maize silage DLG 1 are used, for yields of 10000 kg cow^{-1} a^{-1} ECM grass silage DLG 2 and maize silage DLG 2.

6.2.1 Daily milk yield and energy corrected milk yield

DLG (2005) relates dry matter intake and nitrogen excretion rates to energy corrected daily milk yields.

The daily milk yield is related to the annual duration of the lactation period according to Equation (68):

$$y_M = \frac{Y_M}{t_{lact}} \tag{68}$$

where

y_M daily milk yield, uncorrected (kg cow^{-1} d^{-1} milk)
Y_M annual milk yield (kg cow^{-1} a^{-1} milk)
t_{lact} duration of the lactation period (d a^{-1})

Energy corrected milk yield considers the fat and protein contents of the milk as in Equation (69):

$$y_{ECM} = y_M \cdot \left(a + b \cdot x_{fat} + c \cdot x_{protein}\right) \tag{69}$$

where

y_{ECM} daily milk yield (energy corrected) (kg cow^{-1} d^{-1})
y_M daily milk yield, uncorrected (kg cow^{-1} d^{-1} milk)
a constant (a = 0.3246)
b coefficient (b = 12.86)
x_{fat} mass fraction of fat (kg kg^{-1})
c coefficient (c = 7.04)
$x_{protein}$ mass fraction of protein (kg kg^{-1})

6.2.2 Dry matter intake

6.2.2.1 Total dry matter

The IPCC (2006) methodology does not provide an approach to calculate DM intakes for the high quality forages used in Germany (IPCC, 2006, pg. 10.22).

DM intakes as modelled with CDC09 equal those provided by DLG (2005) (Figure 10). However, DLG (2005) report one digit only with their DM inputs, so differences up to 0.2 Mg cow^{-1} a^{-1} may occur.

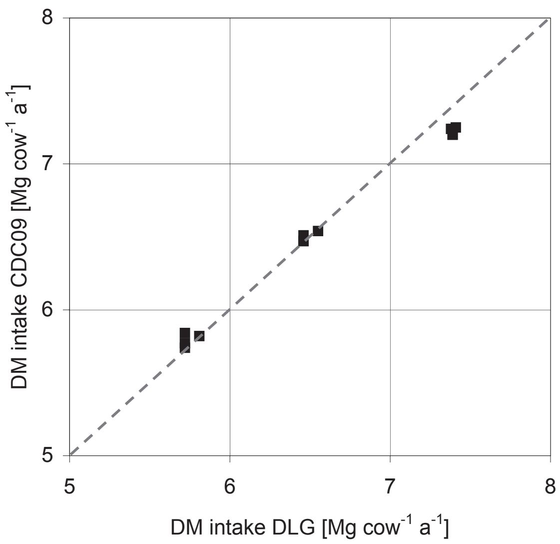

Figure 10:

DM intake for 12 DLG scenarios compared with results of the CDC09 module (broken line: 1 to 1 line)

6.2.2.2 Dry matter with concentrates

The distribution between concentrates and roughage of dry matter intake is decisive for the energy and nitrogen intakes. Figure 11 illustrates that CDC09 and DLG (2005) results deviate slightly, but systematically. Again, as DLG (2005) report one digit only with their DM inputs, differences up to 0.2 Mg cow^{-1} a^{-1} may also apply to concentrates.

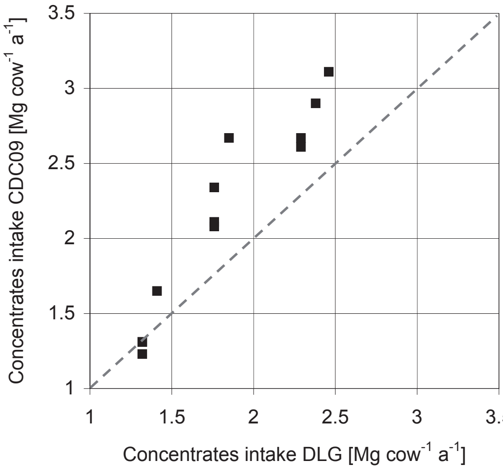

Figure 11:

DM intake with concentrates for 12 DLG scenarios compared with results of the CDC09 module (broken line: 1 to 1 line)

6.2.3 N excretion rates

DLG (2005) provides an independent data set that can be used for a validation of the module. A comparison with CDC09 results is shown in Figure 12. As a whole, CDC09 estimates N excretion rates about 4 % larger than DLG (2005).

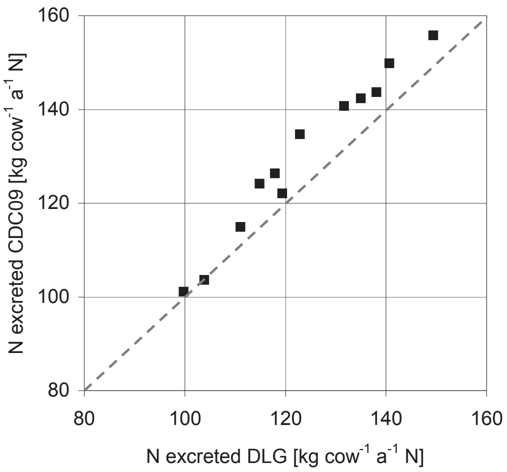

Figure 12:

N excretion rates of 12 DLG scenarios and corresponding results obtained with CDC09

7 Conclusions

The goal to develop a single model from national standard descriptions that allows the assessment of excretions and emissions from a single data set was achieved. The bottom-up CDC09 model agrees well with the top-down DLG calculations based on measurements.

The model can be used to quantify N, TAN and VS excretion as well as CH_4 emissions from enteric fermentation if the amount and composition of feed are known. The model can predict feed intake, if performance data are known adequately. It can thus be a submodel in calculations within a life cycle analysis.

Hence, CDC09 is considered a useful tool for the construction of emission inventories as well as for policy advice.

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Eine kritische Analyse von Ruschs Theorien zur Bodenfruchtbarkeit als Grundlage für die Bodenbewirtschaftung im Ökologischen Landbau

Hans Marten Paulsen*, Stefan Schrader** und Ewald Schnug***

Zusammenfassung

Auf die in den 50er und 60er Jahren publizierten Theorien zur Bodenfruchtbarkeit von Hans Peter Rusch wird im organisch-biologischen Landbau heute noch Bezug genommen. Zu nennen sind hier der Rusch-Test zur Einstufung der biologischen Qualität der organischen Substanz und von Böden, die Systematisierung des Abbaus organischer Substanz im Boden und der Humusbildung sowie die Flächenkompostierung. Ruschs Lehren zielen auf eine Verbesserung der Bodengare durch effiziente Zufuhr organischer Substanz und Humusbildung sowie auf die Verbesserung der Ausnutzung von Nährstoffen. Gegen den Geist seiner Zeit grenzte sich Rusch von der Mineraldüngung ab und hebt die auch heute unbestrittene Rolle des Humus für die Bodenfruchtbarkeit hervor. Die tatsächliche Bedeutung von Ruschs Nachweismethoden zur Charakterisierung der Abbaubarkeit von Materialien und von mikrobiellen Kennziffern im Boden für die Landwirtschaft ist vermutlich gering und wissenschaftlich nicht belegt. Sein systematischer Ansatz zum Abbau organischer Substanz im Boden, zur Humusbildung und seine daraus abgeleiteten Konsequenzen für das Management von Wirtschaftsdüngern und anderen in der Landwirtschaft anfallenden organischen Materialien sind auch nach heutigen Kenntnisstand nicht zu widerlegen, erfordern aber eine Überprüfung.

Schlüsselwörter: Rusch-Test, Bodenfruchtbarkeit, organisch-biologischer Landbau, Bodengare, Humusqualität, Bodenleben, Flächenkompostierung, Hans Peter Rusch

Abstract

A critical assessment of the Rusch theory on soil fertility as basis for soil management in Organic Farming

Organic-biological agriculture still draws on the theories on soil fertility published by the German scientist Hans Peter Rusch in the 1950s and 1960s, namely the Rusch-Test determining the biological quality of organic matter and soils, the decomposition of organic material in soils to humus and the surface-composting. Rusch's teachings address the advancing of soil tilth by efficient application of organic matter and humus formation and an improved nutrient use. Rusch strictly rejected the upcoming mineral fertilization and emphasized the role of humus for soil quality, which is even today beyond debate. Rusch's methods to characterize degradability of organic materials in soil and his microbial parameters are of little importance in practical agriculture and they have not been scientifically proven. However, his systematic approach to the decomposition of organic materials in soil, to humus formation and his consequences for the management of farmyard manure and other sources of organic matter can not be refuted with today's knowledge, but still need to be validated.

Keywords: Rusch-test, soil fertility, organic-biologic agriculture, soil tilth, humus quality, soil life, surface-composting, Hans Peter Rusch

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1 Einleitung

Der Ökologische Landbau adressiert in seinen Bewirtschaftungsregeln die Bedeutung natürlicher Bodenfruchtbarkeit und die Notwendigkeit diese zu erhalten und zu fördern. Kernpunkte für die Ausgestaltung der Nährstoffversorgung der Pflanzen sind dabei die Minimierung des Verbrauchs nicht erneuerbarer Ressourcen und von außerbetrieblichen Produktionsmitteln, die Nutzung von Ressourcen in lokal organisierten landwirtschaftlichen Systemen, die Nährstoffbereitstellung aus dem Boden sowie die Rückführung von Abfällen und Nebenerzeugnissen pflanzlichen und tierischen Ursprungs (Durchführungs-Verordnung (EG) 889/2008; Öko-Basisverordnung (EG) 834/2007).

Der organisch-biologische Landbau ist eine Form des Ökologischen Landbaus, die sich aus den Überlegungen von Hans Müller (1891 bis 1988) und Hans Peter Rusch (1906 bis 1977) entwickelte (Lockeretz, 2007). Rusch hielt unter anderem Vorträge über die „Erhaltung und den Kreislauf lebendiger Substanz“ (Rusch, 1955), verfasste ein Buch über die Humuswirtschaft und den Nachweis von Bodenfruchtbarkeit (Rusch, 1964) sowie eine zusammenfassende und ergänzende Schrift über die Bedeutung seiner Methoden und Erkenntnisse für die Praxis des organisch-biologischen Landbaus (Rusch, 1970). Bis heute wird in den Bioland-Richtlinien auf die Lehren von Rusch als Grundlage für die Bodenbewirtschaftung Bezug genommen (Bioland Richtlinien, 2007). Jedoch ist die Kenntnis über die Inhalte heute weitgehend in Vergessenheit geraten. Im Folgenden sollen daher die Inhalte der Lehren von Rusch wiedergegeben und in den heutigen Forschungstand eingeordnet werden.

