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Environmental Impacts of Short Rotation Coppice (SRC) Grown for Biomass on Agricultural Land

Martin Weih · Ioannis Dimitriou

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This issue of *BioEnergy Research* highlights the activities within a European Union research project funded by ERA-NET Bioenergy, "Reducing environmental impact of SRC through evidence-based integrated decision support tools" (RATING-SRC). The project goal was to provide scientific evidence for evaluating the impacts—positive and negative—of short rotation coppice (SRC) plantations on soil, water, biodiversity, and landscape issues and also to propose ways to mitigate the negative and increase the positive impacts.

Commercial SRC plantations are currently grown at limited scale in some countries (e.g., Sweden), but the area of SRC plantations has to increase dramatically in the future if they are to contribute significantly to the replacement of fossil energy sources. Thus, a rapid increase of agricultural land dedicated to SRC with willow (*Salix* spp.) or poplar (*Populus* spp.) for production of biomass for heat and/or electricity is projected in the short-term in many regions of the world. A large-scale shift from "conventional" arable crops to SRC will have implications on a range of environmental issues, and large-scale implementation of those crops for bio-energy purpose makes sense only if they prove to reduce negative effects on the environment, especially when compared with other alternatives for reduction of fossil energy sources. As a perennial crop, SRC differs from most arable crops in physical traits and management practices. Results so far imply many positive environmental benefits due to SRC implementation, but the effects that SRC will have on the environment depend on the existing or previous land use, the scale of planting and the management practices applied. In addition, SRC is a new production system for most regions in which it might be grown in future, and many uncertainties exist with respect to the environmental impacts of those plantations on soil, water, biodiversity, and landscape issues.

This special issue contains seven papers covering the topic areas of SRC effects on issues of water and soil (two papers by Dimitriou et al., Schmidt-Walter and Lamersdorf), biodiversity (Baum et al.), and the overall impact that these plantations may have on the environment and sustainability (Busch, Englund et al., Langeveld et al.). The paper on soil issues by Dimitriou et al. highlights the results for pH, organic carbon (C), and trace element concentrations in the soil of 14 long-term (10–20 years) commercial willow SRC fields in Sweden when compared with those in adjacent, conventionally managed arable soils. The paper on water issues by Dimitriou et al. reports the effects of SRC on water quality by determining differences in leaching of nitrogen and phosphorus to groundwater of 16 commercial SRC stands in Sweden compared to adjacent arable fields grown with "ordinary" crops. Schmidt-Walter and Lamersdorf describe effects on groundwater quality, specifically the potential effects of SRC grown under different management regimes on groundwater recharge. Based on flora inventories in eight landscapes located in two European regions (Germany and Sweden), the paper by Baum et al. focuses on the diversity of higher plants as an indicator of biodiversity in willow and poplar SRC, including various scales ranging from habitat to landscape level diversity. The contribution by Busch is an attempt to assess the impact of SRC on various landscape functions with the help of GIS-based tools. Englund et al. focus on the potential effects of sustainability requirements within the EU Renewable Energy Directive (RED) on different stakeholders along SRC bioenergy supply chains, assessing their usefulness in ensuring that SRC bioenergy is produced with

M. Weih (✉) · I. Dimitriou
Department of Crop Production Ecology,
Swedish University of Agricultural Sciences,
Box 7043, 750-07 Uppsala, Sweden
e-mail: martin.weih@slu.se

sufficient consideration given to the RED-associated criteria. Langeveld et al. integrate much of the scientific evidence reported in the above contributions and propose a model for semiquantitative impact assessment regarding various environmental issues.

We would especially like to thank the various anonymous reviewers for their contributions to this special issue and the four guest editors who worked so efficiently and partly under tight deadlines to organize peer review of manuscripts for this special issue. The guest editors were: Professor Lena Gustafsson, Swedish University of Agricultural Sciences, Uppsala, Sweden; Dr. Angela Karp, Rothamsted Research, Harpenden, UK; Dr. Ronald S. Zalesny Jr., U.S. Forest Service, Northern Research Station, Rhinelander, WI; and Dr. Georg von Wühlisch, Johann Heinrich von Thünen Institute, Großhansdorf, Germany.

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Impact of Willow Short Rotation Coppice on Water Quality

Ioannis Dimitriou · Blas Mola-Yudego · Pär Aronsson

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Abstract Short rotation coppice (SRC) with willow has been grown in Sweden from the late 1980s to produce biomass for energy on agricultural land. This study evaluated the effects of SRC on water quality by determining differences in leaching of nitrogen and phosphorus to groundwater of a number of commercial "old" SRC willow stands in Sweden compared to adjacent arable fields grown with "ordinary" crops. The study was conducted in 16 locations under three vegetation seasons. NO_3–N leaching from willow SRC fields was significantly lower than that from reference fields with cereals. The opposite was observed for PO_4–P; concentrations in the groundwater of SRC were higher compared to reference fields. Sewage sludge applications were not responsible for the elevated PO_4–P leaching under SRC compared to reference crops.

Keywords Bioenergy · Biomass · Energy forest · Nitrate leaching · Phosphorus leaching · Salix

Introduction

Commercial short rotation coppice (SRC) fields with willow have been grown on agricultural soils in Sweden already since the late1980s to produce wood biomass for combustion in combined heat and power plants. SRC is generally considered as a crop that improves the water quality in a certain area [1,2], and this is mainly attributed to different management practices and crop characteristics compared to arable crops. For instance, herbicides are used only during the establishment phase, in low amounts when compared to other arable crops, and they are not used thereafter. Nitrogen fertilization recommendations for willow SRC have been established and vary between 70 and 120 kg N ha^{-1} $year^{-1}$ in different countries [3–6], being therefore relatively low compared to other arable crops. In SRC, tillage occurs only during establishment of the crop, and the limited soil disturbance probably implies less nitrogen leaching to the groundwater. Moreover, SRC is a perennial crop with a permanent root system still at place in the field during autumn when peaks of nutrient leaching are expected when arable crops are grown.

Besides the crop characteristics, extensive research under controlled conditions has been conducted to estimate the ability of certain willow species to decrease leaching and retain nitrogen. For this reason, high nitrogen application rates supplied both as mineral fertilizers and as wastewater were tested, showing that willow trees are capable of retaining high amounts of nitrogen in the soil–plant system [7,8]. A substantial part of the supplied nitrogen was stored in shoots [9,10], thus enabling a removal with harvest. Losses to the air via denitrification that would imply less leaching to the groundwater (calculated up to 30 % of the supplied N in intensively fertilized plantations [9]) have been also reported. Based on the above-mentioned characteristics of SRC as a crop and of willow species to retain nitrogen, willow SRC has been suggested as a solution to decrease eutrophication problems in water bodies if grown in areas of intensive agriculture [11] or to be used in field-scale systems (vegetation filters, phytoremediation) to treat nitrogen-rich wastewaters [1,12,13].

I. Dimitriou (✉) · B. Mola-Yudego · P. Aronsson
Department of Crop Production Ecology,
Swedish University of Agricultural Sciences (SLU),
P.O. Box 7016, 750 07 Uppsala, Sweden
e-mail: ioannis.dimitriou@slu.se

B. Mola-Yudego
School of Forest Sciences, University of Eastern Finland,
PO Box 111, 80101 Joensuu, Finland

Studies of nitrogen leaching losses from experimental fields applied with mineral fertilizers have shown very low nitrogen concentrations in groundwater in SRC willow fields in Sweden and Denmark [14–17]. The results indicated comparatively small amounts of nitrogen leached to the groundwater once the SRC was established. Also, nitrogen leaching was relatively low from SRC willow fields applied with different wastewaters [18–21], although having greater variations than for fields applied with mineral fertilizers especially when large nitrogen amounts were applied [21,22]. However, and in all cases, the amounts of nitrogen leaching from willow SRC seemed to depend on a series of site-specific factors such as soil conditions (e.g., less in clay than in sand—as in Aronsson and Bergström [7]), plantation age (e.g., substantial during establishment year—as in Goodlass et al. [23]), climate conditions (e.g., higher in areas with high precipitation—as in Larsson et al. [18]), biomass produced (e.g., lower when high biomass produced—as in Dimitriou and Aronsson [8]), and management issues (e.g., higher in intensively fertilized fields —as in Aronsson et al. [21] or when sewage sludge was applied—as in Labrecque and Teodorescu [24] and Sagoo [25] when nitrogen concentrations increased the year after sludge amendments but decreased 2 years after).

Sewage sludge applications to willow SRC is considered a rather common practice in Sweden. Most newly established fields have been amended with sewage sludge or mixtures of sludge and wood-ash (when available) to achieve a more balanced fertilizer. Behind such practices is the political will to recirculate nutrients contained in sewage sludge (phosphorus and nitrogen) to agricultural soils and that the practice can be proved economically beneficial for the farmer [12,26]. Although substantial research has been conducted with regard to potential nitrogen leaching from SRC fields to groundwater as described above, a limited number of studies have evaluated phosphorus leaching to the groundwater in SRC fields, despite the increased focus on phosphorus in relation to eutrophication [27,28].

Werner and Mc Cracken [19] observed no elevated phosphorus concentrations in groundwater when willow SRC was irrigated with wastewater, and Dimitriou and Aronsson [8] reported no phosphorus leaching when wastewater and sludge was applied to willows grown in clay-soil lysimeters, and limited leaching via drainage water from sand-soil lysimeters. The phosphorus amounts supplied with the wastewater at the latter case were exceeding the ones allowed when sewage sludge is applied in Sweden (i.e., 22 and 35 kg ha^{-1} year^{-1} in form of a 7-year dose, for low and high initial soil phosphorus status, respectively), therefore willows seemed to be efficient in treating phosphorus when nutrient rich residues were applied. However, patterns of phosphorus leaching are usually difficult to predict and are dependent on several factors [28,29], and when sludge is applied to willow SRC fields, this is usually applied as a single 7-year dose in spring after harvest. This raises some uncertainty on phosphorus leaching from SRC fields that have been applied with sludge several times, especially when compared with other arable fields in the area.

The objectives of this study were to determine leaching of nitrogen and phosphorus to groundwater of commercial SRC willow fields in Sweden and to quantify the differences in nitrogen and phosphorus leaching between commercial SRC willow fields and adjacent arable fields grown with "ordinary" crops. To estimate the long-term impact of SRC on groundwater quality if SRC replaces arable crops in certain areas, a number of long-term established (preferably over 10 years) commercial SRC stands subject to different management practices were chosen to evaluate their groundwater quality. Our hypotheses were that the NO_3–N and PO_4–P concentrations in groundwater in SRC fields would be lower than that from reference fields with ordinary crops. For the SRC fields amended with sewage sludge, elevated PO_4–P concentrations in groundwater compared to the reference fields were expected.

Materials and Methods

Site Description

The locations of the different SRC fields where groundwater pipes were established are shown in Fig. 1. Most of the fields were located in east-central Sweden, one field in western and two in southern Sweden. The SRC fields were selected upon the terms that they (1) were at least 10 years old; (2) had adjacent arable fields with the same soil texture that could be used as reference; (3) were located at flat areas that could be used for establishing groundwater pipes at the same ground level as for the reference fields (to facilitate comparisons). In Table 1, a description of several features of the different SRC fields and of the reference fields is presented. In the SRC fields where mineral fertilization was conducted, this was done according to the Swedish recommendations developed by Ledin et al. in 1994 [3] (a single N application of 70–80 kg ha^{-1} year^{-1}). The applications with sludge/ash to SRCs were conducted following the Swedish regulations for sludge amendments in agricultural soils, and therefore received 22 kg Pha^{-1} year^{-1} (and almost equal amounts of N, according to averages for Swedish sludge content [8]). For the reference fields, when referring to grass no fertilization had been conducted. In the case of cereals as references, the common fertilization recommendations had been followed (120–140 kg Nha^{-1} for average yields, 10–20 kg Pha^{-1} depending on soil P status and 0–10 kg

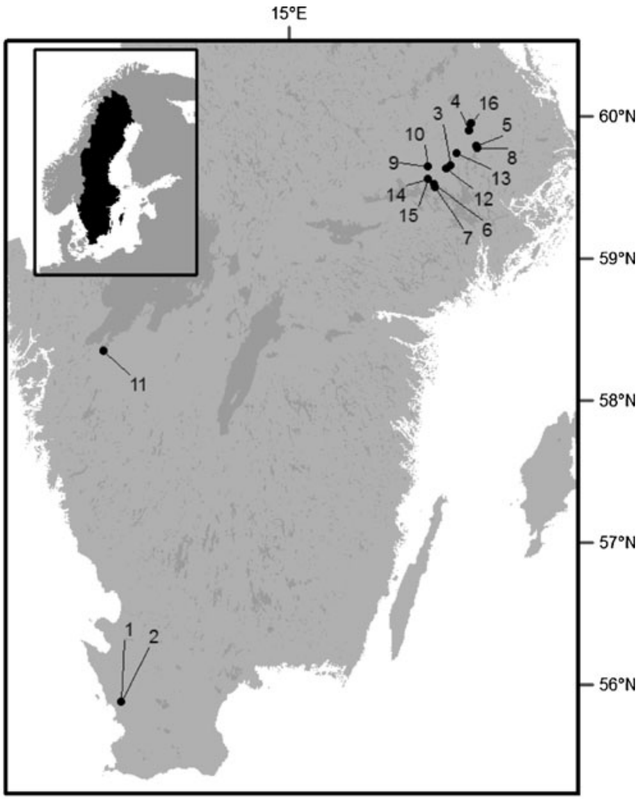

Fig. 1 Location of the studied willow short rotation coppice plantations. Numbers correspond to the locations as described in Table 1

potassium (K) [30]), and a typical crop-rotation cycle had been followed throughout all years when SRC.

Sampling and Analyses

Groundwater pipes were installed at each SRC field and their reference fields in spring 2009. In total, 49 groundwater pipes were installed (34 in SRC fields located near the field border adjacent to the respective reference field) and 15 in reference fields (in places that would not disturb farming mechanical activities, mainly close to field edges near the SRC field when possible or near drainage wells). In several SRC fields, more than one pipe was established to detect and cover variations within the fields (2 pipes in SRC in fields 6, 12, and 13; 3 pipes in SRC in fields 2, 5, 9, 10, 16; and in field 3, 5 pipes in SRC and 2 in reference). For the fields 14 and 15, a common reference pipe was used, and for field 7, no reference pipe was available. Holes were drilled down to the groundwater table using an auger. The average depth for the pipes in the different fields was 1.5 m. In all locations, the pipe length in willow SRC and in respective reference fields was almost similar.

PVC pipes with 50 mm diameter and with slits up to 0.5 m from the bottom were installed in the holes. To prevent clogging with soil particles, the base of each pipe was covered with a fiber cloth up to the slits. The holes were then filled with gravel followed by granulated bentonite clay to prevent short-cut flow of water along the pipe wall. Finally, at the top, a 110-mm PVC pipe with a cap was installed around each pipe to prevent contamination. Samples for chemical analyses were taken using a manual vacuum pump. Before sampling, the groundwater pipes were evacuated, and then a 100-ml sample of fresh groundwater was collected in a plastic jar. Sampling was conducted according to other studies for groundwater sampling for assessing leaching [21], since on structured clayey soils, hardly any other useful non-destructive method to estimate leaching in a field situation exists. Sampling was planned to be conducted from June 2009 to October 2011 on a regular basis (once a month) but was intensified during autumn and snow-melting periods when extensive drainage was assumed. All collected samples were analyzed for NO_3–N and PO_4–P, and two samples each year from each field were also analyzed for total nitrogen and total phosphorus. The analytical methods for nitrogen were according to the Swedish standard SS02813. For phosphorus, the SS-EN ISO 6878:2005 was used. All phospectrometric measurements were performed using the system Konelab Aqua 60.

Estimates of shoot biomass were conducted in each SRC field in spring 2010 and spring 2011, before plants start to grow, by use of a combination of destructive and non-destructive measurements as in Nordh and Verwijst [31]. For some fields standing biomass was estimated by the farmers.

Statistical Analyses

The potential effects on groundwater NO_3–N and PO_4–P concentrations in the fields analyzed were first tested using ANOVA. The tests were aimed to observe overall differences between the SRCs and the references. The variables tested were: SRC fields versus reference sites, measurements taken in spring and in autumn, as well as the use of sludge as fertilizer.

However, the data includes different locations, with different local conditions, that suggest auto-correlation. Therefore, a simple mixed model approach was taken, using field as grouping variable. By these means, measurement across time are grouped and treated by fields. Also, this approach is useful to identify potential local effects from the general effects of the variables analyzed. The variables studied were treated as dummy variables and their combinations were tested. A final model was presented for NO_3–N and PO_4–P. The variables included in the final model had to be significant at the 0.05 level. The model had the studied variables in the fixed part, and the effects of the plot location in the random part. The

Table 1 Description of the different locations where sampling took place

Code	Name	Year planted	Clone	Ref field	Sludge/ash	Harvested	Mineral fertilization	Soil type (0–20 cm)	Biomass 2009	Previous use
1	Billeberga I	2002	Sven	Cereals	Y/N (1)	2008	N	Sandy loam	8.5[a]	Sugarbeet
2	Billeberga II	1994	Torhild	Cereals/rapeseed	Y/N (3)	Annually	N	Loam	2[a]	Cereals
3	Djurby Gård	1990	78021	Cereals	Y/N (3)	2007/2011	N	Silty clay	5.1	Cereals
4	Forkarby	1991	78021	Cereals	N/N	2008	Y (2)	Silty clay	11.1	Cereals
5	French	1994	78021	Cereals (eco)	N/N	2007/2010	Y (8)	Clay loam	9.3	Cereals
6	Hacksta	1994	Jorr, Rapp	Peas/cereal	Y/Y (4)	2008	Y (1)	Clay loam	4.2	Cereals
7	Hjulsta	1995	Jorr	No ref	N/N	2008	N	Clay	9.6	Oil crops/cereals
8	Kurth	1992	Ulv/Rapp	Cereals (eco)	N/N	2007/2010	N	Clay loam	124	Cereals
9	Lundby Gård I	2000	Tora	Cereals	Y/Y (1)	2005	Y (1)	Clay	4.9	Cereals/oil crops
10	Lundby Gård II	1995	78021	Cereals	N/N	2005	N	Clay	2.5	Cereals
11	Puckgården	1992	78112	Cereals	N/N	2008	Y (4)	Silty clay	10[a]	Cereals
12	Skolsta	1993	78021, Orm	Cereals	Y/Y (1)	2004	Y (2)	Silty clay	4.1	Cereals
13	Säva	1993	Rapp, Orm	Grass	Y/N (2)	2007	N	Silty clay	7.4	Cereals
14	Teda I	2000	Tora	Grass	Y/Y (2)	2009	Y (2)	Silty clay loam	8.1	Cereals
15	Teda II	1993	78112	Grass	Y/Y (2)	2007	Y (2)	Clay	1.7	Cereals/Set-aside
16	Åsby	1996	Tora	Cereals	Y/Y (1)	2008	Y (2)	Silty clay	4.2	Cereals

Sludge/ash: *Y*=Yes, *N*=No. In parentheses, number of sludge/ash amendments; harvested: the given years refer to the last harvest occurred in spring. In parentheses, total number from establishment; inorganic fertilization: *Y*=Yes, *N*=No. In parentheses, total number from establishment; biomass: living biomass in t DM ha^{-1} $year^{-1}$ (note that shoots in the different fields were of different ages)

[a] Estimations from the farmer and not real measurements

estimated variances were σ_{plot} and $\sigma_{plot,\ time}$ corresponding to measurements in the same site, and measurements on a specific date in the same site, respectively.

Results

In Fig. 2, the averages of NO_3–N and PO_4–P concentrations in the groundwater for all locations throughout the whole experimental period (June 2009–October 2011) are presented. Average NO_3–N concentrations in groundwater of SRC fields were in all fields except one (field 14) lower than that of the reference fields. The highest differences for NO_3–N concentration averages were observed when reference fields were cropped with cereals. NO_3–N concentrations in the SRC fields were typically around 1 mg/L, whereas for reference fields, the NO_3–N concentrations were substantially higher. When the reference field was cropped with grass, no significant differences were observed as regards the NO_3–N concentrations (fields 7, 13, 14, 15).

PO_4–P concentrations did not follow the same trend as for NO_3–N and were in most cases higher in the groundwater from SRC fields than the reference fields (in only two fields, fields 12, 16, were lower in SRC than in reference). The average PO_4–P concentrations in the groundwater of 14 of the SRC fields in our study were lower than 0.01 mg/L and in only two fields (fields 2, 14) were substantially higher than 0.01 mg/L.

In Fig. 3, the averages of NO_3–N and PO_4–P concentrations pooled together from all fields are presented. NO_3–N concentrations were significantly higher in the groundwater of the reference fields (Fig. 3; Table 2). On the other hand, PO_4–P concentrations in the groundwater of SRC fields were significantly higher than that in the references (Fig. 3; Table 2). Similar differences were observed when comparing with only cereal fields as reference.

Application of sewage sludge/wood-ash seemed not responsible for the elevated PO_4–P concentration in the groundwater of SRC. No significant differences between sludge/wood-ash amended and non-amended SRC fields were observed (Table 2). Similarly, no other parameter tested was responsible for the PO_4–P differences between SRC and reference fields (Table 2).