1.1 Ruschs Leitlinien und Grundüberzeugung

> *„Die Resultate einer planmäßig gelenkten Humuswirtschaft sind einmalig und unerhört. Für den durchschnittlichen Landbaufachmann sind sie nicht nur erstaunlich, sondern vollends unglaubhaft. .... Die Umgestaltung des Landbaues unter dem Einfluss der klassischen Naturwissenschaft ist so tiefgehend, dass man das Wirken natürlicher Kräfte nicht mehr zu sehen bekommt und vergessen hat“* (225)[1].

Dieses Zitat zeigt die tiefe Überzeugung des Arztes Hans Peter Rusch, dass seine Erkenntnisse zur Humuswirtschaft, zur Rückbesinnung auf die natürlichen Kräfte, zur Herstellung von biologischen Gleichgewichten in der Bodenbewirtschaftung nicht zu widerlegen seien. Ihm ist bewusst, dass sie im Lichte der 60er Jahre, in denen die Agrarchemie und Düngungslehre einen enormen Aufschwung hatten, geradezu revolutionär erscheinen. Er war daran interessiert Produktivität und Ertragssicherheit der Landwirtschaft zu steigern. Seine Vision einen großflächigen Bodenaustausch zwischen leichten und schweren Standorten zum Abgleich der Tongehalte von Böden durchzuführen (223) zeugt von der Aufbruchsstimmung seiner Zeit.

Als Leitlinie der Gedanken von Rusch kann so etwas wie: „Die Natur erhält sich selbst“ gesehen werden. Jedoch ist Rusch hier rational. Therapiemaßnahmen im akuten Fall bleiben für ihn sinnvoll. Er löst diesen Widerspruch wie folgt: *„Behandelt man nur das Symptom an sich* ... *fällt ...* man ... *dem Organismus damit in den Arm, im gleichen Moment, da er sich anschickt, die eigentliche Krankheit zu überwinden. Gewiss – Nothilfe muss gegeben werden, sobald es sich nicht um eine erfolgreiche Abwehrschlacht des Körpers handelt ...“* (21).

1.2 Ruschs Thesen zur Bodenfruchtbarkeit

In seinem Buch „Bodenfruchtbarkeit“ aus dem Jahr 1964 legt er zunächst seine Auffassung dar, dass es sich bei künstlicher Düngung nicht um normale physiologische Pflanzenernährung handelt und sie eine Verschiebung des mineralischen Gleichgewichts zur Folge hat (17, 72). Eine ausgewogene künstliche Düngung scheint ihm nicht möglich. Vor allem die mineralische Stickstoffdüngung (19) aber auch die Düngung mit aufgeschlossenen schnell wirksamen Düngern (223) widerspräche der natürlichen Bereitstellung der Nährstoffe aus dem Boden und sei daher abzulehnen. Bei jeder Anwendung von Pflanzenschutzmitteln und Medikamenten sei ungewiss, welche Veränderungen zum Negativen damit bewirkt werden (24). Es werde keine echte Heilung erzielt (20). Die Selbstheilungskräfte würden außer Kraft gesetzt und das natürliche System geriete aus dem Gleichgewicht. Massenhaftes Auftreten von Schädlingen wird von Rusch als Regulativ angesehen, um die Ordnung des Lebendigen, die das Lebende an sich ausmache (33) und die aus dem Gleichgewicht geraten ist, wieder herzustellen (34, 66). Die Konsequenz, den Erreger zu vernichten ändere nichts, wenn das Prinzip der Fehlfunktion nicht verändert werde (39). Er fordert ein biologisches Denken (35) und konstatiert, dass nur natürliche Systeme stabil sind und ihren Bedarf selbst decken. Rusch erkennt in der Natur ein gerichtetes, selbsterhaltendes System. Die Funktion lebendiger Abläufe dürfe nur an vollständigen biologischen Substanzkeisläufen dargestellt werden. In einem solchen *„Ganzheitsexperiment“* seien einzig die Gesundheit und Fruchtbarkeit eines jeden Gliedes der Kreislaufkette Testkriterien. Als Testorganismen zur Beurteilung des gesamten Systems seien nur

1 Kursiv gedruckter Text ist im Wortlaut Ruschs zitiert. Die Zahlen in Klammern geben die Seitenzahl des Zitats in der 6. Auflage von Ruschs Buch „Bodenfruchtbarkeit“ an (Rusch, 1964).

solche geeignet, die nachweislich mit allen Gliedern (Boden, Pflanze, Tier und Mensch) in Beziehung stehen (44). Als Beispiel nennt Rusch die Bakterienflora (42). Einzelne Bewertungsparameter, wie z. B. der Mineralstoffgehalt des Bodens seien für eine umfassende Bewertung der Bodenfruchtbarkeit nicht ausreichend (57). Fruchtbarkeit ist für ihn der Ausdruck gesunden Lebens und er postuliert, dass diese Fruchtbarkeit bereits im makromolekularen Bereich auftritt. Unter dem Eindruck der damaligen Entdeckung der DNS und der viralen Infektionen und Erbgutübertragung vermutet Rusch einen Kreislauf der lebendigen Substanzen (Rusch, 1960). Er meint, dass eine komplette Mineralisation bei der Autolyse organischen Gewebes im Boden nicht vorkomme. Vielmehr würden wertvolle Bestandteile der Organismen für den Neuaufbau „*lebendiger Gestalten*" genutzt (54). Als Beleg führt Rusch die Fähigkeit von Pflanzen an, makromolekulare Stoffe über die Wurzel aufzunehmen. Humus sei Träger des Kreislaufs lebender Substanz (59). Ausgangspunkt Ruschs sind die Vorgänge beim Substanzumbau von unfruchtbaren organischen Resten (Pflanzenreste, Komposte) zu fruchtbaren organischen Düngern im Organismus „*Muttererde*" (59). „*Humus ist kein Stoff, sondern ein Vorgang,... die Trümmer der Lebensabfälle aufs Neue harmonisch zu ordnen (75)... ein Ausdruck der tätigen Beziehung zwischen dem Mutterboden und allen anderen Organismen" (91).* Einzig die Bereitstellung der Pflanzennährstoffe aus den Umbauprozessen im Boden sei geeignet, Pflanzen harmonisch zu ernähren und vermindere die Schädlings- und Krankheitsanfälligkeit der Pflanzen (76) sowie Bodenmüdigkeit (77) im Sinne von Nährstoffmangel. Auch in seiner Schrift aus dem Jahr 1970 postuliert Rusch, dass bei Schädlings- und Krankheitsbefall meist der Boden „*schuld sei*" und dies mit seinen Tests zur Bodengüte (siehe unten) nachvollzogen werden könnte.

Rusch hielt Kontakt zu zahlreichen „Humusbauern" auf deren Betrieben intensiv organisch gedüngt wurde. Immer wieder werden von Rusch Stadtkomposte als wertvolle Dünger genannt (81), deren „*einwohnergleiche"* Aufbringung wichtiger Bestandteil der Sicherung und Steigerung der Bodenfruchtbarkeit ist. Wobei der Zustand organischer Dünger, ihre Leistungsfähigkeit, ihre Gesundheit und Brauchbarkeit von Zustand, Leistungsfähigkeit, Gesundheit und Brauchbarkeit der Organismen abhänge, die sie produziert haben (85). In einer späteren Arbeit (Rusch, 1970) stellt sich Rusch jedoch kritisch zur Verwendung von Stadtkomposten. Er beschreibt, dass seine Tests auf biologische Wertigkeiten hier aufgrund von Fremdstoffen oft schlechte Werte aufwiesen und verweist auf seine mikrobiellen Tests als Indikator für die Brauchbarkeit der Komposte.

An Definitionen zur Komplexität der biologischen Vorgänge im Boden schließen sich allgemeine bodenkundliche Aussagen zu Bodenaufbau und Chemie an. Genannt werden hier einige der auch heute gültigen zentralen Faktoren der Bodenkunde: Bodenart (92) für die Durchlüftung und Wasserhaltefähigkeit, Tonminerale (95) als Nährstoffspeicher und Puffer sowie Kationenaustauschkapazität und Austauscherbelegung (96). Die Verbindung zur lebenden Substanz werde durch die Bildung von Ton-Humus-Komplexen geschaffen (96).

Bodenfruchtbarkeit definiert Rusch als „*die optimale Fähigkeit der makromolekularen Bodenkräfte, die Ordnungen der Abfallsubstanzen des oberirdischen Lebens wiederherzustellen oder zu erhalten und sie der Pflanze zu gegebener Zeit, neu geordnet und wohl versehen mit der zuständigen Garnitur mikromolekularer Stoffe, zur Verfügung zu stellen (103). ... (Sie) kommt nur zustande als Ergebnis des lebendigen Ablaufes »Bodennahrung – Garebildung – Rhizosphäre – Pflanze«, sie ist keine materielle Größe sondern eine biologisch funktionelle Kapazität"* (104).

Im Kapitel „Das Wesen der Bodengare" (105) werden zunächst Huminstoffe als nicht mehr weiter abbaubare polare Substanzen definiert, die fest mit den mineralischen Bestandteilen des Bodens verbunden sein können und eine hohe Austauschkapazität aufweisen. Dadurch seien sie am Aufbau der Bodengare beteiligt. Jedoch bestehe die Bodengare selbst aus „*... Stütz- und Füllsubstanzen, in die alle mobilen Lebensvorgänge eingebettet sind (sie regelt) den Gasabtausch und Wasserhaushalt..."* (107).