The average NO_3–N and PO_4–P concentrations in the groundwater of all studied fields during spring (March until June) and autumn (September until December) are illustrated in Fig. 4. Concentrations of NO_3–N in the groundwater of reference fields were substantially higher than of the SRC fields both during spring and autumn. In the case of PO_4–P concentrations, higher concentrations were observed during the autumn measurements (Fig. 4). Finally, although there were clear effects of NO_3–N reductions in the groundwater of SRC, the between-field variability was large. The variance

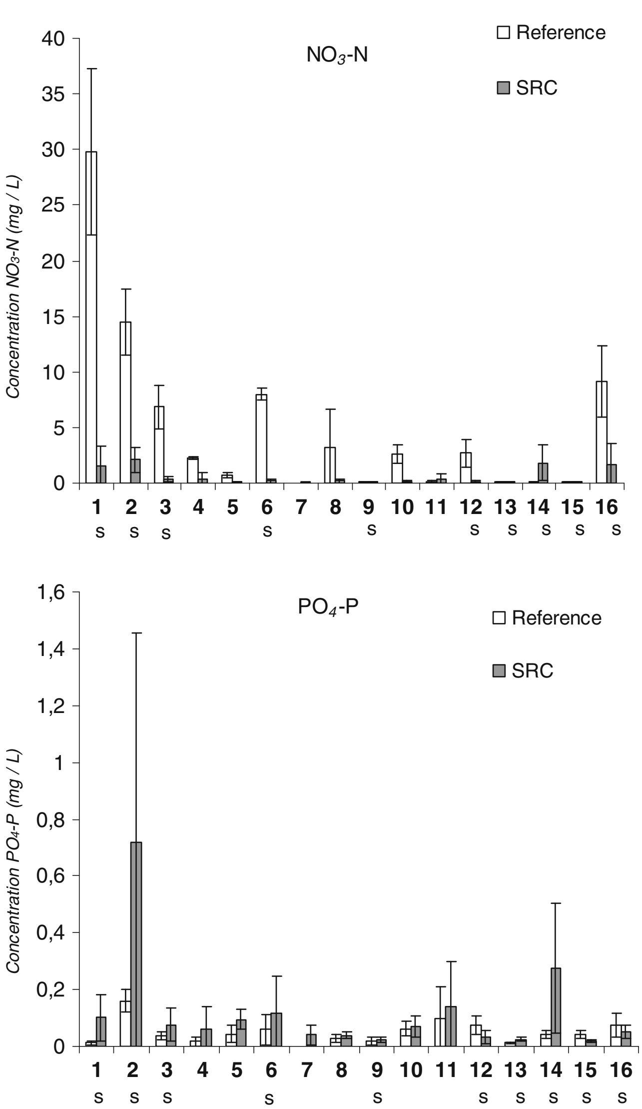

Fig. 2 Means and standard errors of NO_3–N and PO_4–P concentrations in the groundwater of short rotation coppice (SRC) plantations and the reference fields in each of the different locations throughout June 2009 to October 2011 (*s*: sludge was applied in the plantation. *Numbers* correspond to the locations as described in Table 1)

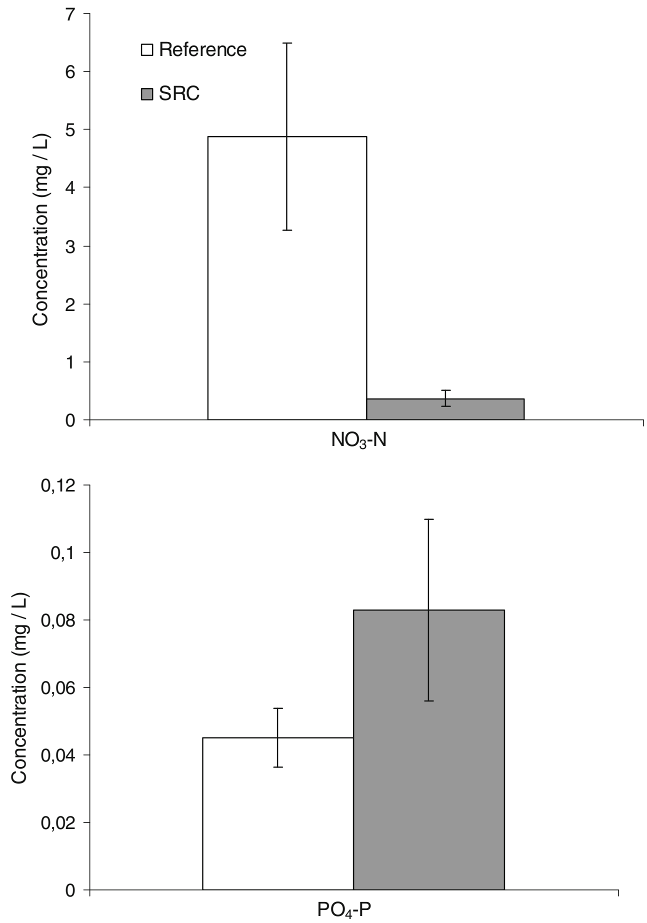

Fig. 3 Averages of NO_3–N and PO_4–P concentrations in the groundwater of all fields pooled together for short rotation coppice (SRC) and reference fields throughout June 2009 to October 2011

estimated for the between-field concentrations of the reference measurements was significant and high in both NO_3–N and PO_4–P concentrations (Table 2).

Discussion

The average NO_3–N concentrations in the groundwater of the SRC fields were significantly lower than the respective ones in the reference fields and were lower than 2 mg/L in all SRC fields (in 14 of 16 were lower than 1 mg/L). These values can be compared with the upper EU limits for NO_3–N in groundwater which is equal to 11.3 mg/L. This is a confirmation of previous results suggesting that NO_3–N leaching in well-established willow SRC plantations is very low and close to zero [9,14,16] and that willow SRC could be cropped instead of other arable crops to reduce risks for nitrate leaching [1,2,17]. SRC could therefore be planted in agricultural areas of high risk for nitrogen leaching (e.g., nitrate vulnerable zones) to improve water quality and reduce eutrophication risks. NO_3–N concentrations were very low and significantly lower than in the respective reference fields even in the SRC fields receiving mineral fertilization. Therefore, the positive effects on NO_3–N leaching cannot specifically be attributed to low nitrogen input. Moreover, no effect on NO_3–N concentrations in the groundwater was observed when comparisons were made for SRC fields receiving sewage sludge, as it has been previously reported in Labrecque and Teodorescu [24] and Sagoo [25]. The above findings are therefore indicative that differences in nutrient inputs were not alone responsible for the lower NO_3–N concentrations in the groundwater in SRC, but probably other differences related to crop characteristics between SRC and reference crops contributed as well.

Table 2 Results of the ANOVA test and the mixed effects model for the concentrations of NO_3–N and PO_4–P in the fields analyzed

ANOVA	NO_3–N			PO_4–P		
Variable	*F*	*p* value	*N*	*F*	*p* value	*N*
SRC	60.047	0.000	366	3.937	0.048	366
Autumn (SRC)[a]	1.022	0.313	242	9.100	0.003	242
Autumn (Ref)[b]	8.671	0.108	124	0.005	2.222	124
Sludge (SRC)[a]	4.435	0.036	242	0.002	0.969	242
Mixed model						
Variable	Estimate	SE	*p* value	Estimate	SE	*p* value
Intercept	4.876	1.087	<0.001	0.050	0.033	0.146
SRC	−4.068	0.453	<0.001	0.039	0.018	0.028
Autumn	–	–	ns	0.051	0.019	0.007
σ_{plot}	16.500	6.363	0.010	0.013	0.006	0.026
$\sigma_{plot,\ time}$	15.379	1.167	<0.001	0.024	0.002	<0.001

SRC short rotation coppice, *Ref* reference, *Autumn* dummy variable for the measurements in autumn, *Sludge* dummy variable for fields fertilized with sewage sludge, *SE* standard error, *ns* not significant and excluded from the model

[a] Only SRC fields

[b] Only reference fields

Autumn leaching peaks were avoided when SRC was compared to other arable crops [23], and this was observed in our study as well. This can be attributed to the already established root system, which in SRC is active during autumn enabling willow trees to use nitrogen late in the vegetation season. It has been shown that willow roots remain active also during winter [33], and winter uptake of 15 N-labelled nitrogen fertilizer was found by Aronsson and Bergström [7]. Furthermore, higher water consumption from willow SRC than the other crops has been reported (Schmidt-Walter and Lamersdorf, this issue). Lower groundwater tables were observed in our fields, where on a number of occasions during autumn groundwater was present in reference pipes but not in the SRC. Less drainage water implies lower leaching losses of nitrogen ending up in the groundwater [8], and a consequent reduction of nitrogen leaching amounts. Moreover, no tillage and less frequent mechanical management are conducted in SRC compared to other arable crops resulting in less mineralization. This can explain parts of the lower nitrogen losses from SRC compared to that from other arable crops. It has been suggested that harvest of SRC fields could result in elevated NO_3–N leaching to groundwater, but no NO_3–N peaks were observed after harvest in the three fields that were harvested during the period of our study. Despite that the limited number of such observations does not allow us to broadly generalize, this comes in agreement with currently unpublished data of Aronsson (personal communication) that did not observe elevated NO_3–N concentrations after harvest in the groundwater of a SRC plantation intensively irrigated with wastewater (at rates corresponding to 150 kg N ha^{-1} $year^{-1}$. Additionally, no peaks of NO_3–N concentrations in the groundwater of SRC were observed in spring during snow-melting, which was observed in samples taken from arable crops. All the above findings indicate that

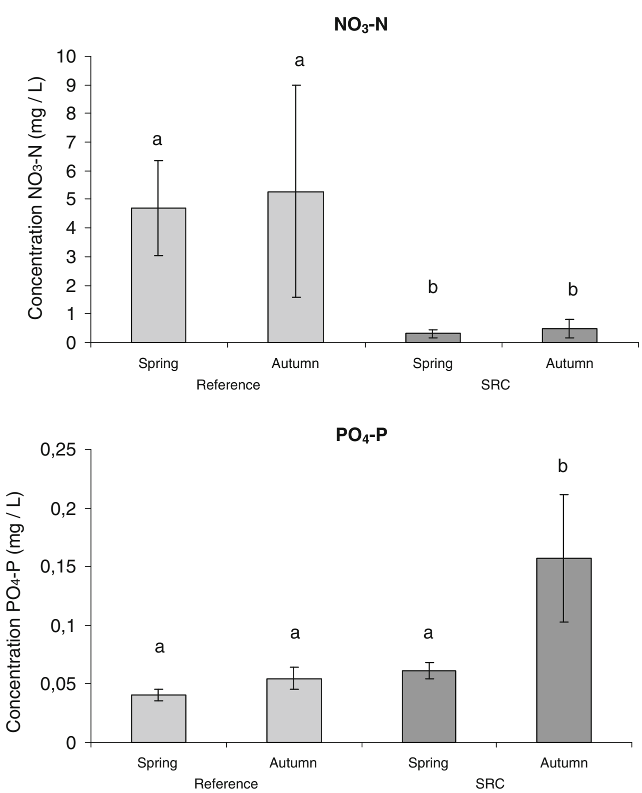

Fig. 4 Averages of NO_3–N and PO_4–P concentrations in the groundwater of all short rotation coppice (SRC) and reference fields during spring (March until June) and autumn (September until December). *Letters* illustrate the differences found, based on an ANOVA test

NO_3–N leaching is reduced when SRC is cropped instead of other arable crops, as we had hypothesized.

NO_3–N and PO_4–P leaching patterns are different, since NO_3–N leaching to drainage water is a process that is related to water percolation and NO_3–N is not bound to soil particles and is very mobile, unlike PO_4–P that is usually bound to soil particles with leaching being usually much lower proportionally than this of NO_3–N. This is depicted in our study as well, where NO_3–N concentrations in groundwater were in general considerably higher than these of PO_4–P. However, the main difference in the leaching patterns found in this study was that the average PO_4–P concentrations in the groundwater for all fields in this study were higher in SRC than in reference fields. Despite the variations between locations, the trend was rather constant and we have found that the average PO_4–P concentrations in groundwater were slightly higher in almost all SRC fields (except two) compared to the respective reference fields. This result was somewhat unexpected, did not follow our hypothesis, and contradicts to the current general impression that cultivation of willow SRC reduces phosphorus leaching compared to other arable crops as stated by several authors [1,2,32]. Low PO_4–P leaching in willow SRC fields receiving high amounts of phosphorus via fertigation with wastewater have contributed to such an impression; PO_4–P leaching was close to zero in Werner and McCracken [19] for a SRC field in N. Ireland and in Dimitriou and Aronsson [8] in clay soil in Sweden.

The differences between SRC and reference fields of PO_4–P concentrations in the groundwater could not be directly attributed to the parameters available for comparisons (e.g., reference crop, soil texture, mineral fertilization, sewage sludge/wood-ash amendments), as it must be taken into account the limitations of the data. General patterns of PO_4–P leaching to the groundwater is difficult to find [28,33], but since the results for PO_4–P concentrations in the groundwater in almost all of our SRC fields are consequently higher than of reference crops, we need to look into differences in crop characteristics for possible explanations. Carlander et al. [34] found that macropore flow through a clay soil column grown with willows was occurring in contrast to soil columns with bare soil. In our study, most of our fields were clay soils and preferential flow of soil particles with bound phosphorus particles could have been further facilitated due to willow root channels in comparison to other arable crops. Preferential flow of phosphorus via root channels and desiccation cracks has been described as the dominant pathway in flat agricultural lands with shallow water tables in northern Europe [33,35,36], which is the case in the fields we used in our study. According to Rytter [37] and Crow and Houston [38], most of the willow root system is concentrated in the topsoil, but ca. 5–25 % of fine and coarse roots can be found in deeper soil layers. Therefore, preferential flow of phosphorus might be an explanation to the differences of PO_4–P concentrations in the groundwater between SRC and reference crops. SRC has been reported to increase the soil organic matter compared to adjacent arable fields mainly due to leaf litter decomposition and increased fine-root turnover [39,40]. Increased organic matter implies elevated phosphorus amounts in the topsoil and this might be a reason for the elevated phosphorus in the groundwater in SRC stands, but phosphorus mineralization patters are difficult to predict [41]. Therefore, it is rather unsafe to speculate about the reasons behind the elevated PO_4–P concentrations in the groundwater of SRC based on our results, since there are a number of factors regarding PO_4–P leaching that might interfere. Similarly, the elevated autumn peaks in the groundwater of SRC were difficult to explain. It is also rather difficult to estimate the effect of the elevated phosphorus leaching to the groundwater on the total effect on eutrophication in a certain area. Surface flow of phosphorus, that might be reduced in SRC compared to other arable crops since it is related to soil erosion, is usually considered as most important for total phosphorus losses at a certain site [28,35].

Fears that the supply of phosphorus with sewage sludge applications would cause elevated phosphorus leaching to the groundwater in SRC fields were not confirmed. There were six fields that had received sewage sludge more than once in our study, and the higher PO_4–P concentrations in the groundwater of SRC compared to reference fields did not occur due to such applications. In the fields studied, most of the farmers did not apply sludge each time after harvest, although sludge application to SRC fields is a practice that has lately gained more interest among farmers [12]. The relatively high total amounts that can be applied to a SRC field in comparison to the annual uptake in willow plants can be a matter of concern, although compared to the phosphorus amounts allowed to be applied with sewage sludge in other countries, the amounts applied in Swedish SRC fields are rather moderate [42,43]. Indicatively, if a field is 15 years old and sludge is applied every time after harvest, then, a maximum of 535 kg phosphorus per hectare can be applied, which is much higher than the phosphorus needs and expected uptake in willow plants and outflow with harvest (if 1 mg/kg the phosphorous willow shoot concentration and annual biomass production of 10 t dry matter per hectare, then 150 kg of phosphorous would be harvested in 15 years). However, our results from the fields amended with municipal sewage sludge come into agreement with Kostyanovsky et al. [44], Samaras et al. [45], Qiang et al. [46], and Shepherd and Withers [47] who did not find any correlation between increased sludge supply and phosphorus leaching when testing for a range of climates, soils, and crops. This was mainly attributed to the fact that phosphorus in sludge is bound to organic matter and is only slowly released.

Finally, the data showed broad variations between fields in both NO_3–N and PO_4–P, that can be attributed to various and confounding factors. Although the present paper has analyzed the overall effects of SRC on water quality, focusing and estimating the differences in leaching of nitrogen and phosphorus to groundwater compared to adjacent arable fields, there are obvious limitations in the data, including the number of fields available for study, that preclude more solid conclusions regarding additional factors such specific reference crops, age of the plantations, or soil textures. In this sense, future research should be focus on these potential factors affecting inter-field variability that can complement the results found in this study.

Conclusions

The main conclusions of our study, in which comparisons of groundwater quality of a number of "old" commercial SRC with adjacent arable fields were conducted, were that:

- NO_3–N leaching from willow SRC fields was significantly lower than that from reference fields.
- NO_3–N leaching in SRC fields was not elevated during autumn or spring when most leaching from commercial fields occur or after harvest.
- PO_4–P concentrations in the groundwater of SRC were higher compared to reference fields.
- Sewage sludge applications were not responsible for the higher PO_4–P concentrations in the groundwater of SRC compared to reference fields.

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Biomass Production with Willow and Poplar Short Rotation Coppices on Sensitive Areas—the Impact on Nitrate Leaching and Groundwater Recharge in a Drinking Water Catchment near Hanover, Germany

Paul Schmidt-Walter ·
Norbert P. Lamersdorf

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Abstract In a lowland drinking water catchment area, nitrate leaching as well as groundwater recharge (GWR) was investigated in willow and poplar short rotation coppice (SRC) plantations of different ages, soil preparation measures prior to planting and harvesting intervals. Significantly increased nitrate concentrations of 16.6±1.6 mg NO_3-N L^{-1} were measured in winter/spring 2010 on a poplar site, established in 2009 after deep plowing (90 cm) but then, subsequently decreased strongly to below 2 mg NO_3-N L^{-1} in spring 2011. The fallow ground reference plot showed nitrate concentrations consistently below 1 mg L^{-1} and estimated annual seepage output loss was only 1.36±1.1 kg ha^{-1} a^{-1}. Leaching loss from a neighboring willow plot from 2005 was 14.3±6.6 kg NO_3-N ha^{-1} during spring 2010 but decreased to 2.0±1.5 kg NO_3-N ha^{-1} during the subsequent drainage period. A second willow plot, not harvested since its establishment in 1994, showed continuously higher nitrate concentrations (10.2±1.7 NO_3-N L^{-1}), while a neighboring poplar plot, twice harvested since 1994 showed significantly reduced nitrate concentrations. Water balance simulations, referenced by soil water tension and throughfall measurements, showed that at 655 mm annual rainfall, GWR from the reference plot (300 mm a^{-1}) was reduced by 40 % (to 180 mm a^{-1}) on the 2005 willow stand, mainly due to doubled rainfall interception losses. However, transpiration was limited by low soil water storage capacities, which in turn led to a moderate impact on GWR. We conclude that well-managed SRC on sensitive areas can prevent nitrate leaching, while impacts on GWR may be mitigated by management options.

P. Schmidt-Walter (✉) · N. P. Lamersdorf
Büsgen-Institute, Soil Science of Temperate Ecosystems,
Georg-August-Universität Göttingen,
Büsgenweg 2,
37077 Göttingen, Germany
e-mail: pschmid1@gwdg.de

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Introduction

To combat climate change and improve security of energy supply, bioenergy derived from forestry and agriculture plays a key role in the European Union (EU). Bioenergy production has almost doubled in production in the last 15 years and currently supplies 7 % of the total EU primary energy [1]. According to the binding targets set by the EU Renewable Energy Directive (RED), all Member States should strive to a 20 % share of renewable energy by 2020. Furthermore, it is required that EU member states achieve at least a 10 % share of renewable energy (biofuel) of the total gasoline and diesel consumed in the transport sector by the year 2020 [2].

Bioenergy crops from agriculture provide the largest potential to fulfill those EU targets. An assessment made by the European Environment Agency found that about 85 % of the potential bioenergy supply can be produced by only seven member states (Spain, France, Germany, Italy, UK, Lithuania and Poland; [3]). To achieve these goals, approximately 17.5 million ha of land will have to be dedicated to the production of energy crops by 2020 [4]. Thus, an additional pressure on farmland biodiversity as well as on soil and water resources can be expected in biofuel production regions in the EU.

In Germany, approximately 2.3 million ha or 19 % of the crop land is already being used for the production of renewable raw materials [5]. Compared to 2001, the area has almost tripled and in 2011 the largest proportion of about 2 million ha fell to the energy plant production with a share of 46 % for biodiesel (mainly canola), 41 % for biogas (mainly maize) and 13 % for the production of bioethanol. Although perennial energy crops like short rotation coppices (SRC) with fast-growing trees have played only a minor role in bioenergy production, the total cultivated area for SRC increased from about 4,000 ha in 2010 to about 5,000 ha in just 1 year [5].

Nevertheless, SRC may provide unique ecological services that warrant consideration. As a result of lower fertilizer requirements as well as a higher N-use efficiency due to effective N-recycling, SRC emit 40 to >99 % less N than conventional annual crops. Furthermore, SRC have the potential to sequester additional carbon (0.44 Mg soil C ha^{-1} $year^{-1}$) in soils if established on former cropland [6]. According to Djomo et al. [7], SRC yielded about 14–86 times more energy than coal per unit of fossil energy input and greenhouse gas (GHG) emissions were 9–161 times lower than those of coal. Consequently, SRC provide an opportunity to reduce dependency on fossil fuels and to mitigate GHG emissions. Therefore, SRC should be part of an overall strategy for achieving the minimum target for GHG emissions reduction as required by the EU RED [7]. Additionally, SRC may also increase agricultural income diversification, enhance biodiversity, and reduce nutrient losses to the groundwater [6].

However, as the area requirements for bioenergy feedstock production increases, the pressure on marginal sites or fallow grounds with unfavorable site conditions may increase and SRC systems applied here may also have negative environmental impacts. Most importantly, it seems that SRC plots have to be optimally prepared by plowing to guarantee weed control during crop establishment [8–10]. Especially on fallow grounds, this may lead to an extra emission of CO_2 and N_20. According to Djomo et al. [7], these impacts depend on various factors such as the SRC cultivation practice, land management, site conditions, downstream processing and distribution routes. Furthermore, indirect impacts have to be considered. For instance, N_2O emissions as a direct impact may be low on well-drained and well-aerated soils, but NO_3 leaching may occur instead and contaminate adjacent water bodies [6, 11, 12].

Accordingly, the given study is focusing on such indirect emission effects, i.e., the risk of nitrate leaching during the establishment of SRC plantations, and the potential of nitrogen binding after the cultivation of SRC on fallow ground.

Our study site was located in the most important drinking water catchment area of the city of Hanover, Germany ("Fuhrberger Feld"). Here, much effort was spent by the water authorities during the last decades to keep the average seepage nitrate concentration on the catchment level below the legal drinking water threshold value of 10.3 mg NO_3-N L^{-1}. Part of these efforts were voluntary agreements with resident farmers to reduce fertilizer applications to a minimum, but also many fields were set aside to lie fallow.

The enhanced nitrate concentrations in the seepage output of Fuhrberger Feld are linked to the prevailing periglacial and sandy soils in the area and the historical land use. With the formerly widespread heath plaggen fertilization process, high amounts of carbon were brought into the sandy soils [13]. In combination with originally high groundwater levels and pasture as predominant land use, high amounts of soil organic carbon accumulated in the topsoils of these areas. As long as these site conditions persisted (i.e., no changes in the water table or the grassland cover) those carbon stocks remained more or less stable. However, since the 1960s, a significant increase of the drinking water demand of the city of Hanover lowered the groundwater table considerably, with the result that the wet grassland fell dry. This lowering of the groundwater table and the subsequent transfer of grassland into intensively used arable land initiated a strong mineralization process, including the transfer of organic N to nitrate. According to model calculations of Springob et al. [14] and Springob and Kirchmann [15], it was estimated that it might take up to 100 years for the soils to achieve a new equilibrium under the present conditions. Under these conditions, Köhler et al. [16] concluded that the only way to reduce the N output to groundwater is to convert the arable land into forests or back into continuous grasslands, while setting aside the land will not reduce the risk of nitrate leaching in the long run. However, another promising land-use for fallow grounds to meet the requirements of groundwater protection might be the establishment of SRC with fast-growing trees like poplar or willow.

The desired positive effect of reducing nitrate leaching losses by SRC might come along with negative effects on groundwater quantity, as higher rates of transpiration and interception evaporation can be anticipated [17–24]. In a review [25], Dimitriou et al. summarized that groundwater recharge rates from SRC stands in general are expected to be lower when compared to arable fields or grassland in the same region. Moreover, there is indication that the amount of reduction beyond precipitation strongly depends on site-specific conditions like soil type, occurrence of drought periods during the growing season and management practices such as the harvesting interval.

Biomass production by SRC might thus conflict with the assigned land use purpose in Fuhrberger Feld, i.e., to provide and guarantee adequate amounts of good quality drinking water for the city of Hanover.

Within this context, the objectives of our study were to evaluate the impact of SRC cultivation on (1) nitrate leaching losses and (2) groundwater recharge, by giving initial insights into basic soil background conditions, seepage nitrate concentrations and water budgets of four SRC stands (two willow and two poplar SRCs) in comparison with a fallow ground reference site. The studied SRC stands differ in stand age, soil preparation measures and harvesting regimes. Water balance components are calculated by applying a soil vegetation atmosphere transport model (CoupModel [26]) to the reference site and a willow SRC established in 2005. The simulations are parameterized with results from field measurements; the model performance is checked by observed soil water tensions and stand precipitation. Finally, we estimate nitrate seepage output rates by combining simulated drainage flux with measured nitrate concentrations. The following questions will be addressed: Is a change in land use from fallow ground to SRC associated with increased N leaching rates? Which factors control N leaching rates? Under present site conditions, does a probable reduction in groundwater recharge interfere with the production of drinking water? How do site and vegetation characteristics affect the water balance and what management options do we have to mitigate a negative impact on groundwater recharge?

Material and Methods

Site and Research Plot Description

The Fuhrberger Feld drinking water catchment is located in northwest Germany, approximately 30 km north of the city of Hanover and has a size of 308 km^2. Within this catchment area, research plots are located northwest of the village Fuhrberg (52° 36′N, 9° 51′E) in a level 2 drinking water sanctuary [27] at an elevation of 41 m asl. Average annual precipitation (1971–2000) is 670 mm, of which 46 % fall during the growing season (May–October) and the mean annual temperature is 9.2 °C.

Table 1 shows the basic site characteristics of the research plots. There are two older poplar and willow plots, planted in 1994 (P94/$_{30}$ and W94/$_{30}$), a younger willow plot from 2005 (W05/$_{90}$) and a poplar plot from 2009 (P09/$_{90}$). A set-aside fallow ground serves as reference site (Ref). Prior to planting, P94/$_{30}$ and W94/$_{30}$ were conventionally plowed to 30 cm soil depth. The former organic topsoil horizon (Ah) was changed to an Ap (plowed) horizon, while the rest of the horizon sequence remained unchanged and comparable to the reference site (Ref; *E*=eluviation horizon due to Plaggen fertilization [13], Bv-rGo=typical cambisol horizon, including indications of relict (r) reduced and gleyic conditions (Gor, Gr) due to a former higher groundwater table, mixed with the Cv horizon).

Prior to SRC cultivation from cuttings, plots Ref, P09/$_{90}$ and W05/$_{90}$ were part of one single arable field, which was set aside for groundwater protection reasons in the early 1990s. The Ref plot is dominated by grasses and some scattered flowers, due to the long time of abandonment. Plots W05/$_{90}$ and P09/$_{90}$ were deeply plowed to a maximum soil depth of 90 cm. As a result, the Ah layer and its seedbank, as found on Ref, was buried at a depth of 30–60 cm (R2+E horizon) and covered with sandy bedrock material (R1). The site preparation allowed willow cuttings a headstart over the competing grasslayer, which is now present in the field. In 2010, the tree mortality rate on W05/$_{90}$ was 1.7 % [28]. During the study period, several plots were harvested. In March 2010, W94/$_{30}$ was coppiced for the first time since its establishment in 1994, in April 2011 W05/$_{90}$ followed. P94/$_{30}$ was cut in March 2011 for the second time after February 2006.

Collection of Soil Solution and Soil Samples

Field installations for soil solution sampling, the sampling process itself, inclusive storage, transport and pre-treatment of the soil solution before the laboratory analysis were in line with the ICP IM manual (2004) for soil water chemistry [29].

Accordingly, six suction lysimeters per plot (polyvinyl chloride (PVC) pipe, 95×2 cm, connected with a *P-80* cup, 5×2 cm, CeramTec Ag, Marktredwitz, Germany) were installed in November 2009, below the main rooting zone in 100 cm soil depth. After predrilling with a slightly smaller auger than the suction cups, the lysimeters were pushed directly into the soil, without applying additional active filling material. Lysimeters were evenly distributed in and between the tree rows. The soil solution was gathered in evacuated (max. 0.6 bar) 1-L glass vials, each connected via buried PVC tubing to one suction lysimeter and placed in a buried cool box next to the lysimeter field. Following the ICP manual (2004), soil solution was sampled bi-weekly to monthly. For transport and storage, solution samples were transferred into 100-ml PVC bottles and immediately stored in dark and cool conditions with a maximum temperature of 4 °C. As samples should be analyzed for all major anions and cations and to avoid any analytical interference, no preservative was added prior to the analysis, which was done within the following month after field sampling. Due to relatively dry soil conditions between June and November 2010, no soil solution could be extracted during this time.

Soil samples for physical and chemical analysis were taken from one soil pit per field plot. From each horizon, three single samples were taken to determine chemical properties. On the research plots that were chosen for water balance simulations (Ref, W05/$_{90}$), volume intact soil samples were taken using steel cylinders (100 cm³) to determine soil physical properties of the soil horizons. At W05/$_{90}$, five replicate samples in soil depths of 5, 15, 25, 35, 65 and 100 cm were taken, to account for the heterogeneous soil profile caused by soil preparation measures. At the more homogenous profile of Ref, three replicate samples in depth 15, 45 and 100 cm depth were taken.

Table 1 Basic site and soil type background conditions of the research plots in the Fuhrberger Feld (SR/DR=single/double row; 2/0.8×0.6 m=2 m between, 0.8 m within DR; 0.6 m within SR)

Plot	Genus Clone Plantation spacing Age (2011)	Soil treatment before planting	Soil type, soil horizon sequence, depth (cm)	Harvesting interventions until 2011
$P09/_{90}$	Poplar Mixture of clone *Max*[a] *1–3* DR (2/0.8×0.8 m)[b] 2	Deep-plowing up to 90 cm soil depth with a double-blade plowshare (i.e., transfer of the humic top layer (30 cm) to a depth of 30–60 cm)	Young Treposol R1 (0–30) R2+E (30–60) rGor (60–80) rGr (>80)	0
$P94/_{30}$	Poplar Row mixture of 18 clones SR (0.5×2 m)[b] 17	Conventional plowing up to 30 cm soil depth	As Ref. lAp (0–30) E (30–40) Bv-rGo (40–80) Cv-rGor (>80)	2 (winter 2005+spring 2011)
$W05/_{90}$	Willow Clone *Tora*[a] DR (1.5/0.8×0.6 m)[b] 6	as $P09/_{90}$	Older Treposol as Ref	1 (spring 2011)
$W94/_{30}$	Willow *S. viminalis* DR (1.5/0.9×0.5 m)[b] 17	As $P94/_{30}$	As Ref	1 (spring 2010)
Ref	Grassland	Abandoned cropland since the early 1990s, i.e., today covered with a grass layer	Plaggenesch-Cambisol over Relictgley Ah (0–30) E (30–40) Bv-rGo (40–80) Cv-rGor (>80)	–

[a] Clone *Max*=*Populus nigra*×*P. maximovizcii*; Clone *Tora*=*Salix schwerinii*×*S. viminalis*

[b] Fuhrberger Feld

Laboratory Analysis

The pH was measured on dried (40 °C, >48 h) and sieved (≤2 mm) soil samples using a digital pH/conductivity meter at a soil to water ratio of 1:2.5 (WTW GmbH Weilheim, West Germany). Total organic carbon (C_{org}) and total nitrogen (N_t) from mineral soil samples was measured from dried (40 °C, >48 h) and grounded samples using a C-N analyzer, (CHN-O-Rapide, VarioEL, Elementar, Germany). Our detection limit for total N is ≤0.2 mg g^{-1} and for total C ≤0.1 mg g^{-1}. The C/N ratio was calculated from the obtained C_{org} to N_t values.

The mineral N content (N_{min}, NH_4^++NO_3^-) was detected after extraction with 0.5 M K_2SO_4. NH_4^+ and NO_3^- were determined by using continuous flow injection colorimetric (Cenco/Skalar Instruments, Breda, The Netherlands). NH_4^+ was determined using the Berthelot reaction method (Skalar Method 155-000), NO_3^- in the K_2SO_4 -extract as well as in the soil solution was determined using the copper–cadmium reduction method (Skalar Method 461-00). Total dissolved nitrogen (TDN) in the K_2SO_4 extract was analyzed by the given nitrate method after NH_4^+ and organic N compounds were converted by an alkaline persulphate and UV digestion to NO_3^-. Dissolved organic nitrogen (N_{org}) was computed as: N_{org}=TDN−(NH_4^+N+NO_3^-N).

All soil water laboratory analyses were applied in line with the aforementioned ICP Manual (2004; here section 8, Data Quality Assurance and Management [29]) and soil solution nitrate analysis was cross-checked by the correlation of NO3-N+NH4-N to total N (R^2=0.985). Furthermore, quality control of our laboratory analysis are regularly applied by the integration of internal standards, replicate measurements and the contribution to external ring analysis (e.g., [30], Lab Code A56 [31] Lab No. 44).

Soil water retention characteristics were analyzed for the $W05/_{90}$ and Ref using the soil cores placed on a pressure membrane apparatus. Volumetric water contents were determined at pressure heads of pF 1.0, pF 1.5, pF 1.8, pF 2.0, pF 2.3, pF 2.5, pF 3.0, pF 3.3, pF 3.5, pF 3.7 and pF 4.2.

The grain size distribution of the fine soil was determined gravimetrically after oxidization of organic carbon using H_2O_2, destruction of binding components using Na–dithionite, following the method of Atterberg [32].

Meteorological Variables

A climate station was set up on an open field approximately 150 m from the $W05/_{90}$ plot and 200 m from the Ref plot, to collect meteorological data to be used as input variables for our water balance simulations. Sensors were mounted on a 10 m tall mast and read out by a datalogger (Dl2e, Delta-T Devices). Data were collected every 5 s, then aggregated at 10-min intervals. We measured precipitation at 1 m height (tipping bucket 0.1 mm, Thies Clima, Göttingen Germany), air temperature, relative humidity (both HMP45D, Vaisala, Vantaa, Finland), global radiation (SP Lite, Kipp & Zonen, Delft, The Netherlands) at 2 m height and wind speed (cup anemometer, Thies Clima) at 10 m height. For the use as model input, the data were checked for plausibility and later aggregated to hourly values. Data gaps in the time series due to equipment failure were filled with values from two near-by monitoring stations run by the German Weather Service. Precipitation values originate from a station about 4 km west of the field plots, while wind speed, relative humidity, air temperature and global radiation were taken from a station 15 km southwest from the field plots. Relative humidity and windspeed were adjusted to our site conditions by scaling daily mean values using linear relationships between our measurements and station data.

In 2010, precipitation was 651 mm, with 351 mm falling during the growing season (May–October). The annual sum for 2011 was 662 mm, of which 409 mm fell during the growing season. In 2010, a drought occurred in June and July, followed by a very wet period in August and September. The year 2011 was characterized by a cool and moist summer and a very dry and warm autumn.

Measurements of Soil Water Tension and Stand Precipitation

Soil water tensions to evaluate the simulation model performance were measured on the Ref and $W05/_{90}$ plots at depths of 30 cm (n=3), 60 cm (n=3) and 100 cm (n=10), using tensiometers (ceramic: *P-80*, CeramTec Ag, Marktredwitz, Germany) equipped with pressure transducers (PCFA6D, Honeywell; Morristown, NJ, USA). Pressure heads at 100-cm depth were recorded in hourly intervals from December 2009 using dataloggers (DL2 and DL2e, Delta-T Devices, Cambridge, UK). Monitoring of shallower soil depths began in May 2010. The tensiometers at 100-cm depth were placed in two parallel transects with a distance of 1 m between and within transects, crossing plant rows with an angle of 45°. The probes were installed at an angle of 30° to the soil surface in order to prevent preferential water flowing down the shaft of the instrument. Data quality assessment of water tension time series was done following the protocol described in Wegehenkel (2005) [33]. Average soil water tensions were excluded from the model performance evaluation for periods where the values of one or more tensiometers had to be rejected, i.e., because dry soil conditions beneath the measuring limit (−850 hPa).

Stand precipitation measurements were conducted on $W05/_{90}$ during the vegetation period 2010 using two 4 m long gutters with a width of 0.16 m (0.65 m^2) made from stainless steel. The gutters were mounted on an 80 cm high wooden rack, water was collected in barrels (30 L) that were emptied when necessary, though at minimum every second week.

Vegetation Characteristics

On the plots chosen for water balance simulations (Ref, $W05/_{90}$), important vegetation characteristics like leaf area per unit ground area (leaf area index, LAI), canopy height and vertical root distributions were surveyed for the use as model input. Information on vertical root distributions came from Punzet (2011, unpublished). Canopy height was measured after the growing seasons using a measuring rod. LAI was measured with a Sunscan light interception probe (SS1, Delta-T Devices, Cambridge, UK) at three dates per growing season. As recommended by the manual of the probe, the measurements were conducted on days with stable light conditions, either on bright and sunny days or on days with a uniform overcast sky. On each measuring campaign, 50 readings were taken on fixed transect points inside the canopy. In order to avoid boundary effects, all measuring points were more than three tree lengths away from the stand edges. Before each 10 readings, the probe was referenced by measuring incident radiation outside the canopy on an open field, as well three tree lengths away from the stand edges.

Statistical Analysis

All soil properties and nitrate concentration data were checked to satisfy the conditions of normal distribution (Chi Quadrat test) homoscedasticity of residuals (Levene's test) prior parametric testing. However, critical values ($p \leq 0.05$) indicated a non-normal distribution and unequal variances of the soil properties data set. Thus, the non-parametric Kruskal–Wallis analysis of variance approach was used to find significant ($p \leq 0.05$) differences between chemical parameters of plots for identical soil horizons (Table 4). Statistics on soil properties were applied using the software package *STATISTICA*, Version 9 (StatSoftGmbH, Hamburg, Germany).

Nitrate concentrations of the soil solution were analyzed using a linear mixed effects model [34], to account for the sample-point identity of measurements. The full model included the effect of the research plot, the drainage period (level "A": spring 2010 and level "B": winter/spring 2010/11) and their interaction effect. Sample point and sampling date were treated as random effects. Model comparisons were done using Akaike's information criterion [35] and likelihood ratio tests [34], with the conclusion that the sampling date could be excluded from random effects. Diagnostic plots were used to check normality and homoscedasticity of residuals and proved no severe violation of assumptions. For identifying differences between nitrate concentration means on the plot and drainage period level, all orthogonal contrasts were specified. The original level of significance (α=0.05) was adjusted to account for multiple comparisons of plots and periods. The analysis was conducted using the NLME package [36] provided by the statistical software R [37].

For evaluating the performance of the water balance simulation model, the coefficient of determination (R^2) of a linear regression between simulated and observed values, the root mean square error (RMSE) and mean error (ME) were used as objective measures. The ME quantifies the mean absolute difference between simulated and observed values, RMSE is calculated as the square root of the mean squared difference between simulated and observed values.

Simulation Model

Model Description

The CoupModel (Version 3.0 [26]), formerly known as SOIL model, was used to estimate the components of the water balance of the Ref (grass cover) and W05/$_{90}$ (willow canopy) plots. The CoupModel is a physical process model that simulates one-dimensional heat and water flows through a layered soil profile, which is covered with vegetation. It produces—after adjustment of soil and vegetation properties to site conditions—reliable estimates of evapotranspiration, groundwater recharge and other variables that are difficult to monitor in the field. In the past, it was successfully applied and verified on willow SRC stands [24, 38], crop production systems [39], forests [40, 41] and grass land sites [42]. Soil water flows are calculated by solving Richard's equation for saturated and unsaturated flow. This approach requires the hydraulic properties of the soil layers, that are described by the formulations of Brooks and Corey [43] (retention characteristics) and Mualem [44] (hydraulic conductivity). Richard's equation allows for soil water sources and sinks, i.e., root water uptake driven by transpiration. Potential transpiration (T_p), interception evaporation (E_i) from wet plant surfaces and soil evaporation (E_s) are calculated separately for one or more canopy layers and the soil surface using the Penman–Monteith combination equation [45]. Actual transpiration as the sum of root water uptake from soil layers is calculated on the basis of potential transpiration, which is reduced by taking actual soil water availabilities, soil temperatures and root densities of the soil layers into account.

Simulation Setup and Parameterization

The simulations of the Ref and W05/$_{90}$ plots were run with hourly resolution from January 2009 (initial soil water tension of all layers, −60 hPa) until the end of December 2011, driven by the meteorological input data set. The period of interest includes the years 2010 and 2011, for which daily output of water balance components and state variables were produced. For both simulations, soil profiles with 20 layers and a total depth of 2.55 m were defined. The thickness of the soil layers gradually increased from 5 cm in the uppermost 35 cm to 30 cm in the two deepest layers. Upper and lower boundary conditions were defined as flux boundaries with the upper boundary taking the stand precipitation into account. As lower boundary condition, a unit gradient gravitational water flow was setup, which in this study represents groundwater recharge. Capillary rise was not considered.

For the description of the physical and physiological properties of the willow canopy (i.e., stomata and aerodynamic resistance functions according to [46, 47]), we used the parameterisation of Persson and Lindroth [38] (Table 3). They simulated evapotranspiration rates of a willow stand on clay soil using the older version (SOIL) of the CoupModel and verified the model with measured stand evapotranspiration. R^2 ranged from 0.73 to 0.79, the model only slightly overestimated evapotranspiration during two growing seasons by 2 and 10 mm [38].