In der oberen Bodenschicht (3 bis 8 cm) bildet sich nach Rusch beim Abbau organischer Substanz eine „*Zellgare"* oder „*mikrobielle Gare"* aus. Definiert ist diese durch die Anreicherung von Mikroorganismen, die die Primärzersetzungsprozesse der organischen Substanz im Boden durchführen. Voraussetzung für die Abbautätigkeit sei jedoch eine natürliche Bodendecke aus nicht verarbeitetem Material als Licht-, Austrocknungs- und Erosionsschutz, sowie für die Luftzufuhr (109). Fehle der Nachschub an organischer Substanz ginge die mikrobielle Gare zugrunde. Übrig bliebe eine „*zellulare Rest-Gare*" (109). Rusch zieht hieraus Konsequenzen für die organische Düngung und die dauerhafte Erhaltung einer Zellgareschicht: „Je vielseitiger die Düngung, umso besser für die Gare. ... Schwere Böden bedürfen einer anderen organischen Düngung als leichte. ... Es soll möglichst nicht zuviel auf einmal gedüngt werden und dies nicht nur einmal jährlich, sondern verteilt auf viele Einzelgaben" (110). Als geeignete Dünger werden in diesem Zusammenhang Stallmist, Kleintiermist, Jauche, Gülle, Unkraut, Gründüngung, Komposte, organische Handelsdünger, Algendünger und Stadtkomposte genannt (110).

Als Fruchtbarkeitsreserve des Bodens, als „*Alte Kraft"*, wird von Rusch die sogenannte „*plasmatische Gare oder makromolekulare Gare"* bezeichnet. Hier seien die typischen und sichtbaren Mikrobenzellen der Zellgare ebenfalls abgebaut und in Makromoleküle und subzellulare Substanzen zerlegt. Die plasmatische Gare sähe *„aus wie lockere feinkrümelige Erde,... und ist mit einer guten Lupe als lufthaltiges, feinspaltiges und feinporiges System erkennbar“* (112). Die Partikel bänden sich an die mineralischen Bodenbestandteile, hätten eine hohe kolloidale Kraft und ein hohes Wasser- und Ionen-Speichervermögen (119). Humus definiert Rusch wie folgt: *„Es entsteht einen Primitiv-Form lebenden Gewebes, eine Urform aus einer Kongregation mineralischer, organischer und lebendiger Substanz ... Das allein sollte man Humus nennen... ein Gewebe in dem die Pflanze lebt“* (112). Rusch merkt an, dass die Pflanzennahrung nicht nur aus der plasmatischen Gare komme, sondern auch schon aus der zellularen Gare von der Pflanze aufgenommen werden könne (115). Weiterhin hebt er die bedeutende Rolle der Bodengare für die Wasser- und Luftkapazität von Böden hervor und schließt, wie auch bei der Nährstofffreisetzung (116) mit der Feststellung, *„dass mit allen anderen biologischen Funktionen auch der Wasserhaushalt nur dann optimal funktioniert, wenn wir uns jeden künstlichen Eingriffs in den Bodenstoffwechsel enthalten“ (123).*

1.3 Die Bestimmung der Bodenfruchtbarkeit nach Rusch

Rusch fordert für die Bestimmung der biologischen Wertigkeit von Böden einen Test, der anhand von Lebensvorgängen, die im Rahmen eines intakten biologischen Substanzkreislaufs ablaufen und anhand eines leicht greifbaren Materials in kurzer Zeit ohne großen Aufwand ein zutreffendes Ergebnis liefern kann (127). Er hält die klassifizierende Bodenmikrobiologie für zu vielfältig (128, 129) und favorisiert mikrobielle Zellen als Summenparameter zur Beurteilung der Fruchtbarkeit von Böden (133) *„Allerdings entsteht nur dort Fruchtbarkeit, wo im Verlaufe der aktiven Bodenperioden ein- oder mehrere Male hohe Zellzahlen aufgetreten sind“ (136).* Für die Praktikabilität eines Bodentests müsse *„...die direkte Zellzählung ... so gestaltet werden, dass sie unabhängig von Bedingungen, die nur temporär vorhanden sind, unabhängig besonders von der Jahreszeit ist“* (136).

In seinen Untersuchungen konnte Rusch durch Bebrütung Wachstumsbedingungen herstellen, die zu konstanten Zählergebnissen bei den Bodenmikroben führten (136). *„Im Prinzip benutzt man ... die Bodenprobe als bakteriologischen Nährboden ohne besondere Zusätze ... und erhält so ein relatives Vergleichsmaß für diejenige Masse an organischer und mineralischer Substanz, die unter Wachstumsbedingungen für die spontan vorhandenen Mikrobien-Gesellschaft umsetzbar ist“* (136).

Für die Bewertung der biologischen Qualität der Böden schlägt er Verfahren vor, die die *„zellulare Gare“* und die *„plasmatische Gare“* in Zahlen fassen (Tabelle 1). Weiterhin führt er ein Verfahren ein, mit dem die biologische Qualität des umsetzbaren Materials durch Zählungen der bei Bebrütung entstehenden unterschiedlichen Bakteriengruppen eingestuft wird (Tabelle 2).

Tabelle 1:

Der Rusch-Test: Bestimmung der Zellzahlen (Rusch, 1964)

Bestimmung der Zellzahl ① (138) (zellulare Gare)

„Sie stellt ein Maß für den Umfang der zellularen Gare dar und daher auch ein Maß für vorausgegangene organische Düngungen... sie... zeigt an welches Maß die Fruchtbarkeit in 1 oder 2 Jahren erreichen wird“ (140).

Methode: 10 g Boden (20 g Kompost) in 50 ml (100 ml) physiologischer Kochsalzlösung schütteln, und bei 27 °C bebrüten. Anschließend erneut schütteln und nur 1 bis 2 ml filtrieren (Papierfilter Nr. 588). Bei 6-facher Vergrößerung (Objektiv 45:1) mikroskopieren. Zellzahlen in Thomakammern auszählen. Wenn der Wert > 25 ist 1:10 verdünnen, sonst ungenaue Zählung.

Bestimmung der Zellzahl ② (156) (Plasmagare)

Sie ist nach Rusch *„tatsächlich Ausdruck und Maß der Bodenfruchtbarkeit“* (156)

Die plasmatische Gare wird von Rusch wie folgt charakterisiert:

„...beim totalen Abbau der organischen Bodennahrungen (entstehen) drei unterschiedliche Stoffgruppen:
1. Mikromolekulare Stoffe (Nährstoffe: An- und Kationen, Wirkstoffe – Vitamine, Hormone, Enzyme und andere –, Eiweißbausteine, Wasser, Gase),
2. Huminstoffe (umgeordnete, unverdauliche Makromolekularsubstanzen, ähnlich den Farbstoffen, die andere Organismen abscheiden),
3. Makromolekulare Stoffe (verdauliche und spezifische lebendige Bodensubstanz)
Diese drei Stoffgruppen, die alle aus dem biologischen Substanzkreislauf stammen, bilden zusammen mit dem örtlich gegebenen Mineral in seinen bodenbildenden Formen die plasmatische Gare aus.“

Die Zellzahl ② enthält auch die Werte der Zellzahl ① (157) und ist damit stets höher.
Methode: Aufschluss wie in ① + 0,1 g (1 g) Laktose-Dextrose-Mischung 48 h bei 27 °C bebrüten, filtrieren und auszählen.

Tabelle 2:

Rusch-Test: Die Bestimmung der biologischen Qualität (Rusch, 1964)

Die Bestimmung der biologischen Qualität (Technik ③ und ④) (189)

Methode: Suspension aus ① (= Technik ③) links und ② (= Technik ④) rechts und auf ENDO-Agar (Fuchsin-Laktose-Agar) bei 27 °C für 48 h bebrüten.

Kolonien in drei Gruppen teilen:

Gruppe I: Kolonien mit gutem Fuchsinglanz

Gruppe II: dunkelrote Kolonien

Gruppe III: rosafarbene, Schleim bildende oder schwarzrote Kolonien und alle anderen

Die Verteilung der Kolonien wird in % angegeben und links der Wert aus Suspension ③ rechts der aus ④ aufgeschrieben.

Die Kolonien aus ③ und ④ werden auf Objektträger aufgebracht, mit Methylenblau angefärbt und mit Ölimmersion unter dem Mikroskop nach ihrem Zersetzungsgrad wie folgt differenziert:

Gruppe I: normalerweise nur kräftige Kurzstäbchen, wenn stark gequollen, granuliert oder sehr klein in Gruppe II

Gruppe II: vorwiegend leidlich gut ausgebildete Zellen, wenn merkliche Anteile ausgebildete Zellen dort sind, in Gruppe I sortieren

Gruppe III: schlecht ausgebildete Zellen

Daraus errechnet Rusch einen biologischen Index zur Bewertung der biologischen Qualität:

$$\frac{(\%\ ③ + \%\ ④ \text{ der Gruppe I}) \times 3 + (\%\ ③ + \%\ ④ \text{ der Gruppe II})}{(\%\ ③ + \%\ ④ \text{ der Gruppe III})}$$

Liegt der Wert über 5 spricht Rusch von Böden mit hochwertiger biologischer Qualität.

Zusammengestellt geben die vier ermittelten Werte nach Rusch ein Maß zur Beurteilung der Bodenfruchtbarkeit. Ein Beispiel der mehrfachen Analyse von Blindproben zweier Böden ist nachfolgend dargestellt (192) (Tabelle 3).

Tabelle 3:

Ergebnisse mehrfacher Analyse zweier Böden mit dem Rusch-Test (Rusch, 1964)

Substrat	Test Nr.	Zell-zahlen	Bakterienflora nach						Biolog. Index
		nach ① / ②	③			④			
Gruppe			I	II	III	I	II	III	
Boden 1	1.	9/34	10	40	50	40	40	20	9,4
	2.	14/37	05	30	65	50	50	00	10,7
	3.	8/46	10	50	40	40	30	30	7,0
	4.	10/31	10	50	40	40	40	20	10,0
Boden 2	1.	45/130	00	10	90	00	30	70	0,5
	2.	46/136	00	10	90	00	20	80	0,3
	3.	42/130	00	20	80	00	00	100	0,2
	4.	46/124	00	10	90	00	20	80	0,3

Die Auswertung der Zellzahlen erfolgte von Rusch anhand empirischer Erfahrungen (161) (Tabelle 4).