Vegetation (Table 3) and soil characteristics (Table 2) were chosen to represent our site conditions with respect to measured LAI, canopy height, root distribution and soil hydraulic properties. Model LAI development during the growing seasons was defined to match the measured values on W05/$_{90}$. For estimating the actual dates of budburst and leaf fall, we applied a critical day length and temperature sum model [26]. A maximum stand average LAI of 4.2 m^2 m^{-2} was reached in June 2010. In the following month, LAI dropped to about 2 m^2 m^{-2}, likely due to enduring water scarcity. In 2010, average canopy height of the willow stand was 7.5 m. After harvest in early 2011, the stand re-sprouted to about 3 m during the growing season. The maximum LAI (3.8 m^2 m^{-2}) of the growing season 2011 was reached at the end of August.

The observed root distribution was in good accordance to other reported observations [48] in SRC stands. Most fine roots (80 %) were found in the upper 60 cm of the soil

Table 2 Soil hydraulic properties used in the water balance simulation for the plots Ref and W05/$_{90}$

Plot	Depth (cm)	Lambda (–)	Air entry (hPa)	Saturation (vol%)	Wilting point (vol. %)	Residual water (vol%)	Matrix cond. (mm day^{-1})	Total cond. (mm day^{-1})	Tortuosity (–)
Ref	0–35	0.196	5.02	52.0	10.3	4.2	2,000	2,000	1
	35–55	0.478	1.84	44.1	6.3	2.0	5,000	5,000	1
	55–255	0.816	0.82	38.8	3.8	6.7	10,000	10,000	1
W05/$_{90}$	0–10	0.464	1.98	46.4	3.6	6.9	1,070	1,070	1
	10–20	0.204	7.75	55.6	9.5	6.6	8,100	8,100	1
	20–35	0.246	8.36	55.4	10.9	17.2	5,400	5,400	1
	35–55	0.118	1.71	51.0	9.9	3.9	740	740	1
	55–70	0.518	1.91	48.9	6.3	13.3	2,350	2,350	1
	70–255	0.816	5.62	38.8	3.8	6.7	10,800	10,800	1

profile. Below the R2+E horizon, roots sporadically occurred down to a depth of 180 cm.

The parameters of the retention function (Table 2) were obtained by least squares fitting of observed water content/pressure head points of the horizons. Hydraulic conductivity functions were derived from the grain size distributions using built-in CoupModel routines. In spring 2011, soil water tensions in 30 cm indicated root water uptake, before the recently harvested willow stand had developed new shoots. To account for this water uptake, a grass layer was defined underneath the willow canopy. This grass layer was assumed to have the same properties as the grass layer defined in the simulation of the Ref plot (Table 3), except for the maximum LAI, which we assigned a lower value (3 m^2 m^{-2}). Vegetation properties to simulate the Ref plot were taken from Lundmark [42], hydraulic properties of the soil horizons and the vertical root distribution were derived from field measurements. On the Ref plot, most fine roots (95 %) were located in the former Ap horizon, only few roots of dicot plants reached down to 90 cm.

With respect to our measured stand precipitation and soil water tension data, adjustments of the original parameter sets [38, 42] had to be carried out, to obtain a better agreement between simulated and measured variables and thus a better estimation of the water balance components. Stand precipitation measurements on the willow plot indicated underestimated interception evaporation when using the canopy storage capacity parameters (I_c, I_{LAI}) of a previous study [38], likely due to the higher temporal resolution (1 h) of our simulation. Adjustments of these parameters and the interception surface resistance (r_{ci}, Table 3) led to a better agreement between simulated and observed stand precipitation. Hydraulic conductivities derived from grain size distributions were adjusted considering the observed soil water tensions at field capacity during winter time. The RWU$_{comp}$ parameter for water uptake compensation was used to improve the agreement between observed and simulated soil water tensions during the growing season.

Table 3 Adjusted CoupModel parameter values for the simulations of the W05/$_{90}$ and Ref plots

CoupModel parameter	Unit	Willow canopy W05/$_{90}$	Grass layer Ref
Interception constant capacity (I_c)	mm	0.2	0
Interception capacity per LAI (I_{LAI})	mm	0.25	0.25
Interception surface resistance (r_{ci})	s m^{-1}	0.5	5
Reference height	m	10	2
Parameter in soil surface resistance function	hPa	1,000	1,000
Crit. threshold for water uptake reduction (Ψ_{crit})	hPa	400	400
Degree of root water uptake compensation from moist soil layers (RWU$_{comp}$)	–	0.45	0.25
Maximum stomata conductance (g_{max})	m s^{-1}	0.015	0.02
Sensitivity of stomata conductance to VPD (b)	Pa	1,318	100
Sensitivity of stomata conductance to global short wave radiation (R_0)	MJ m^{-2} day^{-1}	11.8	5
Aerodynamic resistance (r_a)	s m^{-1}	3–676	30–708
Root depth	m	1.8	0.7–1.0
Canopy height	m	0.1–7.5	0.05–0.35
Leaf area index (LAI)	m^2 m^{-2}	0–4.2	1.5–3.5
Critical air temperature for temperature sum calculation	°C	9	8.5
Temperature sum to start leaf development	°C	50	50

As in the simulation of W05/$_{90}$ plot, adjustments of the Ref parameter set included soil hydraulic conductivities and root water uptake compensation (RWU_{comp}). Additionally, the shape of the seasonal LAI development was adjusted.

Results

Basic Soil Parameters

The C_{org} and the N_t of the soil profiles clearly mirror the deep-plowing effect on the plots P09/$_{90}$ and W05/$_{90}$ (Table 4). C_{org} and N_t values are enhanced in the mid soil layers of 30–50 cm soil depth, whereas highest C_{org} and N_t values of the reference plot (Ref) and the two conventionally plowed (30 cm soil depth) poplar and willow plots (P94/$_{30}$, W94/$_{30}$) were found in the upper 0–30 cm soil depth. With more than 7 %, C_{org} content was highest in the upper 10 cm of the reference plot and lowest in the top layer of the deeply plowed P09/$_{90}$ plot (0.2 % C_{org}). Due to the soil mixture after the plowing on the SRC plots, measured soil values generally indicate a high spatial variability and thus could only be proved to be statistically different in some cases (e.g., for C_{org} and N_t, P09/$_{90}$ *versus* Ref., 0–10 cm soil depth). Because of relatively low C_{org} values in the top and lowest layer of the P09/$_{90}$ plot, C/N ratios are relatively low as well (9.9–10.9). On the other plots, C/N ratios ranged from about 11 in the lower soil horizons (Ref, 30–50 cm soil depth) to more than 26 in the upper soil horizons (P94/$_{30}$). Even if not statistically provable, there is a tendency towards relatively low C/N ratios for all soil layers on the deeply plowed P09/$_{90}$ and W05/$_{90}$ plots, which might already indicate the potential of nitrate leaching on these two plots. Results of the mineral N analysis (N_{min}) indicate a shift towards higher values only in the 10–30 cm soil layer in the P09/$_{90}$ and W05/$_{90}$ plots, while horizons below and above showed reduced values. Conventional plowing of the topsoil (30 cm) alone, as applied at the P94/$_{30}$ and W94/$_{30}$ plots did not change the vertical gradient of the N_{min} values, compared to the reference plot.

Compared to conventional cropland sites of the region N_{min} values are low. Higher N_{min} values of the top soil—respectively the former top soil in 10–30 cm soil depth on the plowed P09/$_{90}$ and W05/$_{90}$ plots—are correlating with higher values of N_{org}. Furthermore, mean percentage of nitrate in total N_{min} (NO_3+NH_4) was not detectable in the 30–50 cm soil layer at the reference plot but also not in the most upper layer of the P09/$_{90}$ plot—which in fact here is also the former deep layer, transferred by deep plowing to the top of the profile. As far as N_{min} was detectable in deeper horizons at all SRC plots, nitrate was the dominant constituent. The pH (1 M KCl) values of all plots and soil layers range between 4.1 and 5.7, with a tendency towards higher

Table 4 Mean (±SD) soil chemical background conditions of the SRC plots in the Fuhrberger Feld

Plot	Soil depth (cm)	pH (1 M KCl)	C_{org} (mg/g)	N_t (mg/g)	C/N (mg/mg)	N_{min} (NO_3+NH_4) (mg/kg)	NO_3/N_{min} (%)	N_{org} (mg/kg)
P09/$_{90}$	0–10	4.4 (0.2)	2.0 (0.2) *a*	0.2 (0.0) *a*	9.9 *a*	≤d.l.	0	2.57 (0.81) *a*
	10–30	5.0 (0.4)	13.0 (10.4)	0.8 (0.6)	14.3 *a*	7.29 (1.09)	79	6.98 (0.87)
	30–50	5.1 (0.2)	40.7 (3.2)	2.4 (0.1)	16.8	1.62 (0.00)	100	2.33 (1.08)
	50–70	4.1 (0.2)	2.3 (1.5)	≤d.l.	–	n.t.	–	n.t.
P94/$_{30}$	0–10	4.6 (0.1)	52.8 (1.9)	2.0 (0.1)	26.4 *b*	2.86 (0.86)	74	5.56 (0.48)
	10–30	4.6 (0.1)	50.5 (3.0)	1.8 (0.1)	27.6 *b*	1.62 (0.00)	100	6.78 (0.68)
	30–50	4.4 (0.1)	10.9 (4.1)	0.5 (0.2)	22.0 *a*	≤d.l.	0	2.69 (0.23)
	50–70	4.3 (0.1)	4.8 (2.2)	0.3 (0.1)	18.6	n.t.	–	n.t.
W05/$_{90}$	0–10	5.2 (0.2)	19.1 (8.4)	1.1 (0.5)	17.2	1.17 (0.00)	100	3.73 (1.14)
	10–30	5.7 (0.3)	44.7 (2.3)	2.6 (0.1)	17.5	3.67 (0.69)	84	7.21 (1.19)
	30–50	5.7 (0.2) *a*	42.8 (8.2) *a*	2.5 (0.5) *a*	17.3	0.66 (0.00)	100	2.78 (0.66)
	50–70	4.1 (0.2)	2.4 (0.4)	≤d.l.	–	n.t.	–	n.t.
W94/$_{30}$	0–10	5.2 (0.2)	53.2 (3.2)	2.2 (0.1)	24.0	2.83 (0.86) *b*	74	5.56 (1.00)
	10–30	4.9 (0.1)	47.7 (4.9)	1.9 (0.1)	25.1	1.52 (0.00)	100	6.78 (0.58)
	30–50	4.6 (0.3)	30.6 (12.9)	1.3 (0.2)	24.3	0.29 (0.00)	100	2.69 (0.70)
	50–70	4.2 (0.2)	5.6 (1.9)	0.3 (0.1)	19.4	n.t.	–	n.t.
Ref	0–10	4.4 (0.1)	71.7 (12.6) *b*	3.2 (0.6) *b*	22.6	2.40 (0.70)	75	6.72 (0.91) *b*
	10–30	4.5 (0.5)	59.3 (31.3)	2.4 (1.1)	24.3	1.76 (0.67)	67	5.47 (1.14)
	30–50	4.3 (0.1) *b*	1.9 (0.3) *b*	0.2 (0.0) *b*	11.4 *b*	≤d.l.	0	2.24 (2.12)
	50–70	4.2 (0.1)	2.1 (1.1)	≤d.l.	–	n.t.	–	n.t.

Samples were taken in June 2010 with n=3 per layer and plot (*d.l.* detection limit, *n.t.* not detected). Different letters indicate significant (p≤0.05) differences between the same soil layer of different plots values on the W05/$_{90}$ plot. However, differences proved to be statistically significant in only one case (30–50 cm soil depth, W05/$_{90}$ *versus* Ref).

Nitrate Soil Solution Concentrations

Figure 1 shows the time series of monthly mean nitrate concentrations in the soil solution at 100 cm soil depth, measured from February to May 2010 (period A) and from December 2010 to June 2011 (period B). Estimated nitrate concentration means including standard errors as well as statistical differences between means of the plots and between periods are given in Table 5. Statistical tests show that plot and period both have a significant influence ($p<0.0001$) on nitrate concentrations. The significant ($p<0.0001$) interaction between both factors suggests different plot behavior for the sampling periods.

In period A (spring 2010), there is a relatively clear sequence of the nitrate concentration levels between a mean of above 16 to below 1 mg NO_3-N L^{-1} (P09/$_{90}$>W94/$_{30}$>W05/$_{90}$>P94/$_{30}$> Ref, Fig. 1). Concentrations of Ref and P94/$_{30}$ thereby do not differ significantly from zero (Table 5a). In period B (winter/spring 2010/11; Table 5b), only the concentrations of W94/$_{30}$ remain on the same high level of period A (no significant differences between periods). Nitrate concentrations of P94/$_{30}$ in period B are as well at the very low concentration level of the reference plot, concentrations of P09/$_{90}$ and W94/$_{30}$ remain significantly higher than concentrations of Ref (Table 5). At the beginning of period B, nitrate concentrations of P09/$_{90}$ started again at an elevated level but then strongly decreased to the level of the W05/$_{90}$ plot. Finally, the estimated mean nitrate concentration of W05/$_{90}$ in period B turned out to be significantly reduced compared to period A (Table 5a, b).

Significant differences of the estimated mean nitrate concentrations in period A were found for the comparisons Ref-P09/$_{90}$, Ref-W94/$_{30}$, P09/$_{90}$-P94/$_{30}$, and P09/$_{90}$-W05/$_{90}$ (Table 5a). In period B (Table 5b), the differences between W05/$_{90}$ and P09/$_{90}$ were not significant anymore, but the differences between W94/$_{30}$ and P94/$_{30}$ became significant.

However, as already described for the soil matrix, the spatial variability of the solute nitrate concentrations is high, especially when the concentration is low. In both sampling periods, the variation coefficient is between 100 and 200 % on the P94/$_{30}$ plot, but is lower (around 14–65 %) when nitrate concentrations are enhanced (P09/$_{90}$ and W94/$_{30}$, both periods).

Simulated and Observed Pressure Heads

Figure 2a and b show the time series of the mean, minimum and maximum observed soil water tensions in 100 cm depth on the Ref and W05/$_{90}$ plots, in combination with the corresponding simulated values of the soil layer in 95–105 cm depth. The seasonal pattern is similar on both plots. During winter, all tensiometers show values around field capacity. With budburst in spring and beginning root water uptake, soil water potentials start to decrease and the spatial variability increases. Pressure heads on Ref start to decrease later, less strong and in fewer locations compared to the pressure heads of the willow plot. As a consequence, thorough rewetting of the soil on Ref is attained earlier in autumn when root water uptake ceases. In 2010, drainage formation on Ref can be expected to start already after a heavy rain storm at the end of August after which soil water potentials in 100 cm indicate field capacity. At the same time, many tensiometers on the W05/$_{90}$ plot did not work properly, indicating soil water tensions near or beyond the measuring limit (−850 hPa) since July 2010.

A similar seasonal pattern is revealed by tensiometry in summer 2011. After a series of heavy rain storms at the end of June, the soil in 100 cm depth on Ref is completely

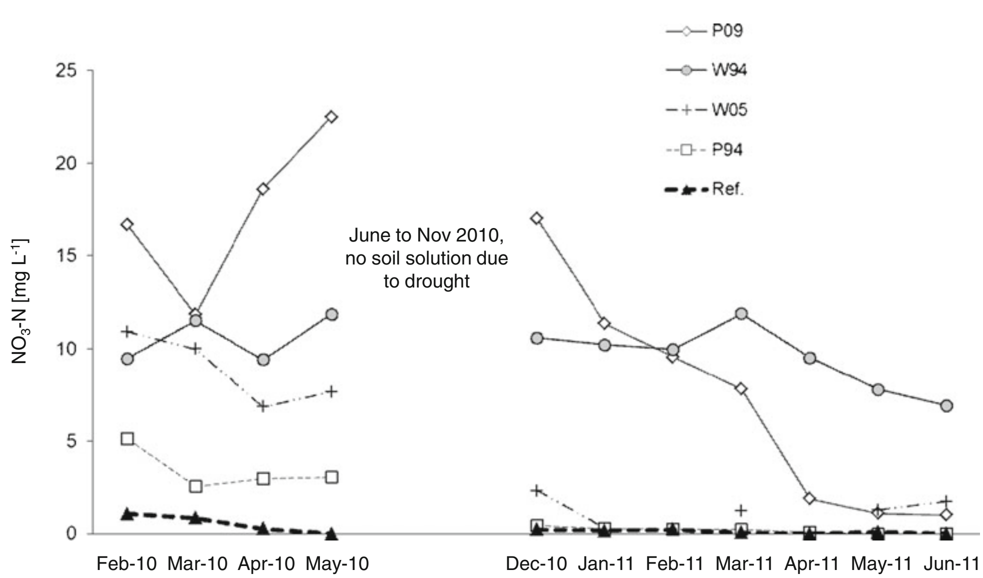

Fig. 1 Monthly mean nitrate concentrations of SRC plots in the Fuhrberger Feld at 100 cm soil depth from Feb 2010 to Jun 2011 (due to dry conditions, no samples could be obtained between June and Nov 2011)

Table 5 Estimated mean nitrate concentrations (NO_3-N L^{-1}) of the study plots, separated for period A (Feb 2010–May 2010; Table 5a) and period B (Dec 2010-June 2011; Table 5b)

Estimate *p* value SE	Ref	$P09/_{90}$	$P94/_{30}$	$W05/_{90}$	$W94/_{30}$
a)					
Ref	0.62 A,B n. sign. 1.77				
$P09/_{90}$	<0.0001	16.64 A <0.0001 1.61			
$P94/_{30}$	n. sign.	<0.0001	3.33 A,B n.sign. 1.63		
$W05/_{90}$	n.sign.	<0.0001	n. sign.	7.81 A <0.0001 1.64	
$W94/_{30}$	0.0007	n.sign.	n.sign,	n. sign.	10.38 A,B <0.0001 1.76
b)					
Ref	0.11 A,B n. sign. 1.59				
$P09/_{90}$	0.0005	9.1 B <0.0001 1.57			
$P94/_{30}$	n.sign.	0.0005	0.27 A,B n. sign. 1.53		
$W05/_{90}$	n. sign.	n. sign.	n. sign.	2.23 B n. sign. 1.75	
$W94/_{30}$	0.0002	n. sign.	0.0002	n.sign.	10.09 A,B <0.0001 1.65

On the diagonal: estimates, intercept significance levels (*n. sign.* not significant) and standard errors (SE) of estimates. Letters denote significant membership to the periods A and B. Significances for plot comparisons are specified below the diagonal. The original significance level α=0.05 was adjusted to account for multiple comparisons

rewetted and remains relatively moist throughout the rest of the growing season. Potentials on $W05/_{90}$ show only slight increases during that period. The soil stays relatively dry, though inside the measuring range until beginning of December 2011. Thus, considerable groundwater recharge cannot be expected before the end of the year.

The visible agreement between observed and simulated soil water tensions, in combination with performance indicators (Table 6) suggests an acceptable performance of the CoupModel. The important points in time like the beginning of root water uptake in spring and the rewetting in autumn meet well with observations, except for the rewetting in autumn 2010 on $W05/_{90}$. Here, the point in time of water flow breakthrough in 100 cm could not be monitored, due to measuring errors caused by the dry subsoil. However, coefficients of determination (Table 6) are relatively good for pressure head data [33] and the absolute deviation from measurements is low. The fact that measured values of soil water storage capacities (retention curves) were used in the simulations and throughfall was only slightly overestimated (ME; 1.4 mm, Table 6, possibly due to wetting and evaporation losses during measurements), strengthens our belief that the estimations of the water balance components are reliable.

Simulated Water Budgets

The monthly and annual sums of the simulated water balance components for the study plots are illustrated in Fig. 3a

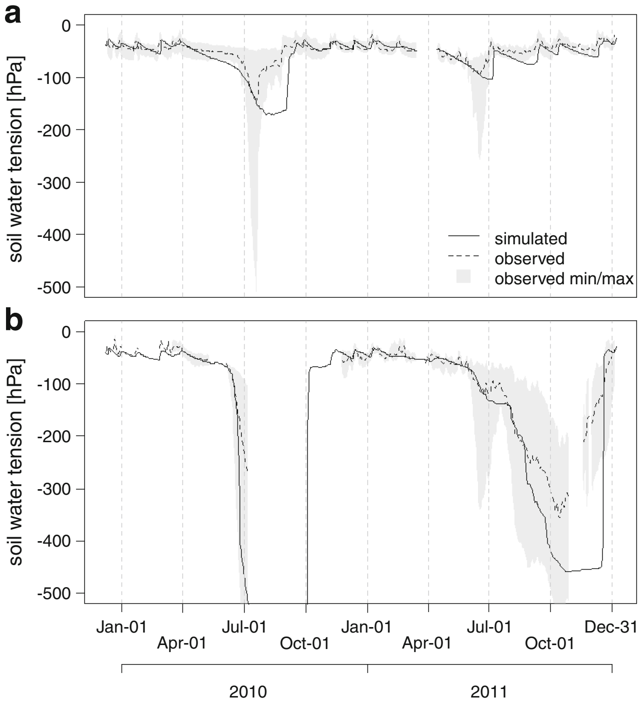

Fig. 2 **a**, **b** Mean observed (n=10) and simulated soil water tensions of Ref (**a**) and W05/$_{90}$ (**b**) study plots at 100 cm soil depths. *Shaded area* minimum/maximum of observations

and b and Table 7. Interception and transpiration of the willow plot include the evaporation rates of the grass layer beneath the willow canopy.