Tabelle 4:

Der Rusch-Test: Richtwerte für die Auswertung der Zellzahlen in Böden und organischen Düngern (Rusch, 1964)

	Zellzahlen (zellulare Gare)	Zellzahlen (Plasmagare)
Mutterböden		
Unfruchtbare	3 - 9	unter 30
Wenig fruchtbare	10 - 15	30 - 45
Mäßig fruchtbare	15 - 25	45 - 75
Gut fruchtbare	25 - 35	75 - 100
Sehr gut fruchtbare	über 35	über 100
Komposte, organische Dünger		
Unbrauchbare	unter 30	unter 100
Wenig brauchbare	30 - 50	100 - 200
Bedingt brauchbare	50 - 100	200 - 500
Mittelgute	100 - 200	500 - 2000
Gute und sehr gute	über 200	über 2000

Nach Rusch sind diese Zahlen wie folgt zu interpretieren: *„Nähern sich die Zahlen ① und ② deutlich, so haben wir es mit einem relativ frisch gedüngten, bisher oder in den letzten beiden Jahren wenig gedüngten Böden zu tun, und je näher die beiden Zahlen stehen, desto eher besteht die Gefahr, dass Pflanzenschäden in irgendeiner Form auftreten....wir haben es dann mit viel Zellgare aber verhältnismäßig wenig Plasmagare zu tun (160). ... Für organische Dünger gilt, dass sie umso besser sind, je größer der Unterschied zwischen der Zellzahl ① und ② ist, denn dann sind sie hochwertig und unschädlich zugleich."*

1.4 Ruschs Systematisierung des Abbaus organischer Substanz im Boden

Rusch demonstriert den Ablauf der biologischen Degradation organischen Materials im Boden mit Hilfe eines *„Grundversuches" (144):*

Hierbei wird auf eine ebene natürliche Bodenfläche, bei der die Krume entfernt wurde, 30 bis 50 cm Erde mittlerer Güte mit weniger als 10 Zellen nach Technik ① geschichtet. Dies wird mit 3 bis 5 cm organischem Material aus pflanzlichen und tierischen Düngern bedeckt. Die Oberfläche wird von Pflanzenwachstum freigehalten. Während der 2 bis 3 jährigen Versuchsdauer kommt es zu einem typischen Ablauf der Zersetzung (145) (Tabelle 5).

Tabelle 5:

Bodenschichtung beim Grundversuch nach Rusch (111, 145)

1. Die Oberfläche der organischen Decke bleibt erhalten, nur feine Partikel werden tiefer geschwemmt, es kommt nicht zu nennenswerten Koloniebildungen von Mikroben.
2. Unter der Deckschicht bildet sich eine biologisch aktive Rotteschicht aus, die mikrobielle Verbauung reicht nach einiger Zeit tief in die darunterliegende mineralische zellarme Schicht hinein.
3. Die entstandene Schicht der mikrobiellen Zellgare besteht aus Teilen der organischen Auflage und der darunterliegenden Schicht und weist Krümel und Hohlräume auf. Nach 3 bis 12 Monaten zerfällt die Struktur zunehmend. Die Zellgare ist aufgrund des Kohlenhydratabbaus abgeschlossen.
4. In den ersten Wochen des Versuches ist die Plasmagare (Zellzahl ②) noch nicht mikroskopisch sichtbar. Das Zerfallsprodukt der Zellgare verteilt sich nach 2 bis 3 Versuchsjahren über die gesamte Erdauflage zum Teil bis tief in die Erdunterlage. Es entstehen zellarme feinstrukturierte, äußerst sorptionsfähige, feinkrümelige, nicht ausschwemmbare Systeme.

Rusch postuliert, dass die Zellzahl ① in der Erdauflage des Grundversuchs bei diesem Versuch kontinuierlich abnimmt und die Zellzahl ② sich nach analoger Abnahme zur Zellzahl ① auf einem stabilem hohen Niveau einpendle (140, 159).

1.5 Eigenschaften der Zellgare, der plasmatischen Gare und die biologische Qualität

Rusch fand in Keimversuchen (140, 141), dass die Region der *„Zellgare"* für die Kulturpflanze unzuträglich ist und daher eine übertriebene Zellgare, wie das stoßartige Einbringen unzersetzter organischer Substanz in die Tiefe der Krume und unnötige, tiefe Bodenbearbeitung, streng zu vermeiden seien. Andernfalls sei die Pflanze gezwungen mit der *„Zellgare"* in Beziehung zu treten was auf jeden Fall folgenschwere Pflanzenschädigungen nach sich ziehen würde (143, 147). Jedoch relativiert er, dass sich *„die Beziehung der Pflanze zur zellulären Gare ... im allgemeinen auf gewöhnlichen Acker-, Wiesen- und Gartenböden nicht darstellen lässt. Die Zellgareschicht ist hier nur ausnahmsweise und ganz vorübergehend stark genug und deutlich ausgebildet... Bei fortlaufend organischer Bodenernährung gehen die einzelnen Vorgänge ... so sehr ineinander über, dass sich die Beziehung zur Pflanze, das heißt ihres Wurzelsystems zur Zellgare nicht studieren lässt (146).* Zur *„plasmatischen Gare"* führt Rusch an, dass Fruchtbarkeit erst dann im Boden entsteht, wenn die *„zellulare Gare"* abgeschlossen sei und ihre Substanzen in die *„plasmatische, makromolekulare Phase"* übergegangen seien. *Demnach lässt sich die Bodenfruchtbarkeit nur in der kolloidalen makromolekularen Stufe des biologischen Substanzkreislaufs messen; nicht die in einer Probe enthaltenen Mikrobienzellen zeigen sie an, sondern die in ihr enthaltenen Makromoleküle... (149).*

Nachweis und Quantifizierung der makromolekularen Verbindungen mit Färbe- oder mittels optischer Methoden (Tyndall-Effekt) (Tyndall, 1872) gelangen ihm aufgrund unterschiedlicher Bodeneigenschaften und Färbungen nicht systematisch (150). Rusch vermutete Kohlenhydratausscheidungen der Wurzeln, die Bakterienwachstum auch in der Plasmaschicht ermöglichen würden und damit zur Ausbildung der Rhizosphärenflora aus Milchsäurebakterien führen (150) und stellt diesen Mechanismus in der Technik zur Bestimmung der Zellzahl ② durch die Zugabe von Milchzucker nach (153). Milchsäurebakterien beschreibt Rusch (1970) als Bakterien, die Milchsäure bilden. Milchzucker sei besonders charakteristisch für Bakterien, die bei Pflanzen, Tieren und Menschen leben. Bakteriensymbiosen bei Mensch und Tier dienen ihm als Beweis für positive Wirkungen der so charakterisierten Bakteriengruppe im Boden (Rusch, 1970). Rusch betont jedoch, dass es keine absolute Relation von den Erträgen zu den Zellzahlen gäbe, weil der Ertrag auch von anderen Wachstumsbedingungen wie z. B. der biologischen Qualität sowie geographischen, klimatischen sowie bodenchemischen und -physikalischen Bedingungen abhänge (162).

1.6. Flächenkompostierung und organische Düngung

Die Vorzüglichkeit einer Flächenkompostierung belegt Rusch anhand von Experimenten zur Zellzahlentwicklung in Böden bei Stallmistausbringung (frischer Stallmist eingepflügt – kompostierter Stallmist eingepflügt – frischer Stallmist flach eingebracht). Rusch schloss, dass nur die Flächenkompostierung in der Lage sei, die *volle biologische Potenzübertragung von organischen Düngern auf den Mutterboden* zu gewährleisten. Dies folge dem Vorbild der Natur, die nur Flächenkompostierung kenne. Eingepflügter kompostierter Stallmist sei zwar pflanzenverträglich, wirke sich aber weniger förderlich auf die plasmatische Gare aus (166, 167). Betriebsdünger sollten daher frisch und oberflächlich und ganzjährig ausgebracht werden (167). Laub wirke als Bodenbedeckung wachstumshemmend (168). Weitere Hintergründe zur Flächenkompostierung finden sich in anderen Arbeiten Ruschs. In seiner Schrift von 1970 fasst Rusch diesen Zusammenhang und die Bedeutung seines Grundversuchs (Tabelle 5) zusammen: Die früher übliche langjährige Kompostierung organischen Materials bis zur Vererdung führe zwar zu guter mikrobiologischer Qualität *„wogegen die im Ausgangsmaterial steckenden Wachstumsenergien bis auf Reste verschwendet wurde".* Diese Erkenntnis hätte in den von ihm begleiteten Betrieben zu einem Verschwinden der Stapelmistwirtschaft und zu erheblichen Steigerungen der Erträge geführt. Aus der im Grundversuch ermittelten *„funktionell streng voneinander getrennten Schichtung der Humusbildung"* folgert Rusch, dass nur eine flache Bodenbearbeitung zulässig ist, um die Schichtung und die Humusbildung im Boden nicht zu zerstören, aber auch die Ausbildung des Feinwurzelsystems der Kulturpflanzen nicht zu behindern. In einem Vortrag schlägt Rusch 1952 aufgrund der von ihm gefundenen *„Kümmerung von Pflanzen"* die in *„Faulungserde"* wachsen, eine *„abgekürzte Kompostierung"* als Vorbehandlung vor. Der für eine vollständige Kompostierung bis zur Krümelbildung benötigte Zeitraum von zwei Jahren sei für landwirtschaftliche Betriebe zu lang. Nur der 4 bis 12-wöchige *„Faulungsprozess"* von *„wirtschaftseigenen Abfallstoffen"* solle daher in Kompostmieten durchgeführt, das Material dann oberirdisch aufgebracht und nicht in den Boden eingearbeitet werden. Für landwirtschaftliche Betriebe schlägt Rusch dafür 50 cm hohe Kompostmieten vor. Das Material darin soll 50 % Wassergehalt haben und durch Drainagen unter den Mieten belüftet werden. Kalt gefaultes Material aus solchen Mieten sei biologisch nicht unterschiedlich zu heiß gefaultem aus *„industriellen Kompostierungsanlagen"* (Rusch, 1955).

Auf leichten Böden zeigen Ruschs Messungen, dass die *„plasmatische Gare"* (Zellzahl ②) bei organischer Düngung relativ niedrig bleibt, die *„zellulare Gare"* (Zellzahl ①) jedoch hoch. Organische Dünger sollten daher auf solchen Böden mehrfach in kleinen Gaben verabreicht werden. Ist auch die Zellzahl ① gering liegt eine geringe Bodenfruchtbarkeit vor (169). Auf schweren Böden ist der Anstieg der Zellzahlen mit mäßig fortlaufender Bodenernährung allmählich konstant ansteigend zu erreichen. Es werden deutlich höhere Zellzahlen ② erreicht. Die Zellzahl ① ist auf schweren Böden nicht höher als auf leichten (170).