Similar to the seasonal variation in the soil water tensions, a clear pattern can be seen in the monthly sums of the water balance components of the study plots. Actual evapotranspiration (ET_a) as the sum of transpiration (T_a), E_i and E_s has the highest values during the growing season, whereas drainage from the soil profiles takes place mainly in winter. The transitions between these hydrological seasons are smooth, with a gradual shift towards ET_a in spring and a more abrupt beginning of the drainage period in autumn or winter. Simulated monthly E_i and T_a from W05/$_{90}$ are higher than from Ref (Fig. 3a); in summer, ET_a generally is equal or even higher than precipitation. In July 2010, T_a is extraordinary low in both simulations, due to low amounts of precipitation in June and July and pronounced soil water deficits in the root zone.

The simulated groundwater recharge from W05/$_{90}$ ceases during summer almost completely, while small amounts of drainage are formed on Ref throughout the whole observation period. On Ref, considerable monthly groundwater recharge rates are attained already in October 2010, while groundwater recharge from W05/$_{90}$ does not start before December. In 2011, the winter drainage period did not start until the end of the simulation period, whereas seepage from Ref started to increase already in October.

The differences in seasonal water partitioning patterns between the simulated land use types are reflected in the annual sums of the water balance components. Especially annual E_i and T_a are higher from the willow stand (Table 7). During the simulation period, average E_i losses from W05/$_{90}$ account for 25 % of precipitation, contrasted by 12 % on Ref. Simulated root water uptake on W05/$_{90}$ is in both years about 60 mm higher.

The differences in annual evapotranspiration rates between the land use types result in large differences in the amount of water leaving the soil profiles and serving as groundwater recharge. On Ref, 345 mm drainage formed in the year 2010 (Table 7), which is more than half of the annual precipitation. In contrast to that, 189 mm (29 % of precipitation) groundwater recharge is formed on W05/$_{90}$. In 2011, these differences are not as expressed, although the soil profile of W05/$_{90}$ was not rewetted completely (Table 7) by the end of the simulation period. For the whole simulation period, average annual groundwater recharge from W05/$_{90}$ (180 mm a^{-1}) is approximately 40 % lower than groundwater recharge from Ref (300 mm a^{-1}).

Discussion

The German Biomass Research Center [49] has calculated that by 2020, there will be a net lack of about 270 PJ per year in the German energy and material related wood market. If this “wood gap” would be filled by the establishment of SRC plantations, it would result in an extra need of about 1.2 million ha [49]. The current (2011) total cropland area in

Table 6 Performance statistics for simulated soil water tensions and throughfall. Indicators for soil water tension performance are means for three (30–35 cm; 55–65 cm) respectively ten (95–105 cm) tensiometers

Variable	Plot	Horizon (cm)	R^2	RMSE	ME	n Obs.
Soil water tension	Ref	30–35	0.67	148 hpa	−81 hpa	512
		55–65	0.52	177 hpa	−71 hpa	606
		95–105	0.49	26 hpa	−10 hpa	761
	W05/$_{90}$	30–35	0.61	299 hpa	−101 hpa	582
		55–65	0.55	218 hpa	−5 hpa	582
		95–105	0.68	185 hpa	−83 hpa	685
Throughfall	W05/$_{90}$	–	0.95	4.1 mm	1.4 mm	21

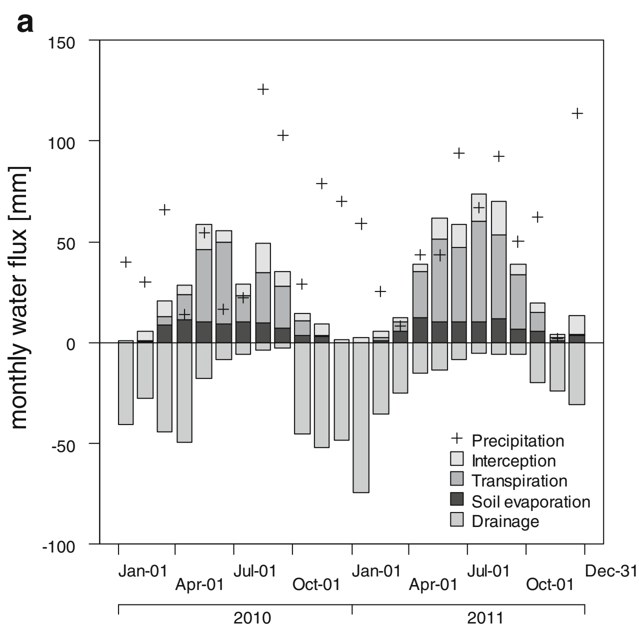

Fig. 3 **a**, **b** Simulated monthly water fluxes for Ref (**a**) and W05/$_{90}$ (**b**). Groundwater recharge (drainage) is presented by negative values

Germany is about 12 million ha, where approximately 2.3 million ha is used for bioenergy raw material production [5]. As of today, SRC contribute to only approximately 5,000 ha [50], it may be unrealistic to fill the forecasted wood gap in Germany by using only SRC.

Thus, and in accordance with Fritsche et al. [51], the cultivation of bioenergy crops on natural or semi-natural land that is currently not under specific production, is expected to increase. However, in comparison with conventional crops for bioenergy production (e.g., canola, maize), SRC may even increase ecological services on the field as well as on the landscape level [52–56]. SRC may act as physical barriers in the formation of "arable deserts" and protect against soil erosion or act as riparian or groundwater buffer strips to protect soil and water qualities in the context of the Water Framework Directive 2006/118/EU [57].

Furthermore, the cultivation of biomass on unused degraded land or on abandoned farmland as applied in this study can be seen as a safeguard against negative indirect land use change effects, as described by Fritsche et al. [51].

In the given study, the general focus of the implemented SRC is to protect drinking water resources from nitrate pollution while simultaneously producing biomass feedstock on an extensive level, i.e., without any further input of N fertilizers or other chemicals. However, as drinking water is the most important product of land use, groundwater recharge rates have to be considered as well.

According to Gadgil et al. [58], the Fuhrberg recharge zone can thus be described as an "ecologically sensitive area" which is both, ecologically and economically important, but, vulnerable to even mild disturbances and hence demands careful management. In Fuhrberger Feld, reduced nitrate loads were achieved by setting aside arable cropland as demonstrated by the low nitrate leaching measured from the reference plot (Fig. 1).

Nevertheless, is it not clear how sustainable this approach is in the long-run. The C- and N-rich topsoil horizon is a potential source of nitrate leaching coming from mineralization pulses and accordingly may be vulnerable to any disturbances such as droughts, subsequent rewetting or even fire, all of which lead to a release of organically bound N sources. Such a disturbance effect can be followed on plot P09/$_{90}$, where the site preparation measures (deep plowing) were followed by a distinct nitrate pulse in period A (Feb–May 2010). Enhanced amounts of organically bound N sources were made available and subsequently leached as nitrate to the groundwater.

However, nitrate leaching may also exhibit a high temporal variability, as evident from plot W05/$_{90}$. Here, nitrate concentrations were in the range of the drinking water threshold value of 11.3 mg NO3-N L^{-1} (i.e., 50 mg $NO3^{-}$ L^{-1} [27]) during period A, but significantly decreased to the very low level of the reference plot in period B (Dec 2010 to Jun 2011). The same trend was evident for P94/$_{30}$, though the starting nitrate concentrations were less pronounced (mean in period A, 3.3 mg L^{-1}) but also exhibited considerable variability around the mean (±3.4 SD). As the nitrate concentrations in P94/$_{30}$ were always clearly below the drinking water threshold, it can be concluded that SRC, at least in the long term, do not leach critical amounts of nitrate.

As indicated in Table 4, higher C and N contents as well as higher C/N ratios are present in the deeper soil horizons

Table 7 Simulated annual water balances of the Ref and W05/$_{90}$ study plots, given in mm and as percentage of precipitation (% P)

Water balance component	2010		2011	
	Ref mm (% P)	W05/$_{90}$mm (% P)	Ref mm (% P)	W05/$_{90}$mm (% P)
Precipitation	651 (−)	651 (−)	662 (−)	662 (−)
Interception evap.	74 (11)	168 (26)	85 (13)	168 (25)
Transpiration	161 (25)	219 (34)	236 (36)	293 (44)
Soil evaporation	73 (11)	75 (12)	80 (12)	77 (12)
Drainage	345 (53)	189 (29)	264 (40)	172 (26)
Δ Water storage[a]	−8 (−1)	−2 (−0.5)	19 (3)	−27 (−4)

[a] Compared to initial soil water storage at −60 hPa

of the P94/$_{30}$ plot. This finding might be regarded as a development towards more resilience against mineralization pulses and unwanted releases of C and N to the atmosphere and/or water sources.

As we do not know how the nitrate concentrations developed on W94/$_{30}$ since establishment in 1994, we only can speculate: a few years after SRC establishment, the biological activity of the site will increase compared to the former cropland, due to the continuous input of leaf and root litter with no respective output losses of organic material by harvesting. In addition, different vertical soil structures (such as C and N accumulations in deeper soil layers) will develop. The most obvious indication of such development is that normally no leaf litter from the previous autumn can be found in spring and C and N is higher in horizons below the plowing depth of 30 cm.

Furthermore, N sources released by mineralization processes are protected from N-leaching as long as N uptake by the vegetation cover is balancing the N release. If plant growth stops for some reason—as it was evident on plot W94/$_{30}$—but mineralization processes continue, nitrate leaching may occur. Harvesting may also stop the N uptake. But, since this is done normally during winter, when mineralization rates are low and the rootstock immediately resprouts once the weather gets warmer, nitrification pulses after harvest are not described so far, even in cases with additional N fertilization [59–61].

Comparable field data of nitrate leaching under SRC without additional N input manipulation (N fertilization, sewage sludge, waste water or compost additions) are rare. However, in one comparable study, Goodlass et al. [60] found enhanced nitrate leaching under a *Salix viminals* SRC, after the former canola field was plowed in winter and sprayed with herbicides in the following spring. Peak nitrate concentrations reached a maximum of 70 mg NO_3-N L^{-1} in spring and were even enhanced to 134 mg NO_3-N L^{-1} during the following autumn. Once the SRC was established, concentrations returned to a lower level (18 mg NO_3-N L^{-1}) and were only slightly affected by harvesting operations and annual applications of nitrogen during the first 3 years. The reference plot was an adjacent arable area where nitrate peaks ranged from 26 to 77 mg NO_3-N L^{-1} with an average value of 54 mg NO_3-N L^{-1} during the crop rotation. Thus it was concluded, that once established, the risk of nitrate leaching from SRC grown at recommended N inputs is small, especially when compared with the nitrate peaks in autumn, which are typical of arable rotations [60]. Moreover, significant losses during establishment of stands would be offset by smaller losses during the productive phase, when compared to average nitrate losses from crop production systems [60].

Land Use-Specific Water Budgets

The water balance simulations for the Ref and W05/$_{90}$ study plots revealed distinct differences in partitioning precipitation into E_i, T_a and groundwater recharge. The differences in water budget partitioning are not surprising and generally agree with the findings of Persson [24], who compared the water budgets of different vegetation types and found highest evapotranspiration rates for spruce and willow and lowest for grassland and barley. However, considering the whole observation period, groundwater recharge from the willow stand was reduced by approximately 40 % (120 mm a^{-1}) compared to the reference site. This is comparable to values for deciduous forests at comparable sites [62], but groundwater recharge from W05/$_{90}$ was higher than values reported for coniferous forests [63] located at Fuhrberger Feld. The reduction thus is smaller than the reduction reported by Persson [24] for comparable land uses and reflects the need for water balance studies including various sites and different meteorological conditions. Furthermore, studies about the water balance of willow SRC on sandy soils with low amounts of plant available soil water storage are scarce in literature and data covering the whole year and not only the growing season are even scarcer.

Groundwater recharge from the willow stand was considerably reduced in comparison with the reference plot. The main reason for this shift from groundwater recharge to evapotranspiration lies in doubled interception losses, which are caused by a closer coupling of SRC stands to the atmosphere [38]. This implies that evapotranspiration rates are mainly controlled by atmospheric vapor pressure deficit and aerodynamic resistance and less by available radiation energy [64]. Together with a higher interception storage capacity of the willow canopy, simulated interception was approximately 25 % of annual precipitation in both years. This amount lies between two extremes reported by Persson and Lindroth [38] (11 %) and Ettala [65] (31 %).

It is surprising that during both years, the willow stand intercepted the same amount of rainfall (170 mm, Table 7) at nearly the same annual sum of precipitation, despite the fact that the stand was harvested in spring 2011. Re-sprouting stands are in the first half of the growing season less well coupled to the atmosphere because of their low stand height [38]. Additionally, leaf area development is delayed compared to a mature stand. This implies a lower canopy interception storage capacity and before canopy closure, a higher amount of precipitation directly reaches the ground. Thus, it is conceivable that the harvested willow stand intercepts relatively less rainfall than a mature stand, and the lack of differences between the years is due to a different temporal distribution of rainfall. In fact, a simulation scenario that assumes a mature instead of a re-sprouting willow stand under the climate conditions of 2011 yields interception losses increased by 25 mm. In turn, a re-sprouting stand under the 2010 climate conditions has 15 mm less interception evaporation per year.

Aside from interception evaporation, transpiration rates from the willow stand are as well higher than transpiration rates from the reference site. With a deeper rooting system, the willow stand draws water from a greater soil volume, thus more water is available for transpiration. In both years, annual transpiration from the willow stand was approximately 60 mm higher compared to the reference site. This amount complies well with the surplus of plant available soil water resources. Tensiometer time series (Fig. 2b) indicate that the investigated willow SRC is able to evapotranspirate all precipitation that falls during the growing season (also see [18]) and additionally develops pronounced soil water deficits. These soil water deficits—being about 60 mm higher compared to the reference plot—in turn reduce the groundwater recharge, as the soil needs to be rewetted before drainage can take place.

A consequence of the willows' high water demand in combination with the relatively low soil water storage capacity of the sandy soils is the exposition to water stress during periods with low amounts of precipitation. This is exactly what the simulation suggested in July 2010, when transpiration collapsed (Fig. 3b) because of exhausted soil water supplies in the root zone. Typical reactions to water stress are leaf shedding [66, 67] and yield losses [18]. Leaf shedding was actually observed on the study plot in July 2010. Yield was not monitored during or immediately after the drought, but mean annual dry mass production at the end of the year 2009 was approximately 5.7 Mg ha^{-1} [28]. This value is relatively low for willow SRCs [18] and indicates that growth conditions are not optimal in the Fuhrberger Feld, likely due to repeated water shortage.

Higher amounts of plant available soil water, either due to a higher soil water storage capacity as found in loamy soils, a greater rooting depth or even direct access to groundwater help to bridge extended dry periods and in terms of yield lead to more robust SRC production systems. However, this increased robustness is at the expense of groundwater recharge, since the soil water storage has to be refilled before drainage can form. Therefore, it can be concluded, that on sites with low plant available soil water capacity and where roots have no access to the water table, a change in land use from fallow to SRC indeed will have a negative impact on groundwater recharge. But on such sites, this impact is, as long as SRCs do not have access to groundwater, moderate: The soil water storage capacity sets a minimum level for groundwater recharge, but also sets the maximum limit to yield.

N released by Nitrate Leaching

Nitrate fluxes from the reference (Ref) and the willow ($W05/_{90}$) plot (Table 8) were calculated from the simulated drainage fluxes. Mean concentrations for all sampling dates were multiplied with the corresponding drainage flux during the sampling interval, nitrate fluxes were cumulated separately for periods A (Feb 2010 to May 2010) and B (Dec 2010 to June 2011). In order to obtain an estimate of the annual nitrate output rate for the year 2010, nitrate seepage concentrations for the months Sep to Nov 2010, where considerable amounts of drainage from Ref took place but concentration measurements were missing, were assumed to be the same as in Dec 2010. For the summer months in 2010, when also no samples could be taken and only minimal drainage occurred, the concentrations measured in May were used to calculate the nitrate flux.

In total, the $W05/_{90}$ plot lost 16.5 kg NO_3-N ha^{-1} a^{-1} in 2010 (Table 8), where 87 % of the losses happened during winter/spring 2010. Nitrate leaching from the reference plot (Ref) was less than a tenth (1.36 kg of $N0_3$-N ha^{-1} a^{-1}) of the amount of the SRC plot. Eighty percent of the annual leaching from Ref occurred during winter and spring 2010.

Calculated nitrate leaching losses for the $W05/_{90}$ plot are slightly higher than reported in previous studies [59, 61]. There, leaching rates were between zero and slightly less than 2 kg ha^{-1} a^{-1}, despite long-term repeated annual nitrogen fertilization of more than 150 kg ha^{-1} a^{-1} [59]. However, our rates were about 10 times lower than rates cited by Aronsson and Bergström [59] for the first year after establishment, when

Table 8 Cumulated drainage water fluxes and nitrate leaching loss (±SD) for the Ref and $W05/_{90}$ study plots during sampling periods A (Feb 2010-May 2010) and B (Dec 2010 –June 2011), as well as the annual sum for 2010

	Drainage flux (L m^{-2})		Nitrate leaching (kg ha^{-1})	
	Ref	$W05/_{90}$	Ref	$W05/_{90}$
Period A	171	143	1.08±0.85	14.3±6.55
Period B	216	180	0.42±0.38	2.03±1.46
Jan–Dec 2010	345	189	1.36±1.08	16.5±7.95

willow was cultivated in lysimeters, highly fertilized and irrigated on sandy soils (140 kg $N0_3$-N ha^{-1} a^{-1}). Similar N loss with seepage of 90 kg $N0_3$-N ha^{-1} a^{-1} were measured by Dimitriou and Aronsson [68] from a lysimeter experiment, where irrigated willows in sandy soil were fertilized with 320 kg of N ha^{-1} in form of sewage sludge. These losses occurred within a time span of 6 months (May to Oct) and were mainly attributed to the high N fertilizer input and not to the chemical composition of the fertilizers. As in our study, almost 90 % of the annual leaching from the $W05/_{90}$ plot happened during the winter/spring 2010 there might be some site specific but until now unknown reason for a relatively high leaching flush. No direct N input by fertilization or any comparable input occurred and possible mineralization artifacts potentially produced by the installation of the suction lysimeters can also be excluded, since they were installed 4 months earlier.

Increased nitrate leaching may also favor N_20 emissions [69]. As part of a Diploma thesis, a series of five N_20 measurement campaigns between June and Oct 2010 was conducted on all plots, except for $W94/_{30}$ [70]. Results indicate that N_2O emissions increased after heavy rainfalls at the end of August and in September 2010 at the $P09/_{90}$ plot. Here, maximum mean values in August reached emissions of 65.0 (±20.5 SD) µg N_2O-N m^{-2} h^{-1}, but values fell back to a baseline of below 20 µg N_2O-N m^{-2} h^{-1} in October 2010 again. However, also the reference plot showed peak values during this period (August 2010: 43.5±18.6 SD µg N_2O-N m^{-2} h^{-1}), whereas the $W05/_{90}$ and $P94/_{30}$ plots never had higher emissions than 20 µg N_2O-N m^{-2} h^{-1} [70]. In agreement with other studies [71, 72], it is concluded that SRC, once established, emit considerably less N_2O, compared to other bioenergy crops or even less than fallow grounds [70]. Estimated annual emission rates for the established SRC plots in the Fuhrberger Feld were below 1 kg ha^{-1} a^{-1} [70].

Conclusions

Former cropland which was abandoned due to protection reasons of nitrate pollution in lowland drinking water catchment areas can well be used for the cultivation of SRC to increase the land use value by the production of woody biomass. Our results showed that in SRCs of willow and poplar clones with different age (2–17 years) and different soil preparation measures (standard and deep plowing), the mean nitrate concentrations in 100 cm soil depth with few exceptions stays below the drinking water threshold value of 10.3 mg NO_3-N L^{-1}. There are two stages, where relatively increased amounts of nitrate might be leached from SRC cultivations, i.e., (1) when SRC are newly installed and intensive or even deep plowing was applied before cultivation (example $P09/_{90}$) and (2) when the sink, respectively the export function for N compounds by tree uptake and harvesting measures is offset (example $W94/_{30}$). Harvesting itself obviously did not initiate a nitrate flush, but nitrate release from an over-aged, never harvested willow stand was significantly increased.

Furthermore, we conclude that groundwater recharge rates, which are also of concern in drinking water sanctuaries, were not excessively reduced by SRC cultivations. Soils with low amounts of plant available soil water storage capacity and high permeability, as often found in lowland drinking water catchment areas, set a minimum level for groundwater recharge by limiting transpiration, as long as roots have no access to the groundwater table. Thus, less precipitation is needed to refill the soil water storage than on soils with higher water storage capacity, and groundwater recharge begins earlier after transpiration ceases in autumn.

Another finding, which needs more investigation though, might provide an opportunity to manipulate the water balance of SRC stands by management. Less interception loss can be expected from stands during the first year after harvested and thus higher groundwater recharge rates might be obtained by choosing a shorter rotation cycle, as the stand then is more often in the resprouting state.