Für die Bestimmung der biologischen Qualität von Böden wurde von Rusch die aerobe Stäbchenflora herangezogen, die, so Rusch, *„gegenüber bakteriostatischen Hemmstoffeinwirkungen die bei weitem empfindlichste Flora ist"* (187). Er stellt fest, *„dass Böden, die sich anhand anderer Zeichen als gesund erweisen und deren Produkte zum Beispiel keine Mangelerscheinungen haben, höheren Anteil an Zellen der Gruppen I und II als der Gruppe III haben"* (193) (Tabelle 2). Anzeichen für Gifte in Böden sind nach Rusch an irregulärer Koloniebildung und irregulären Kolonieformen erkennbar (201). Hemmstoffwirkungen, z. B. durch tief eingearbeitete, unzersetzte organische Massen, sind am insgesamt geringerem Mikrobenwachstum deutlich (198, 203). Ein weiteres Qualitätskriterium ist für Rusch die Zahl der Regenwürmer als Indikator für *ausreichende organische Bodenernährung* (205 bis 206). So seien in fruchtbaren Böden höhere Abundanzen anzutreffen als in weniger fruchtbaren.

1.7 Ruschs Schlussfolgerungen für die Bodenbewirtschaftung

In den abschließenden Richtlinien für die Humuswirtschaft (215) hebt Rusch noch einmal praktische Erfordernisse hervor. *„Bei der Bodenbearbeitung ist es grundsätzlich geboten, jede irgendwie entbehrliche Bodenarbeit zu vermeiden"* um die für die Humusbildung wichtige Schichtung aus Bodendecke, Zell- und Plasmagare nicht zu stören. Eine oberflächliche Bodenbearbeitung z. B. zur Saat wird als verhältnismäßig unschädlich bezeichnet, aber *„auf der Höhe des Zellgare, in den wärmsten Monaten, sollte der Boden möglichst in Ruhe gelassen werden"* (214, 215). Wo immer es möglich sei müsse eine leichte, natürliche Bedeckung des Bodens angestrebt werden (217). *„Für die optimale organische Düngung soll der Dünger nach seinem natürlichen Anfall sobald und so frisch wie möglich auf den Boden aufgebracht werden."* Frischer organischer Dünger dürfe nicht in den Boden eingearbeitet werden solange die Zellzahl ① noch hoch ist (220). Rusch rechnet damit, dass man nach Einführung der Humuswirtschaft *„durchschnittlich mit 3 bis 4 Jahren zu rechnen hat, bis ein misshandelter Bodenorganismus gesund wird"* (221).

Das Buch „Bodenfruchtbarkeit" schließt mit Beispielen, wie die Entwicklung der Kennwerte Ruschs zur Bodenfruchtbarkeit mit steigenden Ertragsleistungen im Pflanzenbau und in der Tierproduktion einher gehen (231ff). Rusch fasst als Leitlinie für die Bodenbewirtschaftung unter anderem zusammen: *„Im Mittelpunkt der Bemühung des ökologischen Landbaus steht die vollkommene Ernährung des Bodenorganismus und der Verzicht auf jeden Eingriff in die Beziehung Boden-Pflanze"* (235).

2 Verankerung der Erkenntnisse im biologischen Landbau heute

In den Bioland Richtlinien (Bioland e.V., 2007) wird bereits im Vorwort auf die organisch-biologische Landbaumethode nach Dr. Hans Müller und Dr. Hans Peter Rusch verwiesen und im Ergebnis eine Flächenkompostierung nach den Grundsätzen der von ihnen postulierten Humuswirtschaft empfohlen. Konsequenzen aus den Erkenntnissen Ruschs zur Flächenkompostierung und zum Aufbau des Humusgehaltes sind in den Richtlinien jedoch nicht zwingend verankert sondern in das Ermessen der Landwirte gestellt. Auch die von Rusch als natürlich angesehene Rückfuhr von Nährstoffen und organischer Substanz aus der Nahrungskette ist nicht Bestandteil der Richtlinien. In der Öko-Basisverordnung (EG) 834/2007 wird auf die Rezyklierung von Abfällen und pflanzlichen und tierischen Nebenprodukten aus lokal organisierten landwirtschaftlichen Systemen verwiesen, um den Anbauflächen die Nährstoffe wieder zuzuführen. Ein Rückfluss aus dem Konsumentenbereich wird dort jedoch nicht explizit benannt. Aus Sicht der Nährstofferhaltung wird heute die Entwicklung von Techniken zur schadstoffarmen Rückfuhr von Nährstoffen und Humus auch aus diesem Bereich gefordert (Rahmann et al., 2009).

3 Diskussion

Im folgenden wird diskutiert, welche von Ruschs Thesen zur Humuswirtschaft heute noch Bedeutung haben und Anregungen zur Weiterentwicklung gegeben.

3.1 Ruschs Humuswirtschaft und die Nährstoffversorgung von Pflanzen im Ökologischen Landbau

Rusch hebt mit dem Tongehalt, der Kationenaustauschkapazität und der Austauscherbelegung die wesentlichen, auch in der konventionellen Standortkunde anerkannten Fruchtbarkeitsparameter der mineralischen Bodensubstanz hervor (Scheffer und Schachtschabel, 2008). In den heutigen Standardbodenanalysen, die für landwirtschaftliche Betriebe durchgeführt werden, werden Daten zur Austauscherbelegung jedoch nicht mehr erfasst. So stellt der Verzicht auf die Bestimmung des H-Wertes zur Ermittlung des Kalkbedarfes von Böden (VDLUFA, 2000), der ein indirektes Maß für die Kationenbelegung der Austauscher darstellte (Bundesanstalt für Geowissenschaften, 2005; Lecher et al., 2001) und die ausschließliche Bestimmung des pH-Werts als Grundlage für die Kalkbedarfsbestimmung, einen Verlust von Details chemischer Bodeneigenschaften dar. Der pH-Wert stellt nur eine vereinfachte Form der Analyse der Austauscherbelegung dar. Hier wird unterstellt, dass H^+ Ionen die primären Konkurrenten der strukturfördernden zweiwertigen Ca- und Mg-Ionen sind. In Systemen mit erhöhtem Zufuhr an K-Ionen jedoch, wie z. B. Güllewirtschaft, können auch diese zweiwertige Kationen von den Austauschern verdrängen und so durch eine verstärkte Peptisierung der Bodenkolloide die Bodenstruktur nachteilig beeinflussen. Der Anteil von K-Ionen an der Austauscherbelegung wird jedoch bei einer einfachen pH-Bestimmung nicht berücksichtigt. Eine bloße Bestimmung des pH-Wertes als Grundlage für die Bestimmung des Kalkbedarfes von Böden kann daher durchaus eine zu gering bemessene Kalkmenge zur Folge haben. Besonders Düngungssysteme die auf der Nachlieferung von Nährstoffen aus dem Boden basieren, wie im Ökologischen Landbau, sind auf die Kenntnis bodenchemischer Details angewiesen (Haneklaus et al., 2005; Balzer, 2000).

Ruschs These, dass unter normalen Umständen ohne künstliche Eingriffe der Tonkristallschwund verhindert wird, und *„die Natur ein fließendes Gleichgewicht zwischen Ionenverbrauch und Ionenschonung, zwischen Tonkristallbildung und –Zerstörung"* aufrecht erhält widerspricht jedoch der Kenntnis um die langfristige Bodenentwicklung, in der es, wenn auch in sehr langen Zeiträumen, zu natürlicher Bodendegradation mit Tonauswaschung kommt (WBGU, 1993). Zweifellos verändert die Zufuhr von Mineraldüngern immer auch das Ionengleichgewicht im Boden. Hier wäre zu klären, ob Böden der Rusch'en Humuswirtschaft tatsächlich weniger der regelmäßigen Kalkungen bedürfen und ob die von Rusch postulierte „natürliche" pH-Regulierung nahe des Neutralpunktes und das spontane Aufrechterhalten der Ionengleichgewichte (99) sich nachhaltig einstellen. Einvernehmen herrscht darüber, dass durch Humusmehrung eine verbesserte Pufferkapazität gegenüber spontanen pH-Schwankungen auftritt, jedoch sinkt der pH durch den vermehrten Umsatz organischer Substanz (Godsey et al., 2007; Astier et al., 2006). Weiterhin wird berichtet, dass höhere Leguminosenanteile in der Fruchtfolge den pH-Wert des Bodens absenken (Kirchmann et al., 2007). Ruschs sehr positive These zur Stabilisierung der Boden pH-Werte bei Humuswirtschaft wird durch diese Ergebnisse deutlich in Frage gestellt und es kann keinesfalls auf regelmäßige Bodenanalysen und Kalkungen verzichtet werden. In Studien zur pH-Entwicklung bei unterschiedlicher Düngung wurden meist höhere pH-Werte der Böden mit Stallmist- oder Kompostdüngung gefunden, da die pH-Werte der Ausgangsmaterialien bereits hoch waren (Fliessbach et al., 2007; Steinwender et al., 2000; Richter et al., 1997; Alföldi et al., 1993). Die von Rusch benannte K-Sperre im Boden durch stabilen Einbau der K-Ionen in Schichtsilikate bei Säureüberschuss (97) und eine P-Sperre bei Einarbeiten P-armen Materials (98) sind in der Literatur belegt (Scheffer und Schachtschabel, 2008).

Interessant ist Ruschs These, dass organisch gebundenes P die maßgebliche Quelle für die P-Ernährung der Pflanzen darstelle und ein Gleichgewicht zwischen Ca-Phosphaten und organisch gebundenem P bestehe (98). P sei im biologischen Substanzkreislauf reichlich vorhanden und anorganische Bodenreserven brauchen in der Humuswirtschaft kaum angegriffen zu werden, da das organisch gebundene P ausreiche die Pflanzen zu ernähren. In der Tat erscheint in der traditionellen Düngerlehre die Bedeutung organisch gebundener P-Fraktionen zur P-Ernährung der Pflanzen kaum gewürdigt. Alle gebräuchlichen Methoden der Bodenuntersuchung auf pflanzenverfügbares P beziehen sich ausschließlich auf anorganische P-Fraktionen obgleich lange bekannt ist, dass etwa die Hälfte des gesamten Bodenvorrats an P organisch gebunden ist. Historisch erklärt sich dieser Mangel aus der Dominanz primär produktorientierter Forschung zu P-Düngern.

Auch bei anderen Nährstoffen reiche nach Rusch die natürliche Mobilisierung aus um den Bedarf der Pflanzen zu decken. Koinzidenz bestehe zwischen der natürlichen Bodenerwärmung, der Nährstoffmobilisierung und dem

Beginn des Pflanzenwachstums (73). Zu diesen Thesen Ruschs besteht Klärungsbedarf hinsichtlich zeitgerechter Nährstoffverfügbarkeit (David et al., 2004) und der Ausgeglichenheit der Nährstoffbilanzen (Loes und Ogaard, 2001; Lindenthal, 2000). Die Rolle der weitgehenden Nährstoffmobilisierung aus der organischen Substanz, der stabilen Fließgleichgewichte im Boden und die These des Aufrechterhaltens der Bodenfruchtbarkeit durch Humuswirtschaft unter dem Aspekt der Verwendung ausschließlich betriebseigener Stoffe sollten anhand umfassender Bilanzstudien in ökologischen Betrieben untersucht werden, insbesondere um der Gefahr eines „nutrient mining“ und daraus folgendem Verlust an Bodenfruchtbarkeit rechtzeitig begegnen zu können (Nelson und Janke, 2007; Watson et al., 2002). Hierzu gehört auch die eingehende Betrachtung der Kalkbilanz bzw. die Entwicklung der pH-Werte derart bewirtschafteter Böden.

Denn im Gegensatz zu Ruschs euphorischer Formulierung zum P-Gleichgewicht bei Humuswirtschaft zeigen historische Betrachtungen von Anbausystemen nach Umbruch nativer Böden, dass Nährstoffmangelsymptome durch Humusabbau lange verdeckt werden können (z. B. Prärieumbruch), dass aber keinesfalls ein Gleichgewicht entsteht und dass in über lange Zeit produktive historische Agrarsysteme Nährstoffe stets nachgeliefert wurden (z. B. Nildelta) (Newman, 1997). Bereits Liebig forderte trotz seiner Erkenntnisse zur mineralischen Düngung einen Nährstoffrückfluss auf die landwirtschaftlichen Betriebe mit den Worten: *„Es giebt ein Recept für die Fruchtbarkeit unserer Felder, und für die ewige Dauer ihrer Erträge..: Ein jeder Landwirth, der einen Sack Getreide nach der Stadt fährt, oder einen Centner Reps, oder Rüben, Kartoffeln etc., sollte eben so viel (wo möglich mehr) von den Bodenbestandteilen seiner Feldfrüchte wieder aus der Stadt mitnehmen, und dem Feld geben, dem er sie entnommen hat“* (Liebig, 1878). Die von Rusch im Sinne Liebigs geforderte Zufuhr von externen Materialien (Humusstoffen) und den darin enthaltenen Nährstoffen in die Landwirtschaft sollte daher als wesentlicher Bestandteil der Humuswirtschaft nicht vergessen werden. In diese Richtung weisen z. B. auch Ergebnisse des Schweizer DOK-Versuches (Anbauvergleich Biologisch-Dynamisch, Biologisch-Organisch, Konventionell) in dem Anbausysteme mit organischer Düngung massive Ertragsvorteile gegenüber Systemen ohne Nährstoffzufuhr aufweisen (Fliessbach et al., 2003). Die Verwendung von Klärschlamm und Komposten aus Haushalten ist jedoch im Ökologischen Landbau nicht zulässig, da sie mit unerwünschten Stoffen kontaminiert sein können (Kratz und Schnug, 2005). Die Extraktion der darin enthaltenen Nährstoffe wird jedoch diskutiert (Adam et al., 2008; Gethke et al., 2008).

Auch bei reiner Humuswirtschaft müssen sich Landwirte daher einen Überblick über die Nährstoffbilanzen ihres Betriebes verschaffen und ggf. Veränderungen durch geeignete Dünger oder Umstellungen der Fruchtfolge vornehmen. Nur so können langfristig Nährstoffmangelsituationen insbesondere bei P und K (Römer, 2006; Quintern et al., 2006a; Römer und Lehne, 2004a; Römer und Lehne, 2004b) vermieden werden. Wegen der systembedingten Beschränkung in der Wahl der Mittel zur Ergänzung von Nährstoffverlusten ist ein umfangreiches und ständiges Monitoring von Merkmalen der Bodenfruchtbarkeit in ökologisch wirtschaftenden Betrieben unabdingbarer Bestandteil der guten fachlichen Praxis.

Zu den Verfahren der Bodenanalyse ist anzumerken, dass die heute angewandten Nährstoffgrenzwerte für Böden aus gedüngten Hochertragssystemen entwickelt worden sind. Es ist unklar, welche Relevanz Standard-Bodengrenzwerte tatsächlich für das niedrigere Produktionsniveau und die andere Nährstoffdynamik von Böden im Ökologischen Landbau haben (Askegaard et al., 2003; Askegaard und Eriksen, 2002; Raupp, 2001; Balzer und Balzer-Graf, 1984). Zum Beispiel wird folgerichtig für die ökologische Bewirtschaftung auch ein höherer optimaler Humusgehalt im Boden gefordert, da dieses Landbausystem mehr auf die Nachlieferung aus der organischen Fraktion angewiesen ist als konventionelle Systeme (Küstermann et al., 2008; Leithold, 2006). Durch die Bewirtschaftung mit Fruchtfolgen mit hoher Biodiversität weisen ökologisch bewirtschaftete Böden auch bessere Voraussetzungen für eine nachhaltige Humusanreicherung auf (Rahmann et al., 2008; Lal, 2004). Die Rusch'e Humuswirtschaft wirkt durch die Forderungen zur Bodenbedeckung und zur Bodenbearbeitung in umsatzschwachen Jahreszeiten und zur organischen Düngung in die gleiche Richtung. So hebt Rusch die Bedeutung des Humus für den Wasser- und Lufthaushalt des Bodens und für die Nährstoffnachlieferung und Nährstoffbindung zu Recht hervor. Jedoch fehlen in seinen Ausführungen konsequente Ansätze zur Nährstoffbilanzierung und gezielten Nährstoffergänzung. Für ein weiter ansteigendes Ertragsniveau im Ökologischen Landbau sind diese jedoch unabdingbar.

3.2 Verifizierung des Rusch-Tests

Ruschs Theorie zum *„Kreislauf der lebenden Substanz“* ist sehr kritisch einzustufen. Rusch wollte sich eindeutig von der Mineralstofftheorie Liebigs und der Mineraldüngung abgrenzen. Er verwies 1949 auf klinische Erfolge bei der Humantherapie mit Bakteriensuspensionen (Rusch, 1955) und implizierte aus den Erkenntnissen zu DNS und viraler Erbgutübertragung, dass nicht näher von ihm beschriebene *„Lebendsubstanzen“* aus dem Humus durch die Pflanzen in großem Umfang aufgenommen werden und das Pflanzenwachstum maßgeblich bestimmen. Humus ist für ihn *„Primitivform des lebenden Gewebes“*

(112) und wird gleichgestellt mit etwas aktiv Fruchtbarem (118). Das ist sicher nicht so. Die Humustheorie von Albrecht Thaer (1811), die fälschlich auf einer weitgehenden Aufnahme des pflanzlichen Kohlenstoffs aus dem Humus basiert (Feller et al., 2003) nennt er nicht. Rusch glaubt jedoch, dass Abbauprodukte mehr oder weniger direkt aus dem Humus in den pflanzlichen Stoffwechsel gelangen, dort eingebaut oder integriert werden bzw. in großem Umfang direkte Wirkungen haben. Dies ist nach der auch heute noch gültigen Mineralstofftheorie Liebigs falsch. Die Aufnahme von makromolekularen Stoffen durch Pflanzen ist zwar unumstritten, jedoch werden die aufgenommenen Moleküle, z. B. Aminosäuren, als Nährstoffquelle gesehen oder sind Wirkstoffe mit indirekten Wirkungen. Zahlreiche Studien existieren zur Aufnahme von N aus organischen Verbindungen durch Pflanzen (Svennerstam et al., 2008; Biernath et al., 2008; Syltie, 2002; Falkengren-Grerup et al., 2000; Mozafar und Oertli, 1992; Nishizawa und Mori, 1977) oder zu aufgenommenen DNS als P-Quelle (Doskarova, 1966) für Pflanzen. Aus Wirtschaftsdünger kann offensichtlich Vitamin B12 über die Wurzel in die Pflanze gelangen und so die Nahrungsmittelqualität verbessern (Mozafar und Oertli, 1992). Ebenso wird aber auch die Aufnahme von organischen Schadstoffen durch Pflanzen untersucht (Redshaw et al., 2008; Schroder et al., 2001; Polder et al., 1995). Zudem adressiert die Forschung heute die chemisch-biologischen Wirkungen von Verbindungen im Boden und im Humus auf die Pflanzen. Zum Beispiel wird die Wirkung der Applikation von Huminsäuren, die aus Abfällen, Böden oder Torf gewonnen wurden, untersucht, wobei eine verbesserte N- und P-Aufnahme in Gerste (Ayuso et al., 1996) und ein verbessertes Wachstum durch die C-Zufuhr bei Raygras (Ortega und Fernandez, 2007) gemessen wurden. Weiterhin nehmen Huminsäuren Einfluss auf die Genexpression in den Wurzelzellen, die Eisenaufnahme durch die Proliferation chelatierender Stoffe (Elena et al., 2009) oder auf die Proteinbildung (Carletti et al., 2008). Zudem haben verschiedene Huminstofffraktionen unterschiedliche Wirkungen auf die Phosphorversorgung von Tabak-Zellen (Zancani et al., 2009). Bei einer Blattspritzung von Huminsäure auf Wein wurden erhöhte Chlorophyllgehalte und verminderte Säuregehalte gemessen sowie größere Trauben geerntet (Ferrara und Brunetti, 2008).