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Changes in Organic Carbon and Trace Elements in the Soil of Willow Short-Rotation Coppice Plantations

Ioannis Dimitriou · Blas Mola-Yudego · Pär Aronsson · Jan Eriksson

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Abstract Short rotation coppice (SRC) is a biomass production system for energy usually grown on former agricultural land with fast-growing tree species. In Sweden, willow SRC has been grown since the late 1980s. SRC on arable soils may induce changes in some soil quality parameters due to differences in crop characteristics and management practices. In this study, pH, organic carbon (C), and trace element concentrations in the soil of 14 long-term (10–20 years) commercial willow SRC fields in Sweden were compared with those in adjacent, conventionally managed arable soils. The results showed that organic C concentrations in the topsoil and subsoil of SRC fields were, on average, significantly higher (9 % in topsoil, 27 % in subsoil) than in the reference fields. When comparisons were made only for the ten sites where the reference field had a crop rotation dominated by cereal crops, the corresponding figures were 10 % and 22 %. The average concentration of cadmium (Cd), which is considered the most hazardous trace element for human health in the food chain, was 12 % lower in the topsoil of SRC fields than in the reference fields. In the corresponding comparison of subsoils, no such difference was found. For chromium (Cr), copper (Cu), nickel (Ni), lead (Pb), and zinc (Zn), there were no significant differences in concentrations between SRC fields and the reference fields in either topsoil or subsoil. Negligible differences in pH in the same comparisons were found.

Keywords Bioenergy · Cadmium · Energy forest · Soil organic carbon · *Salix*

I. Dimitriou (✉) · B. Mola-Yudego · P. Aronsson
Department of Crop Production Ecology,
Swedish University of Agricultural Sciences (SLU),
P.O. Box 7016, 750 07 Uppsala, Sweden
e-mail: ioannis.dimitriou@slu.se

B. Mola-Yudego
School of Forest Sciences, University of Eastern Finland,
P.O. Box 111, 80101 Joensuu, Finland

J. Eriksson
Department of Soil and Environment,
Swedish University of Agricultural Sciences (SLU),
P.O. Box 7014, 750 07 Uppsala, Sweden

Introduction

Short rotation coppice (SRC) is a system for biomass energy production on agricultural soil using fast-growing tree species. The species employed have the ability to resprout from the stumps or roots after harvests occurring at short intervals (i.e., 2–6 years). The most commonly used species in SRC plantations are shrubs or trees such as willow (*Salix* spp.) and poplar (*Populus* spp.). Many of the management practices for SRC, e.g., weed control, planting, fertilization, and harvest, resemble those of traditional annual arable crops more than those of forestry. However, soil tillage is only carried out at stand establishment, and the soil is then usually left more or less undisturbed until the stand is broken up after 10–25 years. Sweden is a pioneer in SRC-related research and development, as SRC cropping systems were developed back in the 1970s with the intention of replacing fossil fuels and nuclear power with renewable energy sources [1]. Different species and hybrids of willow have been used as planting material in Swedish SRC plantations, and the first commercial fields were established in the 1980s, covering around 14,300 ha [2].

Incentives to promote cultivation of SRC on productive arable land have been introduced in several countries in Europe to increase bioenergy production, but widespread expansion has not occurred to date. However, different stakeholders have estimated that millions of hectares of arable land cultivated with traditional crops will be converted to SRC [3] as the demand for alternatives to fossil fuels increases. A shift from traditional arable crops to SRC can lead to subsequent changes in soil quality in these areas. SRC is a perennial crop and thus replaces arable crop rotations for a number of years (10–25 years depending on market conditions and/or national regulations). Harvest of SRC is carried out in winter or early spring, plants are deeper rooted than annual crops, no annual soil tillage is practised and considerably less agrochemicals are used.

Current SRC management practices in Sweden will probably affect soil quality in comparison with other arable crops. Commercial SRC plantations in Sweden are sparsely fertilized with mineral fertilizer, despite current recommendations because growers do not consider this to be sufficiently profitable [4]. However, in many commercial Swedish willow SRC plantations, sewage sludge (or mixtures with wood ash, when available, for a more balanced fertilizer) is applied as fertilizer. Farmers even get paid to receive sludge on their fields [5].

Several studies have looked into the possible long-term effects on soil quality of establishment and management of willow SRC, focusing mostly on changes in soil organic carbon (C) and hazardous compounds, mainly trace elements [6]. The potential of SRC for storing C in agricultural land was recognised at an early stage. This is attributed to specific features of the crop, i.e., repeated leaf litterfall and rapid fine root turnover, accumulation and decomposition of larger roots and stumps, and no tillage [7, 8]. However, empirical studies estimating changes in C storage in the soil of willow and poplar SRC have provided conflicting results, with increases in C stocks in the soil reported in some cases [9–11] and decreases in others [12–14]. It has been concluded that the site-specific variability in the effects of SRC on the soil C pool is high, that previous studies may not have covered a sufficiently long period to detect significant changes in soil C stocks, and that the fundamental mechanisms responsible for soil organic C accumulation in SRC are not well understood [13, 15].

Another soil quality parameter that has been broadly referred to in connection with SRC cultivation from its early development stages is trace element concentration, mainly cadmium (Cd) [16, 17]. Cd entering the food chain from agricultural soils is generally considered the most hazardous trace element to human health. According to Åkesson et al. [18, 19], even moderate everyday exposure to Cd poses a risk of renal malfunction and osteoporosis. Recently, the European Food Safety Authority (EFSA) reduced the recommended maximum weekly intake of Cd from 7 to 2.5 $\mu g\ kg^{-1}$ body weight [20]. In Sweden, there has been a rather restrictive attitude to Cd in agriculture. This has led, for example, to general use of phosphorus (P) fertilizers (NPK and NP) containing less than 5 mg Cd/kg P. In the 'Swedish Seal of Quality,' a certification system for product quality and environmental impact in conventional production, a low Cd level in grain delivered is one criterion for certification. The current limits are 80 $\mu g\ kg^{-1}$ fresh weight for winter wheat and oats and 100 $\mu g\ kg^{-1}$ fresh weight for spring wheat [21]. The Cd issue is also important in the discussion on whether it is appropriate to apply sewage sludge on land for food production. In fact, there has generally been a great reluctance among Swedish farmers in recent decades to accept sludge on land used for food crops. Today, some food companies also do not accept crops from sludge-amended land as a condition of their food quality assurance schemes. However, it has been fairly common for sewage sludge to be spread in willow SRC, since that is not a food crop. The argument is that this practice is acceptable even when fields currently cropped with willow SRC are used for food crops in the future, since more Cd is removed with the willow than is added with the sludge [22, 23].

The ability of willow plants to take up rather high amounts of Cd in their shoots, which can then be removed from the field at harvest, has been proposed as a solution to combine biomass production and remediation of moderately contaminated soils [6, 22, 23]. According to Andersson [24], the average Cd concentration in Swedish top soils has increased by more than 30 % in the past century. Any attempt to remediate soils in this way presupposes that the ash, where most of the trace metals end up when willow biomass is incinerated, is not recirculated to arable land. An alternative method to reduce recirculation of Cd with ash is to use only the bottom ash for soil remediation. Such a system had been developed at the thermal power plant in Enköping, west of Stockholm in Sweden. The Enköping plant only uses biomass as fuel and the bottom ash produced contains approximately 85 % of all ash produced and approximately 10 % of all Cd. The Cd-rich fly ash is deposited in landfill [25].

Uptake of other trace elements such as copper (Cu), lead (Pb), zinc (Zn), chromium (Cr), nickel (Ni), and arsenic (As) has been studied too, but most of these studies have been performed as experiments in pots or under hydroponic conditions [26–30]. When sewage sludge is applied to willow SRC, trace elements are also supplied, but several studies of trace element balances in willow stands suggest that plant uptake is able to compensate for this. A reduction in Cd even after sludge amendment is highly probable, but it is questionable whether this is the case for the other trace elements investigated [22, 23, 31]. As for C, uptake of trace elements depends on many factors and may be difficult to estimate. If current concentrations and amounts in harvested biomass of a newly established SRC are extrapolated over a period of up to 20 years, for example, the output may be overestimated. The reason is that an efficient reduction in trace element concentrations in soil may result in decreasing uptake over time when the most soluble fractions are removed. To enable reliable conclusions to be drawn about the possibility of reducing trace element concentrations in arable soils by SRC cropping, more systematic long-term studies in the field are required [6, 32, 33].

The long-term impact on soil quality parameters, such as pH and C, nitrogen (N), and trace element concentrations of growing commercial willow SRC plantations on regular arable land in Sweden, has not been investigated in any great detail. Given the site specificity of such investigations, comparisons between soils under willow stands and those in adjacent fields with annual crops could provide information concerning changes in soil C or total concentrations of trace elements when willow SRC is introduced. The majority of previous studies have examined this issue in rather recently established plantations that are in their first or second rotation (e.g. [8–11, 17, 22, 23]). However, the status of plantations that have been in place for a number of rotations would be of greater interest. The aim of this study was, therefore, to evaluate the potential impact of SRC plantations on agricultural soil on the above-mentioned soil quality parameters in a number of long-term SRC fields in Sweden. To achieve this aim, a number of commercial Swedish SRC plantations in different stages of growth (ranging from 10 to 20 years of age) were selected and compared with adjacent, conventionally managed arable fields on soils that could be expected to initially have been similar in properties. The main hypotheses were that:

1. C concentrations in topsoil and subsoil are similar in SRC fields and in adjacent reference fields.
2. Cd concentrations in topsoil and subsoil of SRC fields are lower than in reference fields, whether amended with sewage sludge or not.

Materials and Methods

Sites

The SRC fields investigated were located in areas of Sweden with a high frequency of willow plantations (Fig. 1). Most of the sites were in east-central Sweden, one field was in southwest Sweden, and one in southern Sweden (Fig. 1). The criteria used in selection of SRC fields for the study were that the SRC stand should have been established and cultivated for a long period, involving at least two rotations, and that the fields should be located in flat areas where texture and other soil properties could be expected not to vary too widely between the SRC stand and the reference field. We did not excavate soil profile pits for closer examination of the soils, but the rather clayey soil type selected would be classified as a Eutrocyept (or Eutrudept at the most southerly site) according to Soil Taxonomy [34] and Eutric Cambisol according to WRB [35]. The selected fields were also representative of the common system of management used for commercial Swedish plantations (Table 1).

This project was initiated in 2009, but unfortunately there were no detailed records on the management of the selected sites during the 10–20 years that had elapsed since the willow stands were established. Information obtained from farmers on the number of times mineral fertilizer was applied indicated that fertilization was only carried out occasionally at the start of the growing season after winter harvest (Table 1). However, it is uncertain whether a single N application of 70–80 kg ha^{-1} $year^{-1}$ as recommended by Ledin et al. [36] was made or whether a higher dose intended to last up to next harvest was applied. The low biomass production in many of the stands indicates that the N doses were far from optimal. For application of sludge to arable fields, there are legal limits on the amount of nutrients and trace metals that are allowed to be spread per hectare [37]. Most commonly, it is the P content that is the limiting factor, but in some cases, it is the concentration of Cu or Cd or other trace elements that sets the limit. Maximum permissible annual input of P to the type of soils studied here is 22 kg P ha^{-1}. One-time application of doses lasting for more than 1 year is allowed. In practice, this means that 5- to 7-year doses of approximately 4–6 t sludge are applied. The number of sludge applications, presumably of this magnitude, made in the SRC fields studied here is shown in Table 1.

The reference fields with conventional annual crops have presumably been fertilized according to recommendations from the Swedish Board of Agriculture [38]. For winter wheat, which is commonly grown in the areas where the study sites were situated, this means 120–140 kg N ha^{-1} for average yields, 0–15 kg P ha^{-1} depending on soil P status and 0–10 kg potassium (K). The illitic, rather heavy Swedish clays usually need no addition of K to cereal crops. The reference fields with grassland had not been fertilized or harvested in the previous 10 years.

Sampling and Analyses

Samples of topsoil (0–20 cm) and subsoil (40–60 cm) were taken from all SRC fields and their corresponding reference fields in autumn 2009. In the SRC fields, three composite samples were taken with a soil auger parallel to the reference field, approximately 5–10 m from the border of the SRC field and with a distance of approximately 15 m between sampling points. Each topsoil sample analysed consisted of six pooled auger subsamples taken across three different willow double rows (two from between plants, two from between rows, and two from between double rows). The topsoil samples from reference fields were taken in a similar way, but in that case, the six subsamples were taken within a

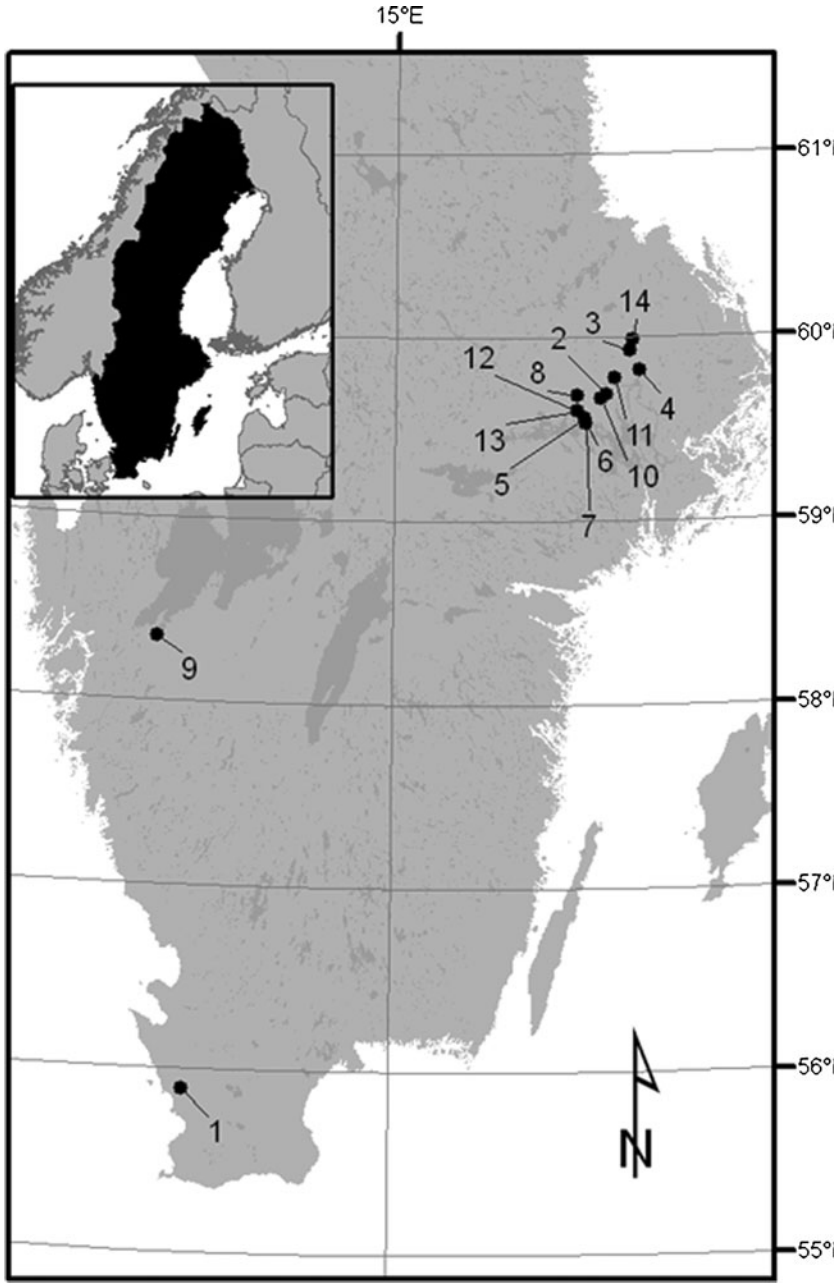

Fig. 1 Geographical location of the Swedish willow short rotation coppice (SRC) fields studied. *Field numbers* are the same as in Table 1

circle with a radius of 2 m. The subsoil samples were taken in the same way and in the same plots as the topsoil samples in both the SRC and the reference fields, but in that case, only four soil subsoil cores were taken and pooled together into one bulk sample.

The samples were dried at 30–40°C until constant weight. Soil pH was measured in a suspension of 5 ml soil +25 ml deionized water. The suspension was shaken for 15 min, left standing overnight, and then, immediately before measurement on the next day, the suspension was again shaken for 1 min. Total C and total N were measured using an elemental analyzer (LECO CH-2000) in which 1 g of sample was heated to 1,250°C for 5 min. Soils with high pH may contain carbonates, and therefore, C in the form of carbonates was measured in soils with pH higher than 6.6. For this, 1 g of sample was first heated to 550°C for 5 h in a laboratory oven to ignite all organic C. Then, the sample was heated to 1,250°C in the elemental analyzer as described above, and carbonate C content was determined. In this procedure, control samples with known content of organic C were included to check that no carbonate C was lost during ignition in the first step. Only some soils with pH higher than 7 were found to contain carbonate C and for those, the organic C content was calculated as total C minus carbonate C. For other soils, organic C was considered to equal total C. Pseudototals of Cd, Cr, Cu, Ni, Pb, and Zn

Table 1 Description of the study sites. *Sludge/ash/mineral fertilizer* number of times applied since short rotation coppice (SRC) establishment indicated in brackets; *latest harvest* years refer to the latest harvest, with total number of harvesting occasions since establishment given in brackets; *Biomass* living above-ground woody biomass in ton of dry matter per hectare per year (note that shoots in the different fields were of different ages)

	Site	Texture, topsoil	Willow SRC fields Year planted	Variety	Sludge/ash applied	Mineral fertilizer applied	Latest harvest	Biomass (2009)	Crops before SRC	Reference fields Crops
1	Billeberga II	Loam	1994	Torhild	Yes/no (3)	No	Annually	2[a]	Cereals	Cereals/rape seed
2	Djurby Gård	Silty clay	1990	78021	Yes/no (3)	No	2007 (5)	5.3	Cereals	Cereals
3	Forkarby	Silty clay	1991	78021	No/no	Yes (2)	2008 (5)	11	Cereals	Cereals
4	French trial	Clay loam	1994	Mixture	No/no	Yes (8)	2007 (5)	9.3	Cereals	Grass
5	Hacksta	Clay loam	1994	Jorr	Yes/yes (4)	Yes (1)	2008 (3)	4.2	Cereals	Pea/cereals
6	Hjulsta I	Clay	1995	Jorr	Yes/yes (2)	No	2008 (3)	4.5	Oil crops/cereals	Cereals
7	Hjulsta II	Clay	1995	Jorr	No/no	No	2008 (3)	9.6	Oil crops/cereals	Cereals
8	Lundby Gård II	Clay	1995	78021	No/no	No	2005 (2)	2.5	Cereals	Cereals
9	Puckgården	Silty clay	1992	78112	No/no	Yes (4)	2008 (4)	10[a]	Cereals	Cereals
10	Skolsta	Silty clay	1993	Orm	Yes/yes (1)	Yes (2)	2004 (2)	4	Cereals	Cereals
11	Säva	Silty clay	1993	Rapp	Yes/no (2)	No	2007 (3)	7.4	Cereals	Grass
12	Teda I	Silty clay loam	2000	Tora	Yes/yes (2)	Yes (2)	2009 (2)	8	Cereals	Grass
13	Teda II	Clay	1993	78112	Yes/yes (2)	Yes (2)	2007 (3)	1.7	Cereals/fallow	Grass
14	Åsby	Silty clay	1996	Tora	Yes/no (1)	Yes (2)	2008 (3)	4.2	Grass	Grass

[a] Estimates obtained from farmers, not actual measurements

were determined after extraction with 7 M nitric acid (2.5 g soil, 20 ml acid) at 120°C for 2 h on a Tecator heating block. Trace element concentrations were measured with ICP-MS (Perkin Elmer Elan 6100 DRC).

Soil texture was determined with a combination of sieving for larger particle fractions and sedimentation analysis with the pipette method. Soil particles were dispersed by means of treatment with hydrogen peroxide and addition of sodium pyrophosphate decahydrate, followed by overnight agitation in an end-over-end shaker.

Measurements of above-ground living woody biomass were conducted in each SRC field where the soil samples were taken in spring before the growing season in 2010 and 2011. Above-ground biomass was measured annually by nondestructive methods according to [39].

Statistical Analyses

For every soil parameter, the values for each plantation were compared with those for the reference fields, based on relative differences. In a preliminary analysis, the parameters were tested for the assumption of normality, using a one-sample Kolmogorov–Smirnov test. In the case of Pb, the resulting p value was 0.049. In the other cases, the resulting p value was above 0.1. Therefore, it was assumed that the variables analyzed followed a normal distribution. Afterwards, the values were tested using a t test against the hypotheses of no differences between the willow SRC plantation and the reference field ($p<0.05$). The comparisons were expressed as relative differences.

It was possible to divide the reference fields into two groups, one where the field was cropped with cereals and one where it was cropped with grassland. Another subdivision was based on whether the reference fields had received mineral fertilization or sewage sludge. In comparisons between each type of reference field and its corresponding SRC field, the reference and SRC fields were considered treatments and any differences between them were tested using ANOVA analysis.