Richtig ist die von Rusch hervorgehobene Rolle der Bakterien im Boden zum Beispiel für die N-Bindung aus der Luft. Aber auch über Wirkungen gasförmiger, von Bakterien entlassenen Verbindungen auf qualitative Eigenschaften von Pflanzen wird heute berichtet (Banchio et al., 2009). Ebenso ist die symbiontische Leistung der Mycorrhiza z. B. bei der Phosphoraufnahme bekannt (Burleigh und Bechmann, 2002). Jedoch sind die potentiell nutzbaren Zusammenhänge bisher nicht zu in landwirtschaftlicher Praxis anwendbaren Techniken geworden, da die Wirkung von vielen Umweltbedingungen abhängt (Raupp et al., 2009; Toljander et al., 2008; Aryal et al., 2006; Ryan und Graham, 2002). Auch die Wirkung der sogenannten Effektiven Mikroorganismen (EM) die zur Bodenverbesserung ausgebracht werden ist höchst umstritten (Mayer et al., 2008). Einvernehmen herrscht allerdings darüber, dass Humusbildung, organische Düngung und Ökologischer Landbau die pilzliche und bakterielle Besiedlung im Boden fördern und dass unter Nährstoffmangelbedingungen die biologische Nährstoffmobilisierung höher ist (Quintern et al., 2006b; Mäder et al., 2005; Gollner, 2003).

Die Rusch'en Tests auf Zellzahlen (Tabellen 1 und 2) erscheinen aufgrund der heutzutage exakt messbaren Summenparameter mikrobieller Aktivität und der Möglichkeiten der Identifikation einzelner Bodenorganismen mittels DNA-Analyse veraltet. Eine Übersicht über zur Verfügung stehende Methoden zur quantitativen Erfassung von Bodenmikroben geben (Römbke et al., 1997). Kurz genannt werden folgende: Keimtest (Plattengussverfahren), direkte Beobachtung (Mikroskopie), Chloroform Fumigation Inkubation (CFI), Chloroform Fumigation Extraktion (CFE), Substrat-induzierte Respiration (SIR), Adenosintriphosphat (ATP), Enzymaktivität, Kalorimetrie und Bodenatmung. Die von Rusch angewandten Verfahren Plattengussverfahren und Mikroskopie verlangen als optische Methoden eine große Erfahrung des Laborpersonals. Zu diesem Punkt wird berichtet, dass die Methoden sich nicht durchsetzten, da nach dem Ausscheiden von Ruschs qualifizierter Assistentin keine Person mehr über ausreichendes Know-how verfügte und die Methode daher nicht mehr weiter verfolgt wurde (Hoffmann, 2004). In der Literatur finden sich auch keine verifizierenden Analysen zu Ruschs Techniken von Dritten. Die von Rusch für die Bestimmung der plasmatischen Gare eingesetzte Technik zur Bestimmung der Zellzahl ② unter Zusatz von Kohlenhydraten (bei Rusch: Laktose-Dextrose-Mischung) mit anschließendem Auszählen ist heute durch die Substrat-Induzierte-Respiration (SIR) ersetzt. Hier wird der CO_2-Ausstoß nach Zusatz eines Nährsubstrates gemessen. Bei dieser Methode können aus dem Kurvenverlauf zusätzlich noch Rückschlüsse auf die Aktivität der Bodenmikroben gezogen werden. Insgesamt sind die heutigen Verfahren leichter zu standardisieren und liefern exakte Messwerte. Zur Einstufung mikrobieller Bodenaktivität werden weiterhin z. B. der Gehalt an mikrobiellem Kohlenstoff (C_{mik}) (Jörgensen, 1995; Vance et al., 1987) sowie die Basalatmung bestimmt. Die Lebensbedingungen für die Mikroorganismen im Boden werden durch den mikrobiellen P- und N-Gehalt weiter charakterisiert (Brookes et al., 1985; Brookes et al., 1982). Der Ergosterolgehalt (Djajakirana et al., 1996) kennzeichnet den Gehalt an pilzlicher Biomasse.

Der Forschungsstand zur Bestimmung der Leistung der mikrobiellen Populationen beschreiben Römbke et al. (1997) wie folgt:

„Ein methodischer Ansatz, der zunehmend Beachtung findet, ist die Bestimmung der spezifischen Leistung einer mikrobiellen Population. Dazu werden die mikrobielle Biomasse und eine Stoffwechselaktivität (z. B. die Respiration) in Beziehung zueinander gesetzt. Der Quotient aus Biomasse und Respiration, in der Literatur häufig als „metabolischer Quotient" oder „spezifische Atmung" angegeben, wird zur mikrobiellen Charakterisierung eingesetzt. Damit konnten, unabhängig von Bodeneigenschaften, Unterschiede zwischen Bewirtschaftungsformen in der Land- und Forstwirtschaft (Pietikainen und Fritze, 1995; Anderson und Domsch, 1990), Auswirkungen chemischer Belastung (Schonborn und Dumpert, 1990) und Unterschiede zwischen verschiedenen ökologischen Sukzessionsstadien (Santruckova, 1992; Insam und Haselwandter, 1989) festgestellt werden. Othonen (1994) weist jedoch darauf hin, dass der metabolische Quotient von zahlreichen Faktoren beeinflußt wird. Dieser Autor konnte keinen signifikanten Zusammenhang zwischen der Kontamination eines Kiefernwaldes mit Schwefel und dem metabolischen Quotienten beobachten. Auch die Studie von Wardle und Ghani (1995) kommt zu dem Ergebnis, dass der metabolische Quotient nur bedingt als Bioindikator von ökosystemaren Störungen geeignet ist, da er in erheblichem Maße durch natürliche Stressoren beeinflußt wird.

Einen weiteren mikrobiologischen Orientierungswert für Agrarböden fand Insam (1990) in der Regression zwischen dem Verhältnis von organischem und mikrobiellem Kohlenstoff einerseits und dem Verhältnis von Niederschlag und Verdunstung andererseits. Abweichungen von der Regressionsgeraden sollen auf Kohlenstoffzu- oder -abnahmen im System hinweisen. Damit sind Unterscheidungen zwischen kohlenstoffanreichernden und -verlierenden Systemen möglich. Voraussetzung ist jedoch, dass sich die betrachteten Böden im Kohlenstoffgleichgewicht befinden.

In den USA versuchen Zak et al. (1994), die Bodenbakterien in verschiedene Gesellschaften zu unterteilen, die über ihre funktionelle Diversität (die Fähigkeit, bestimmte Substrate zu metabolisieren) definiert werden. Zur Charakterisierung der Bakterien-Gesellschaften werden, unterteilt in 6 Stoffgruppen, insgesamt 128 verschiedene Kohlenstoffverbindungen als Substrat angeboten und die Anzahl der metabolisierten Substrate pro Gruppe ermittelt.

Zusammenfassend kann gesagt werden, dass es derzeit keine empirisch ermittelten Richtwerte für mikrobielle Kenngrößen gibt, weder in taxonomischer Hinsicht noch im Hinblick auf Biomassegehalte oder physiologische Eigenschaften, auf deren Grundlage eine eindeutige Standort-Klassifizierung vorgenommen werden könnte. Daher können die mikrobiellen Eigenschaften eines Standortes gegenwärtig nicht einem gültigen Orientierungswert im Sinne einer Klassifizierung gegenübergestellt und der Standort entsprechend eingeordnet werden.

Unbenommen davon besteht die Notwendigkeit, aufgrund der enormen Bedeutung der Mikroflora für das Ökosystem, bei bodenbiologischen Untersuchungen eine detaillierte Charakterisierung der mikrobiellen Eigenschaften vorzunehmen. Dazu sollten der Biomassegehalt sowie die physiologische Aktivität (z. B. Atmung) der Bodenmikroflora erfasst werden. Der Vergleich der ermittelten Daten mit den Daten anderer Standorte erlaubt dann eine Beurteilung der quantitativen und qualitativen Bedeutung der Mikroflora für das untersuchte System und kann damit wichtige Hinweise auf mögliche Beeinträchtigungen aufgrund von Belastungen geben."

Jedoch widersprechen die Methoden und Erkenntnisse nicht denen Ruschs. Auch er ermittelte den quantitativen Gehalt an Bodenmikroben (Zellzahlen). Für die qualitative Bewertung konzentrierte er sich auf ausgewählte Organismen und wählte die ubiquitär vorkommenden Stäbchenbakterien als Indikator für die bakterielle Leistungsfähigkeit und die bodenbiologische Qualität. Hier ist jedoch zu kritisieren, dass Ruschs Fokussierung auf Milchsäurebakterien die komplexen und umweltabhängigen Lebensgemeinschaften in der Rhizosphäre (Wurzelraum) ohne wissenschaftliche Begründung als allein maßgebend für seine Bodenfruchtbarkeitsparameter hervorhebt (Pinton et al., 2007; Walker et al., 2003).

Der qualitative Nachweis von Bodenmikroorganismen ist heute durch die DNA/RNA Charakterisierung eindeutig möglich und hat die optischen Methoden abgelöst. Torsvik und Øvreås (2002) geben einen Überblick über den heutigen Stand der Forschung (zusammengefasst aus dem Englischen):

„Die mikrobielle Diversität kann durch den Nachweis einzelner Spezies, die Anzahl der Arten, ihr relatives Vorkommen und funktionelle Gruppen beschrieben werden. So kann man ein sogenanntes Metagenom im Boden charakterisieren, das sowohl durch bekannte Einzelgenome spezieller Organismen als auch durch funktionelle Gruppen von Bodenorganismen ausgemacht wird. Zu verstehen, wie mikrobielle Gruppen auf unterschiedliche Umweltbedingungen, auf unterschiedliche Substratzufuhr und auf Nährstoffgehalte reagieren ist Gegenstand der heutigen Forschung. Weiterhin wird probiert, funktionelle Gruppen einzelnen Spezies zuzuordnen. Es wird versucht, die

in bestimmten Stoffwechselsituationen des Bodens aktiven Mikoorganismen durch die Aufnahme von Tracer-Substanzen zu identifizieren und zu charakterisieren. Darüber hinaus wird über die räumliche Isolation und Verteilung von Mikrooorganismen im Boden geforscht. Die Charakterisierung solcher Mikrohabitate ermöglicht es, bodenphysikalische und bodenchemische Faktoren zu finden, die mikrobielles Wachstum bzw. einzelne Spezies fördern. Beschrieben wird z. B. dass 80 % der Bakterien in den Mikroporen stabiler Mikroaggregate des Bodens zu finden sind. Weiterhin hat die Korngröße des Bodens einen höheren Einfluss auf die mikrobielle Diversität als z. B. der Boden-pH oder die Art des Substrates. Ebenso wird über die Zusammenhänge von Bodennährstoffangebot und Bakteriendiversität und -wachstum sowie über Vorkommen und Ursachen der saisonalen Fluktuation der Mikrobenarten geforscht. Offensichtlich bleibt die mikrobielle Biomasse der Arten über die Jahreszeiten relativ konstant, ändert sich aber in der Zusammensetzung. Auch aus den neueren Arbeiten wird geschlossen – wie auch schon von Rusch postuliert –*, dass eine hohe mikrobielle Diversität, eine hohe Rezyklierung der organischen Komponenten absichert. Die Effizienz des Nährstoffrecyclings ist Kernparameter für das Bodenökosystem hinsichtlich der Produktivität und Biomassebildung funktioneller Gruppen. Gefunden wurde, dass eine hohe katabolische Gleichmäßigkeit von Böden (Grünland) im Vergleich zu katabolischer Diversität (Acker) resistenter gegenüber Umwelteinflüssen ist. Studien zur Gensequenz in Mikrohabitaten und der Genexpression unter verschiedenen Umwelteinflüssen können dabei helfen, neue Bewirtschaftungsmethoden zur Förderung spezieller Bodenmikroben oder deren Gesamtheit zu geben".*