Results

The data on soil properties revealed the variability between sites, and the concentration ranges obtained were typical for arable soils in the parts of Sweden where most of the sites were situated (Fig. 2 and Table 2). Some of the soils had carbonates in the subsoil, with the highest values measured in field 7 (5.1 % $CaCO_3$ equivalents in SRC soil and 3.2 %

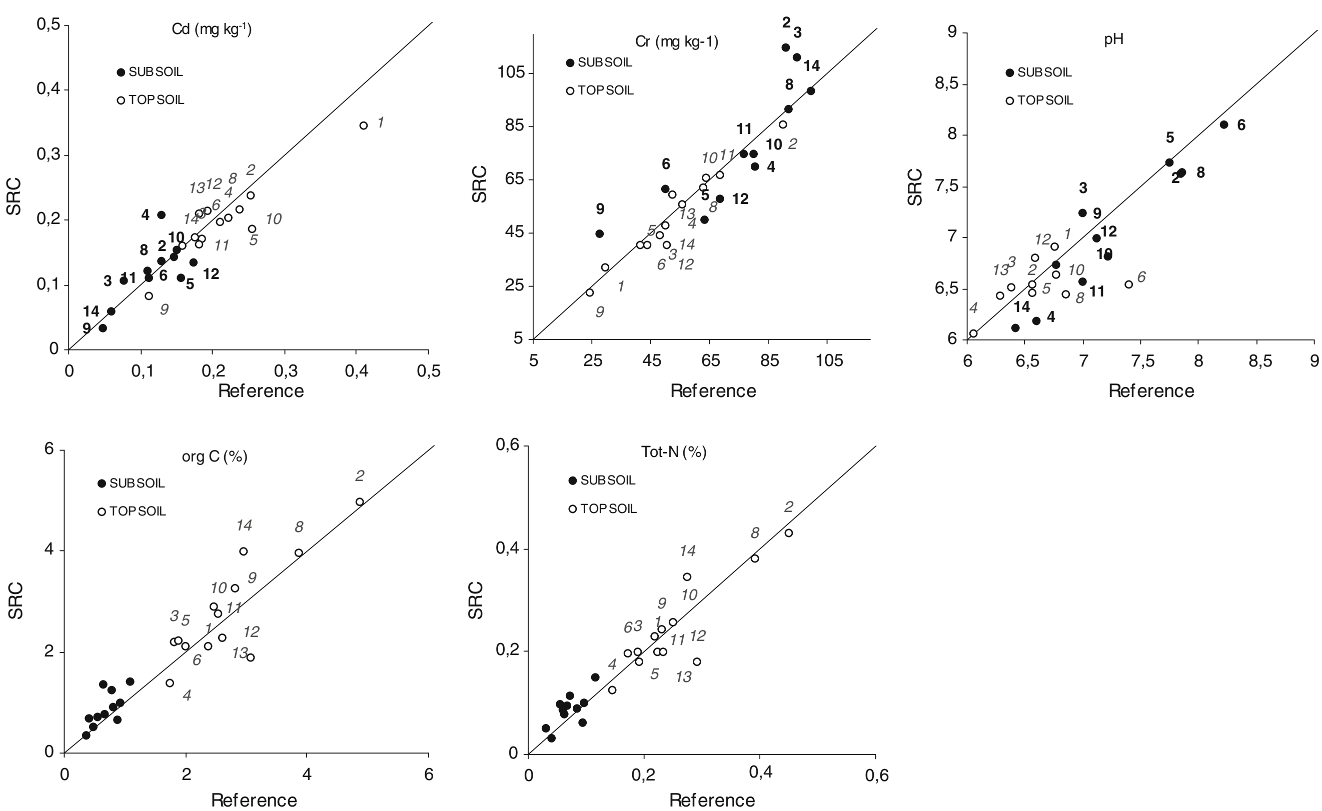

Fig. 2 Cadmium (Cd), chromium (Cr), organic carbon (C), and total nitrogen (N) concentrations, and pH in topsoil (0–20 cm) and subsoil (40–60 cm) in the willow short rotation coppice (SRC) and in the reference sites. *Field numbers* are the same as in Table 1. The nearer the values are to the 1:1 diagonal, the greater the similarity in concentrations in topsoils and subsoils

Table 2 Soil properties in the short rotation coppice (SRC) fields and in the reference fields studied. Means and standard error. *N* number of observations. In some cases two SRC fields shared the same reference field, so there are fewer observations for reference fields than SRC fields

	SRC		Reference	
	Mean	Standard error	Mean	Standard error
Topsoil (0–20 cm)	(*N*=42)		(*N*=36)	
Organic C (%)	2.7	0.2	2.6	0.2
Total N (%)	0.24	0.01	0.24	0.02
C/N	11	0.1	11	0.1
pH	6.4	0.1	6.5	0.1
Cr (mg kg^{-1})	51	2	53	3
Ni (mg kg^{-1})	23	1	23	1
Cu (mg kg^{-1})	27	1	25	1
Zn (mg kg^{-1})	92	4	91	3
Cd (mg kg^{-1})	0.19	0.01	0.22	0.01
Pb (mg kg^{-1})	20	0.5	21	0.7
Subsoil (40–60 cm)	(*N*=12)	SE	(*N*=11)	SE
Organic C (%)	0.78	0.10	0.64	0.07
Total N (%)	0.08	0.01	0.07	0.01
C/N	12	2	11	2
pH	7.2	0.2	7.3	0.2
Cr (mg kg^{-1})	76	7	75	6.5
Ni (mg kg^{-1})	30	2	31	2.4
Cu (mg kg^{-1})	29	2	28	2.7
Zn (mg kg^{-1})	96	7	96	7.4
Cd (mg kg^{-1})	0.12	0.01	0.12	0.01
Pb (mg kg^{-1})	19	0.6	19	0.93

in reference soil) and field 6 (3.3 % $CaCO_3$ equivalents in both). Field 2 had 1.6 % $CaCO_3$ equivalents in both SRC and reference soils, while the values in fields 5 and 8 varied between 0.1 and 0.4 %. Some of these fields also had traces of carbonate C in the topsoil.

Based on single relative differences calculated for each site, the SRC topsoils had on average 9 % higher organic C concentration and 7 % lower concentrations of Cd than the reference topsoils (Table 3, Fig. 3). When the same comparison was made between SRC fields and corresponding reference fields cropped only with cereals, the relative differences were larger, 10 % higher organic C concentration and 12 % lower Cd concentrations in SRC fields (Fig. 3). The N concentration covaried with organic matter concentration, since most soil N is a component of organic matter. For some reason, unlike the organic C concentration, the average N concentration in the topsoil was not higher in SRC fields than in reference fields, probably as an effect of differences in fertilization with inorganic N.

Table 3 Differences in soil quality parameters between willow short rotation coppice (SRC) fields and reference fields expressed as absolute values for pH and as relative values (in percent) for the rest of the elements. Positive values mean the studied parameter is higher in the SRC plantations than in the reference fields. Negative values mean the opposite. The *p* values are resulting from a *t* test, testing whether the differences differ from 0 (for pH) or from 1 (for the rest of the elements)

Variable	Topsoil (0–20 cm) Mean difference	*p* value	Subsoil (40–60 cm) Mean difference	*p* value
pH	−0.01 U	0.145	−0.2 U	0.010
org C	9.5 %	0.001	27.2 %	0.039
Tot-N	2.8 %	0.170	21.6 %	0.046
Cr	−1.3 %	0.569	6.4 %	0.350
Ni	0.7 %	0.764	1.1 %	0.823
Cu	2.7 %	0.432	4.9 %	0.257
Zn	−0.07 %	0.977	−0.4 %	0.866
Cd	−6.8 %	0.006	1.9 %	0.805
Pb	−2.1 %	0.283	3.8 %	0.209

The concentrations of organic C and total N and Cd were higher in the topsoil than in the subsoil in SRC fields, while the reverse was observed for Cr (Table 2). For other trace elements, there was no difference. The subsoil in the SRC fields had higher concentrations of organic C and total N (27 % and 22 %, respectively) than that in the reference fields (Fig. 3). Regarding trace elements in subsoil, no significant relative differences were found between SRC and reference fields.

When SRC fields were grouped according to fertilizer regime, mineral fertilizer, or sewage sludge, the only significant effect was lower Cu concentration in SRC fields than in the reference fields. Furthermore, there was no relative difference in soil properties when SRC fields were grouped on the basis of rotation length or age of the stands and tested against their corresponding reference fields.

Discussion

The organic C concentration in the topsoil of SRC fields was on average 9 % higher than in adjacent fields with annual crops. When only fields used for cereal production (grassland soils omitted) were considered in the comparison, the organic C concentration was 10.5 % higher in SRC fields. There was no discernible effect of application of sewage sludge, but as mentioned earlier, we had no reliable data on the amounts of sludge applied. This confirms findings by Kahle et al. [11], Jug et al. [9], and Schmitt et al. [40], all of whom reported higher soil organic C concentrations in the topsoil of willow SRC than in that of adjacent fields with annual crops. The SRC stands in those investigations were 12,

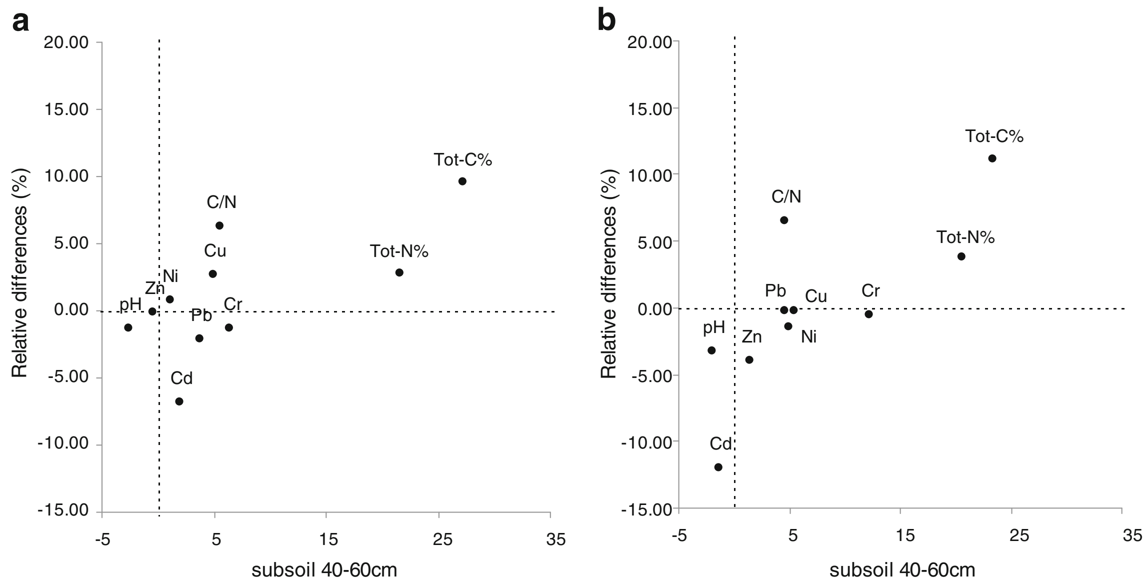

Fig. 3 Relative differences in soil properties between willow short rotation coppice (SRC) plantations and the references. **a** Including all fields. **b** Including only those with cereals as reference values. The values are the averages for the three general samples each from SRC plantations and the references, respectively. Differences between topsoils (0–20 cm) on the *y*-axis and subsoils (40–60 cm) on the *x*-axis. *Positive values* means the studied parameter is higher in the willow SRC plantations than in the references. *Negative values* mean the opposite

4, and 4 years old, respectively, i.e., younger than the majority of our stands. These results indicate that SRC cropping leads to a slow increase in organic C content in the topsoil due perhaps to absence of tillage and higher leaf litterfall. Furthermore, it has been reported that most fine roots in SRC stands are concentrated in the topsoil and are renewed continuously, resulting in very high annual turnover (approximately 7 tha^{-1} [41]). This may contribute to elevated soil C in the topsoil. In the 'old' SRC fields we studied, decomposition of coarse roots, which are mainly found in upper 40 cm [42], and dead stumps had probably also contributed to the increased soil C concentrations in the topsoil. The difference in C content between soil in willow plantations and adjacent grassland tended to be small, presumably because grassland also tends to promote accumulation of organic matter.

The higher C concentrations in the subsoil of SRC stands observed here contradict the results of a similar study in a 4-year-old willow SRC plantation [40], which found no significant differences in subsoil C concentrations between SRC and cereals. On the other hand, our findings agree with those of Kahle et al.'s [43], who reported higher organic C in the subsoil of a 15-year-old willow SRC plantation than in the reference fields. A small percentage (approximately 5–25 %) of total fine and coarse roots can be found in deeper soil layers [42, 44]. The increased C content in both topsoil and subsoil suggests that SRC cultivation may potentially cause C sequestration in the whole soil profile in arable soils. The increase in organic C concentration is smaller in the subsoil, but considering organic C stocks and taking into account the greater volume of the subsoil, overall organic C accumulation in the subsoil was almost as high as in the topsoil. It was not possible to correlate any differences in soil C concentrations between SRC and reference fields to differences in soil texture or fertilization (mineral fertilizer, sewage sludge, sewage sludge, and wood ash). If such correlations exist, we may have failed to detect them because the number of study sites was small, we do not know whether the amounts of sludge applied were enough to have an effect, and our SRC fields and their corresponding reference fields may already have differed in organic C content at the time the willow was planted. Difficulties in finding general patterns in soil organic C changes in SRC plantations established in different locations compared with reference fields have previously been reported by others [11, 13] and attributed to the variability of conditions at different locations (such as soil management and climate conditions).

Stronger acidification resulting in lower pH in willow SRC fields than arable fields with annual crops has been reported previously [9, 11]. However, we saw no such trend in our study, as differences in pH between SRC and reference fields were small. Any difference in acidification rate between SRC and reference fields in our case would mainly be due to removal of biomass in harvest. However, according to the data in Table 1, the offtake of biomass in SRC on an annual basis was lower than or similar to the possible offtake in cereal crops. Furthermore, the soils at the study sites were generally rather clayey and some even had calcareous parent material, as indicated by the presence of carbonate C in the subsoil.

Cd was the only trace element for which concentration in the topsoil differed significantly between SRC fields and reference fields, with concentrations being, on average, approximately 7 % lower in SRC fields. The difference was even larger, approximately 12 % lower in SRC fields, when only SRC plantations and reference fields cropped with cereals were compared. This indicates that approximately 15 years of willow SRC cultivation can significantly reduce Cd concentrations in topsoil, supporting estimates of potential output of Cd from willow SRC based on stem Cd concentrations reported by Klang-Westin and Eriksson [22], Dickinson and Pulford [32], Dimitriou [33], Berndes et al. [45], and Witters et al. [46]. For example, Klang-Westin and Eriksson [22], Dimitriou [33], and Berndes et al. [45] calculated that after 20–25 years of SRC cultivation, a reduction of approximately 25–30 % in Cd concentration in the topsoil could be expected if biomass production was 10 t DM ha^{-1} $year^{-1}$. The corresponding effect in our study was smaller, which could be due to lower yields or use of different clones [26] as the basis for calculations in the above studies. Furthermore, we do not know whether our SRC and reference fields were actually equal in topsoil Cd concentrations when the SRC plantations were established. Our finding that willow SRC cultivation has little effect on the concentrations of trace elements other than Cd compared with conventional annual crops is also in agreement with findings in some of the studies cited above.

The relative differences in the effect of SRC on soil Cd concentrations between different locations were not correlated with sludge and/or wood ash amendments. This agrees with Klang-Westin and Eriksson [22] and Dimitriou et al. [23], who found that the potential output of Cd with willow may be up to eight to ten times larger than the legally accepted input with sludge and other amendments. The maximum permissible input of Cd with sewage sludge in the past 10 years has been rather low, 0.75 gha^{-1}, on an annual basis [37]. Furthermore, other parameters, e.g., soil texture, number of harvest occasions, and application of mineral fertilizer, could not explain differences between SRC and reference fields in effect on Cd concentration, presumably for the same reasons as discussed for organic C above.

Despite the uncertainty about any initial difference in Cd concentrations between the SRC and reference fields, we believe that our results confirm that commercial willow SRC plantation has the ability to reduce soil Cd concentration. Thus, biofuel production with willow will have a remediating effect on moderately polluted soils in the long-term, at least if fly ash is not recirculated to the soil. While temporary accumulation in below-ground plant parts [47] could have contributed to our results, the main reason for the reduced Cd in the topsoil was presumably the extraction from the field in the harvested biomass. This is indicated by the smaller difference in Cd concentration between willow fields and reference fields with grass than reference fields with cereal crops. Offtake of Cd is higher for grass than for cereals due to higher output of biomass with grass (whole-shoot harvesting more than one occasion per year).

Conclusions

The main conclusions of this study, which compared a number of soil quality parameters in 10- to 20-year-old commercial SRC fields and adjacent fields with annual crops were that:

- Organic C concentrations in the topsoil and subsoil of SRC fields were significantly higher than in the respective reference fields (starting hypothesis rejected)
- Cd concentrations in the topsoil of SRC fields were significantly lower than in the topsoil of respective reference fields (starting hypothesis confirmed)
- Cd concentrations in the topsoil of SRC fields treated with sludge and/or wood ash were still lower than in the topsoil of reference fields (starting hypothesis confirmed), and the differences were just as pronounced.

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Short Rotation Coppice (SRC) Plantations Provide Additional Habitats for Vascular Plant Species in Agricultural Mosaic Landscapes

Sarah Baum · Andreas Bolte · Martin Weih

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Abstract Increasing loss of biodiversity in agricultural landscapes is often debated in the bioenergy context, especially with respect to non-traditional crops that can be grown for energy production in the future. As promising renewable energy source and additional landscape element, the potential role of short rotation coppice (SRC) plantations to biodiversity is of great interest. We studied plant species richness in eight landscapes (225 km^2) containing willow and poplar SRC plantations (1,600 m^2) in Sweden and Germany, and the related SRC α-diversity to species richness in the landscapes (γ-diversity). Using matrix variables, spatial analyses of SRC plantations and landscapes were performed to explain the contribution of SRC α-diversity to γ-diversity. In accordance with the mosaic concept, multiple regression analyses revealed number of habitat types as a significant predictor for species richness: the higher the habitat type number, the higher the γ-diversity and the lower the proportion of SRC plantation α-diversity to γ-diversity. SRC plantation α-diversity was 6.9 % (±1.7 % SD) of species richness on the landscape scale. The contribution of SRC plantations increased with decreasing γ-diversity. SRC plantations were dominated more by species adapted to frequent disturbances and anthropo-zoogenic impacts than surrounding landscapes. We conclude that by providing habitats for plants with different requirements, SRC α-diversity has a significant share on γ-diversity in rural areas and can promote diversity in landscapes with low habitat heterogeneity and low species pools. However, plant diversity enrichment is mainly due to additional species typically present in disturbed and anthropogenic environments.

Keywords Agriculture · Biodiversity · Bioenergy · Poplar (*Populus*) · Structural heterogeneity · Willow (*Salix*)

S. Baum (✉) · A. Bolte
Institute for Forest Ecology and Forest Inventory,
Johann Heinrich von Thünen-Institute (vTI),
Alfred-Möller-Straße 1,
16225 Eberswalde, Germany
e-mail: sarah.baum@vti.bund.de

A. Bolte
e-mail: andreas.bolte@vti.bund.de

S. Baum · A. Bolte
Department of Silviculture and Forest Ecology of Temperate Zones, Georg-August-University Göttingen,
Büsgenweg 1,
37077 Göttingen, Germany

M. Weih
Department of Crop Production Ecology,
Swedish University of Agricultural Sciences (SLU),
P.O. Box 7043, Ulls väg 16,
750 07 Uppsala, Sweden
e-mail: martin.weih@slu.se

Introduction

Against the background of global biodiversity loss largely caused by intensive agriculture [1–5], the diversity of entire agricultural landscapes, the γ-diversity, is of great research interest. The γ-diversity addresses the species diversity of a landscape with more than one kind of natural community, and it includes the diversity within (α-diversity) and among communities (β-diversity, terminology of Whittaker [6]). Unlike species richness, species diversity takes the proportional abundances of species into account [7]. Many scientific papers address the question of the importance of structural heterogeneity in agricultural landscapes and agree that landscape heterogeneity is beneficial for biodiversity [i.e. 8–12]. According to Forman [13], a matrix of large patches of plant communities supplemented with small patches scattered throughout the landscape characterizes an optimum landscape as small patches provide different benefits for biodiversity compared to large patches.

The cultivation of bioenergy crops as renewable energy source is debated widely [cf. 14–17]. To reach the EU target of producing 20 % of the primary energy consumption from renewable energies in the year 2020, vast areas of land will be necessary for energy crop cultivation [18–20] for biomass production to be a promising option [i.e. 14, 21]. The large areas needed and economic cost of transporting raw biomass material to end-use locations raise concerns about large-scale biomass crop monocultures [18]. Short rotation coppice (SRC) plantations are perennial lignocellulosic energy crops with high biomass yields; they are expected to play a major role (together with perennial grasses like miscanthus, reed canary grass and giant reed) in increasing the amount of renewable energy from biomass in Europe [22, 23]. The potential contribution of SRC plantations to biodiversity as an additional landscape element in agricultural areas is described in various studies [e.g. 24–33], which reported predominantly positive effects.