In der Nachfolge von Rusch wurden zahlreiche Untersuchungen zur Wirkung ökologischer Bewirtschaftungsverfahren auf die Biodiversität unter Berücksichtigung verschiedener Raum- und Zeitskalen durchgeführt. Eine kürzlich publizierte Meta-Analyse zeigt, dass Verfahren des Ökolandbaus aber nicht immer positiv auf Artenreichtum und Abundanz der Organismen wirken (Bengtsson et al., 2005). Neuerdings wird auch der Einsatz konservierender Bodenbearbeitungsverfahren im Ökolandbau versucht und kontrovers diskutiert (Peigne et al., 2007). Diese wirkte jedoch im Ökolandbau nicht positiv auf die Regenwurmpopulation (Peigne et al., 2009; Metzke et al., 2007). Hingegen begünstigen ökologische Bewirtschaftungsverfahren ansich höhere Abundanzen und Biomassen sowie ein größeres Artenspektrum an Regenwürmern (Bengtsson et al., 2005; Pfiffner, 1993). Rusch erkannte richtig, dass die Qualität der organischen Substanz für die Häufigkeit der Regenwürmer entscheidend ist und nicht die Quantität. Daraus folgert er, dass Regenwürmer eher ein Zeichen für biologische Güte als quantitatives Maß Bodenwertigkeit sind. Allerdings differenziert er nicht weiter in einzelne Arten und deren spezifische Ansprüche. Auch fand er keine Beziehung zwischen Individuendichte der Regenwürmer und pH-Wert (207), was sicherlich an der gewählten Spanne von pH 6,0 bis 7,4 lag. Diese entspricht ziemlich genau dem Optimalbereich aller heimischen Arten. Neuere Studien zeigen einen deutlichen Einfluss der pH-Werte auf die Regenwurmpopulationen (Joschko et al., 2006; Didden, 2001). Zu Ruschs Zeit wurden auch bereits erste Analysen zur Bedeutung von Regenwürmern auf den Nährelement-Haushalt des Bodens durchgeführt. So gelang Graff (1970) als einem der ersten der Nachweis verbesserter Bodenfruchtbarkeit durch Regenwurmaktivität in Abhängigkeit unterschiedlicher Mulchmaterialien an der Bodenoberfläche. Die Losung in den Regenwurmgängen zeigte höhere Gehalte an Stickstoff, Phosphor, Calcium und Magnesium als der umgebende Boden.

4 Nutzbarkeit und Erweiterung der Erkenntnisse zur biologischen Bodenqualität nach Rusch

Die Widersprüche zwischen der Theorie zur Humuswirtschaft von Rusch und den heute gültigen Erkenntnissen zur Rolle des Humus bzw. des Umsatzes organischer Substanz im Boden liegen überwiegend in Ruschs Annahme *„Lebendsubstanzen"* würden die Pflanzenernährung maßgeblich bestimmen. Zudem ist die einsseitige Fokussierung seiner biologischen Tests auf Milchsäurebakterien nicht wissenschaftlich belegt. Es ist nicht von der Hand zu weisen, dass den organischen Stoffen im Boden, deren Abbauprodukten und dem Bodenleben eine enorme Bedeutung für das komplexe System der Bodenfruchtbarkeit zukommt. Jedoch sind die direkten biologisch-chemischen Zusammenhänge mit dem Pflanzenwachstum bis heute nur äußerst schwer zu verifizieren. Richtig sind Ruschs Aussagen zur Pflanzenunverträglichkeit von frisch eingearbeiteten unreifen Komposten oder anderen organischen Materialien (Requena et al., 1996; de Bertoldi et al., 1983). Sicher ist auch, dass Humus über Nährstoffbindung, Nährstoffbereitstellung, Wasserhaltevermögen, Temperaturen etc. positiv und direkt auf die Bodenfruchtbarkeit wirkt. Die tatsächliche Bedeutung von Ruschs Nachweismethoden zur Charakterisierung der Abbaubarkeit von Materialien und von mikrobiellen Kennziffern im Boden für die Landwirtschaft ist aber vermutlich gering. Zweifellos sind unterschiedliche Materialien im Boden unterschiedlich abbaubar. Es erhebt sich aber die Frage, ob solche externen Materialien für die organische Düngung in der Landwirtschaft überhaupt in großem Umfang zur Verfügung stehen, denn dieser Aspekt spielte in Ruschs Gedankenwelt eine entscheidende Rolle. Zwischenfruchtanbau, Gründüngung und die Fruchtfolgegestaltung – Maßnahmen, die landwirtschaftlich eine wichtige Bedeutung für den Kreislauf der organischen Substanz im Boden haben und die in der Praxis auch umsetzbar sind – wurden von Rusch nur untergeordnet diskutiert bzw. als ohnehin notwendig unterstellt. Die Grenzen konsequenter Umsetzung der Humuswirtschaft hat Rusch selbst formuliert. Sie liegen in der technischen Umsetzbarkeit nicht wendender Bodenbearbeitung und in der ausreichenden Zufuhr organischer Substanz z. B. in viehlosen Betrieben. Die Auswirkungen einer Bodenbearbeitung unter konsequenter Beachtung der typischen boden-biologisch bedingten Bodenschichtungen ohne Brüche in der Bewirtschaftung sind zu überprüfen. Hier sollten bodenschonende, nichtwendende Bewirtschaftungsverfahren z. B. mit Schichtengrubbern anderen Verfahren gegenübergestellt werden.

Hinsichtlich der Bewertung der Bodenfruchtbarkeit müssen übergeordnete bodenchemische und bodenphysikalische Bewertungsparameter (KAK, Austauscherbelegung, Bodenart und -schichtung), die Gehalte und Qualität der organischen Substanz des Bodens, seine Humusgehalte und -qualitäten und die mikrobielle Besiedlung von Böden in Abhängigkeit von der Bewirtschaftung näher analysiert werden. Aufgrund der dargestellten neueren Erkenntnisse zu biologischen und mineralischen Bodenfruchtbarkeitskennziffern ist neu zu überlegen, ob die grundsätzlichen Theorien von Rusch zum Umbau der organischen Substanz im Boden (Grundversuch, Zellgare, Plasmagare), zum Management (Flächenkompostierung), zur qualitativen Einstufung organischer Materialien (Zellzahlen, Biologischer Index) und zur Rolle der Bodenmikroben im Boden für die Praxis des biologischen Landbaus von Bedeutung sind.

Lagerung, Ausbringung und Einarbeitung von Wirtschaftsdüngern sollten mit geringen Nährstoffverlusten durchgeführt werden. Die Humusbildung im Boden bzw. ausgeglichene Humusbilanzen müssen wichtiges Ziel der Bewirtschaftung sein, um einem Bodenfruchtbarkeitsschwund zu begegnen. Es ist zu prüfen, ob die Bewirtschaftung nach den Prinzipien von Rusch tatsächlich positive und nachweisbare Ergebnisse für die Bodenfruchtbarkeit im Ökologischen Landbau bringt. In diesem Kontext ist es besonders wichtig eine praktikable Methode zu entwickeln, die eine Charakterisierung des Garezustandes und der biologischen Besiedlung von Böden als Summenparameter einer gelungenen ökologischen Bodenbewirtschaftung ermöglicht. Mikrobielle Details sind mit einfacher Analytik nur schwer darzustellen. Daher wären für die praktische Anwendung einfache Tests zur Beurteilung der Bodenqualität wünschenswert. Beispiele hierfür sind z. B. die Spatendiagnose zur Definition des Gefüges und des Humuszustandes (Beste und Hampl, 1999; Goerbing, 1947), Zahl und Art von Regenwürmern als einfacher Indikator für die Bodenfruchtbarkeit und C_{org} und C_{mik} zur groben Einstufung der Humusversorgung und des aktiven Bodenlebens. Die Bodenfruchtbarkeit sollte weiterhin auch anhand von pH- und Nährstoffanalysen im Boden überprüft werden, die um die konsequente Aufstellung von Nährstoffbilanzen zu erweitern sind. Die aus gedüngten Systemen abgeleiteten Bodengrenzwerte und Analysemethoden sind hinsichtlich Validität für den Ökologischen Landbau kritisch zu überprüfen. Dem Potential der Nährstoffmobilisierung von Böden in ertragsschwächeren Systemen wie dem Ökologischen Landbau sollte intensive Beachtung geschenkt werden. Besonderes Augenmerk zukünftiger Forschung sollte der Bedeutung bodenbiologischer Prozesse für den P Umsatz, sowie einer Bewertung organischer P-Fraktionen für die P-Ernährung der Pflanzen gelten (Guppy und McLaughlin, 2009; Eichler-Löbermann et al., 2007; Fortune et al., 2005; Oehl et al., 2002).

Unstrittig ist die Bedeutung des Humus für die Bodenfruchtbarkeit (Manlay et al., 2007). Die Nutzbarkeit der Rusch-Tests ist jedoch wissenschaftlich nicht belegt und die praktische Relevanz der Werte ist fraglich. Im Allgemeinplatz „Mutterboden“ bleibt dem auch von Rusch geprägten Begriff der „Mutter Erde“ aber ein bleibendes Andenken: *„Die Fruchtbarkeit kehrt dahin zurück woher sie kam“* (Rusch, 1955).

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