The aim of our study is to analyse the suitability of SRC characteristics and landscape matrix characteristics for predicting the contribution of α-diversity of SRC plantations to vascular plant γ-diversity in fragmented agricultural landscapes. As an alternative to the equilibrium theory of island biogeography by MacArthur and Wilson [34] and Duelli [35, 36] developed the mosaic concept for agricultural landscapes claiming habitat variability (number of biotope types per unit area), habitat heterogeneity (number of habitat patches and ecotone length per unit area) and the proportional area of natural (untouched), semi-natural (perennial vegetation or cultures with low input) and intensely cultivated areas (mainly annual crops and monoculture plantations) as the most suitable factors for predicting biodiversity of an agricultural mosaic landscape. Evidence for this theory was found by Simmering et al. [11]: while at the patch scale, habitat type, area and elongated shape were the main determinants of plant species richness, non-linear habitat richness, the gradient from anthropogenic to semi-natural vegetation and the proportions of natural vegetation and rare habitats were predictors for species richness at the multi-patch (1 ha each) scale, in a highly fragmented agricultural landscape in central Germany. A positive relationship between vascular plant species richness, number of habitat types and habitat patches per area was also found by Waldhardt et al. [12].

The plant species richness of willow and poplar SRC plantations smaller than 10 ha and grown for biomass energy was related to γ-diversity of the corresponding five Swedish and three German landscapes. In reference to the mosaic concept [35, 36], we explore the hypotheses that the share of SRC plantation α-diversity on γ-diversity depends on (1) landscape structure and (2) γ-diversity itself. In contrast to landscapes with homogenous structures, we expect a higher γ-diversity but lower SRC plantation α-diversity in areas with heterogeneous structures characterized by high numbers of habitats and habitat patches with long edges. Further, we expect a higher γ-diversity in areas with higher proportions of semi-natural vegetation and rare habitats, and a higher SRC plantation α-diversity share in species-poorer landscapes than in species-richer ones.

Material and Methods

Study Areas and Sites

Our survey on plant species diversity was conducted on eight landscapes of 15×15 km, corresponding to 225 km^2 surface area. Five areas were located in Central Sweden in the Uppland province and three in Northern Germany in the states of Brandenburg (one study area) and Lower Saxony (two study areas). We selected study areas (landscapes) in which SRC plantations were a representative element. Within each landscape, we chose one or several SRC plantations of 1 to 10 ha, and we delimited the landscapes so that the SRC plantations were situated centrally. We chose SRC plantations for which we had sufficient information regarding plant material and management history. The SRC plantations contained mainly willow clones but also poplars of various ages and rotation regimes. Former land uses also varied (for further descriptions of SRC study sites see Table 1). Due to overlaps with another research project we used four landscapes in which two SRC plantations each were considered (SRC study sites Franska/Kurth, Hjulsta, Lundby), and one landscape in which three SRC plantations were regarded (study sites Bohndorf I, II and III). The SRC plantations located in the same landscape cannot be considered independently in statistical analyses. Thus, we used mean species numbers, shoot ages and plantation ages for SRC plantations located in the same landscape.

The Swedish sites were exposed to lower temperatures and received less precipitation than the German sites: mean annual temperature was about 5.5 °C for the Swedish study sites and 8.5 °C for the German sites. During the growing season (May–September) mean monthly temperature was 13.5 °C for the Swedish and 15 °C for the German sites. Annual precipitation was about 530 mm (monthly mean during the growing season: 55 mm) for the Swedish sites and about 640 mm (monthly mean during the growing season, 60 mm) for the German sites (data bases: long-term recordings from 1961 to 1990 [37, 38]).

The Swedish study sites were characterized by cohesive soils with high clay content. The bedrock is predominantly granite and gneiss. Sand deposits, which were covered with sandy soils, were the prevailing parent material at the German

Table 1 Overview of the SRC study sites

Landscape	SCR site	Country	Geographical location N	E	Size (ha)	Estab.	Rot. No.	Last harvest	Sampled crops	Previous land use
Åsby (AS)	Åsby	S	59°59′07″	17°34′57″	8.2	1996	4	2008	Willow:‘Tora’	Arable land
Bohndorf (BD)	Bohndorf I	D	53°10′33″	10°38′52″	1.2	2006	2	2009	Willow: ‘Tordis’,‘Inger’	Grassland
Bohndorf (BD)	Bohndorf II	D	53°10′31″	10°37′53″	1.5	2008	1	–	Willow: ‘Tordis’	Grassland
Bohndorf (BD)	Bohndorf III	D	53°10′18″	10°37′37″	1.7	2007	1	–	Willow: ‘Tordis’	Grassland
Cahnsdorf (CD)	Cahnsdorf	D	51°51′30″	13°46′05″	1.6	2006	2	2008	Poplar: ‘Japan 105’	Arable land
Djurby (DJ)	Djurby	S	59°41′20″	17°16′34″	2.3	1990	5	2006	Willow: ‘L78101’,‘L78021’	Arable land
Franska/Kurth (FK)	Franska	S	59°49′10″	17°38′28″	0.7	1994	5	2007	Willow: ‘Anki’,‘Astrid’,‘Bowles Hybrid’, ‘Christina’,‘Gustaf’,‘Jorr’,‘Jorun’, ‘Orm’,‘Rapp’,‘Tora’,‘L78021’	Arable land
Franska/Kurth (FK)	Kurth	S	59°48′29″	17°39′25″	1.2	1993	4	2007	Willow: ‘L81090’,‘L78021’	Arable land
Hamerstorf (HT)	Hamerstorf	D	52°54′36″	10°28′06″	3.2[a]	2006	1	–	Poplar: ‘Hybrid 275’,‘Max 4’,‘Weser 6’; Willow: ‘Tora’,‘Tordis’,‘Sven’,1 unknown	Grassland (*Populus*), arable land (*Salix*)
Hjulsta (HS)	Hjulsta I	S	59°31′55″	17°03′00″	3.0	1995	4	2008	Willow: ‘Jorr’	Arable land
Hjulsta (HS)	Hjulsta II	S	59°32′01″	17°02′54″	6.2	1995	4	2008	Willow: ‘Jorr’	Arable land
Lundby (LB)	Lundby I	S	59°40′42″	16°57′18″	1.2	1995	3	2005	Willow: ‘L78021’	Arable land
Lundby (LB)	Lundby II	S	59°40′44″	16°57′43″	9.5	2000	2	2005	Willow: ‘Tora’	*Salix* (died), before 1995: arable land

D Germany, *S* Sweden

[a] *Populus*, 2.1; *Salix*, 1.8 ha

sites. The landscape structure is described in the result section under the subheading "Landscape structure and the landscape SRC diversity effect on γ-diversity".

Spatial Analyses

Spatial analyses were conducted to test how SRC plantations contribute to species diversity of the surrounding landscape and to look for structural elements that are indicative for the SRC contribution to landscape γ-diversity. The spatial scale γ-diversity referred to is not explicitly defined [7, 39], but Whittaker [40] distinguished γ-diversity (species diversity of a landscape comprising more than one community type) from ε-diversity that describes the diversity of geographical areas across climatic or geographic gradients. The reference area for γ-diversity is about 100 km^2, but for ε diversity it is about 10^6 km^2 [41]. We defined the landscape scale in terms of areas of 225 km^2 for the evaluation of γ-diversity, and those areas were overlaid with CORINE (Coordinated Information on the European Environment) Land Cover data [42]. The availability of those data for both Sweden and Germany enabled us to evaluate structural landscape attributes on the same database. Base year for the land cover data was 2006. CORINE provides land cover data on three different levels [42]. Higher levels cumulate land cover classes of the lower level. The broadest classification is 'level 1' distinguishing the five land cover classes 'Artificial surfaces', 'Agricultural areas', 'Forest and semi-natural areas', 'Wetlands' and 'Water bodies'. All five classes of level 1 were present in our study areas. Twelve classes were present on level 2 and 21 on level 3 (Table 1).

Floristic and SRC Vegetation Assessment

For comparing SRC vegetation data with the diversity of the higher landscape scale, species lists from the nation-wide German floristic mapping [43] and region-wide Swedish mapping (for the province of Uppland) [44] were used. The data were provided by the German Federal Agency for Nature Conservation (BfN) and by the Swedish Species Information Centre (ArtDatabanken, SLU) for 5×5-km map excerpts. Nine map excerpts—one with the SRC in the centre, and eight bordering map excerpts—were used to determine the reference areas for the higher landscape scales in order to avoid any SRC being located close to the margin of the map area. The entire set of maps encompassed approximately 225 km^2 area (15×15 km). Flora species lists were simplified to species level to avoid overestimations.

SRC vascular plant species abundance was recorded in 2009 from May until July in Germany and from July until August in Sweden. At each SRC site, the species in 1,600 m^2, corresponding to 144 plots of about 11 m^2 size, were assessed in four 400 m^2 areas (20×20 m). For each plot a species list was compiled. The nomenclature follows Rothmaler [45].

Data Analysis

In a first step, species–area curves from SRC vegetation mappings were calculated to determine the minimum area for representative species numbers [46] and to test the representativeness of our 1,600 m^2 plots for deriving SRC plantation α-diversity values. For all area units (one plot to 144 plots), species numbers of all possible plot permutations [cf. 47] were calculated and averaged per unit area by EstimateS 8.2.0 [56].

In a second step, the relationship between the SRC diversity and the γ-diversity was investigated. A linear positive relationship would indicate that the share of SRC diversity on γ-diversity does not change with increasing γ-diversity. The contribution of SRC plantation α-diversity to plant γ-diversity of the surrounding landscapes, defined here as 'landscape SRC diversity effect', was calculated by Eq. 1 where α-diversity is the species number recorded in 1,600 m^2 SRC plantation, and γ-diversity is the species number found on landscape scale (225 km^2).

$$\text{landscape SRC} - \text{diversity effect} = \frac{\alpha - \text{diversity}}{\gamma - \text{diversity}} \qquad (1)$$

Linear regression analysis and test of homoscedasticity of residuals was applied using γ-diversity as predictor variable and landscape SRC diversity effect as response variable. To determine whether SRC variables and landscape matrix variables were significant predictors of the 'landscape SRC diversity effect' and of 'γ-diversity' (landscape matrix variables only, Fig. 1), multiple regression analysis was conducted. For the response variable 'γ-diversity', Poisson regression for count data was used (procedure PROC GENMOD, SAS 9.2)

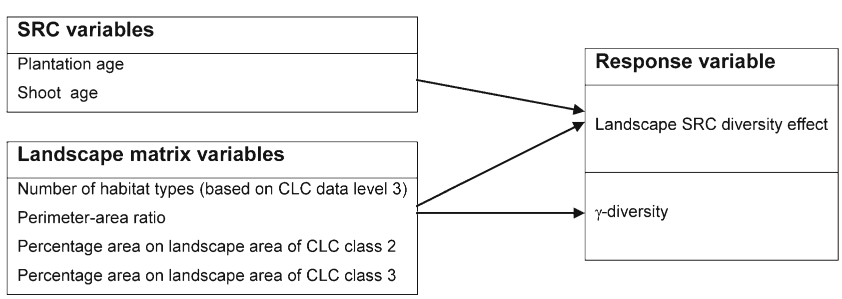

Fig. 1 SRC variables and landscape matrix variables included in multiple regression analyses for the response variables 'landscape SRC diversity effect' and 'γ-diversity'. *CLC class 2* agricultural areas, *CLC class 3* forest and semi-natural areas

and overdispersion was corrected by Pearson's χ^2. The landscape matrix variable 'perimeter–area ratio' (P: perimeter, A: patch area, cf. [48]) was calculated by Eq. 2:

$$P/A = \sum_{i=1}^{m} P_i / \sum_{i=1}^{m} A_i \tag{2}$$

The decision on the best-fitted model was based on the Akaike information criterion (AIC), in which a smaller value indicates a better fit of a model. However, the AIC does not provide information on the absolute model fit, i.e. its significance has to be tested. Inter-correlations among explanatory variables were investigated with Pearson's product moment correlation. Since no significant correlations were found (significance level: $p<0.05$), multiplicative interactions were not included in multiple regression analysis.

To compare landscape SRC diversity effect and γ-diversity, the plants were assigned to plant communities according to Ellenberg et al. [49]. The Shapiro–Wilk test was applied to test the proportions of plant communities for normal distribution. For normally distributed data the *t* test was applied to compare plant community proportions of SRC plantations with those of the landscape. For data not normally distributed the non-parametric Mann–Whitney *U* test (two-sided) was chosen.

Results

Representativeness of SRC Vegetation Samplings and Its Relationship to Landscape γ-Diversity

The species–area curves validated our sample size of 1,600 m^2 per SRC plantation as suitable for comparisons with the γ-diversity (Fig. 2). The increase in species number with area size slowed down rapidly from area sizes above approximately 200–300 m^2 sampled area. At areas between circa 600 and 1,000 m^2, 90 % of the species recorded in 1,600 m^2 were detected. As the sample size is representative, SRC plantation size was excluded from multiple regression analysis.

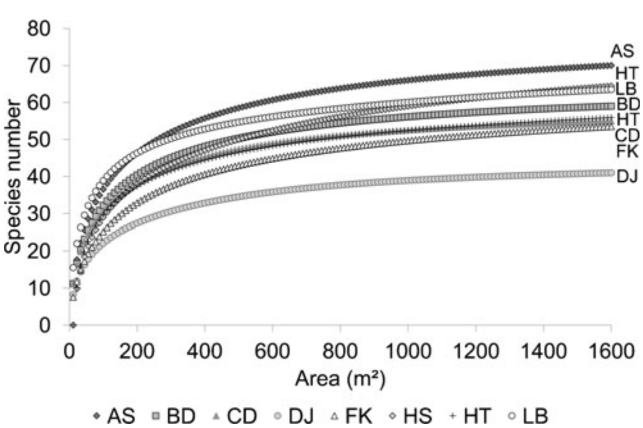

Fig. 2 Species–area curves of the SRC plantations. All possible permutations of the 144 plots per SRC plantation were calculated and averaged per area unit (1 plot=11.11 m^2). Abbreviations of SRC plantation names see Table 1

No linear relationship was found for SRC α-diversity vs. landscape γ-diversity (R^2=0.16, p=0.3290, Fig. 3a) indicating a variable contribution of SRC diversity to landscape diversity with increasing γ-diversity.

Landscape Structure and the Landscape SRC Diversity Effect on γ-Diversity

All study areas were dominated by non-irrigated arable land (34–58 % land cover) and coniferous forests (19–31 % land cover, Table 2). With the exception of 30 % water body cover at study area Hjulsta and 10 % cover of discontinuous urban fabric at study area Franska/Kurth, the proportion of all other land cover was below 8 %. The number of habitat types in the study areas ranged from 10 to 16 (CORINE land cover (CLC) data level 3) for 110 to 139 habitat patches. No relationship between number of habitats and number of habitat patches was found.

The species number for landscape (γ-diversity) ranged from 659 to 1,084 (Table 3). The SRC plantations encompassed 41 to 70 species. The species proportion of 1,600 m^2 SRC plantations on 225 km^2 of the surrounding landscape varied between 4.6 and 9.0 % (mean, 6.9±1.7 % standard deviation). The lower the species number of the landscape, the higher was the landscape SRC diversity effect (Fig. 3b, R^2=0.72, p=0.0077).

Explanatory Variables on γ-Diversity and Landscape SRC Diversity Effect

The significant model with the best AIC value was the one including all four landscape matrix parameters (Table 4), whereas only the number of habitat types influenced γ-diversity significantly (Table 5). The γ-diversity increased with increasing number of habitat types.

Multiple regression models with the response variable 'landscape SRC diversity effect' were calculated for all possible combinations of the variables: SRC plantation age, SRC shoot age, number of habitat types, perimeter–area ratio, percentage area CLC class 2, and percentage area CLC class 3. Two models were significant ($p<0.05$) and the 'landscape SRC diversity effect' was best explained by the model including the number of habitat types and the SRC shoot age (Table 6). Both the number of habitat types and the SRC shoot age were negatively related to the 'landscape SRC diversity effect' but this was only significant for the number of habitat types (Table 7, overall model: R^2=0.71, p=0.0459). Linear regression analysis resulted in an increasing 'landscape SRC

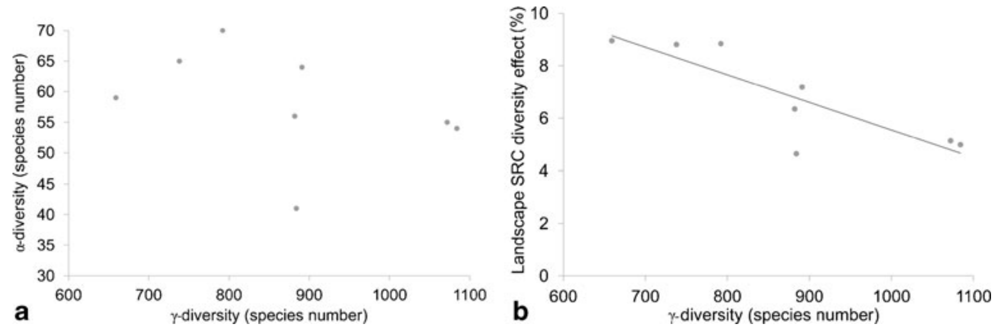

Fig. 3 Relationship of α- and γ-diversity: **a** scatterplot of SRC species number (α-diversity) and landscape species number (γ-diversity) and **b** linear regression analysis of the landscape SRC diversity effect on γ-diversity (%) vs. γ-diversity. R^2=0.72, p=0.0077. Regression equation: y=−0.0105x+16.08. Area SRC plantations, 1,600 m^2; area landscapes, 225 km^2; N=8

Table 2 CORINE land cover levels and land cover proportions of the study landscapes

CLC code	CLC level 1	CLC level 2	CLC level 3	AS	BD	CD	DJ	FK	HS	HT	LB
111	Artificial surfaces	Urban fabric	Continuous urban fabric					1		<0.5	
112	Artificial surfaces	Urban fabric	Discontinuous urban fabric	2	2	4	1	10	<0.5	6	3
121	Artificial surfaces	Industrial, commercial and transport units	Industrial or commercial units			1		4		1	1
122	Artificial surfaces	Industrial, commercial and transport units	Road and rail networks and associated land	1			1	1		<0.5	<0.5
124	Artificial surfaces	Industrial, commercial and transport units	Airports			<0.5				<0.5	
131	Artificial surfaces	Mine, dump and construction sites	Mineral extraction sites	<0.5	<0.5	<0.5					
133	Artificial surfaces	Mine, dump and construction sites	Construction sites	<0.5		<0.5					
141	Artificial surfaces	Artificial, non-agricultural vegetated areas	Green urban areas			<0.5		1		<0.5	<0.5
142	Artificial surfaces	Artificial, non-agricultural vegetated areas	Sport and leisure facilities	<0.5	<0.5	<0.5	<0.5	1			<0.5
211	Agricultural areas	Arable land	Non-irrigated arable land	57	56	55	58	35	34	46	57
231	Agricultural areas	Pastures	Pastures	1	3	10	2	1	1	3	2
242	Agricultural areas	Heterogeneous agricultural areas	Complex cultivation patterns	<0.5	<0.5		<0.5	1	1	2	<0.5
243	Agricultural areas	Heterogeneous agricultural areas	Land principally occupied by agriculture, with significant areas of natural vegetation	1	3	4	1	2	1	4	2
311	Forest and semi-natural areas	Forests	Broad-leaved forest		3	1	<0.5	1	2	2	
312	Forest and semi-natural areas	Forests	Coniferous forest	26	31	19	25	31	20	31	29
313	Forest and semi-natural areas	Forests	Mixed forest	3	1	1	1	3	7	5	1
324	Forest and semi-natural areas	Scrub and/or herbaceous vegetation associations	Transitional woodland-shrub	6		1	5	3	3		3
333	Forest and semi-natural areas	Open spaces with little or no vegetation	Sparsely vegetated areas			1					
411	Wetlands	Inland wetlands	Inland marshes			1	1		<0.5		<0.5
511	Water bodies	Inland waters	Water courses					<0.5			
512	Water bodies	Inland waters	Water bodies	1			2	8	30		

Table 3 Diversity of landscapes (γ-diversity, 225 km^2) and SRC plantations (1,600 m^2)

Country	Area and SRC site	Species numbers SRC	Landscape	Landscape SRC Diversity effect (%)
S	Åsby	70	792	8.8
D	Bohndorf	59	659	9.0
D	Cahnsdorf	55	1,072	5.1
S	Djurby	41	884	4.6
S	Franska/Kurth	54	1,084	4.9
D	Hamerstorf	56	882	6.3
S	Hjulsta	65	738	8.7
S	Lundby	64	891	7.1

D Germany, *S* Sweden

diversity effect' with decreasing number of habitat types (R^2= 0.60, p=0.0242).

Plant Communities

The SRC plantations had a higher proportion of species assigned to plant communities of frequently disturbed and anthropo-zoogenic habitats than landscape species pools. The proportion of species in the plant communities 'herbaceous vegetation of frequently disturbed areas' and 'anthropo-zoogenic heathlands and lawns' was greatest in both the landscape species pools and the SRC plantations (Fig. 4). The greatest difference between plant communities in the landscape species pools and the SRC plantations occurred for the proportion of 'freshwater and bog vegetation' species, which was 14 % in the landscape species pools and almost absent in the SRC plantations. 'Deciduous forests and related heathland' species reached 13 % in SRC plantations and 14 % in the landscape species pool. Nineteen percent of the species found in SRC plantations and 8 % of the landscape species pool comprised indifferent species with no real affinity for a particular community. The standard deviations showed that variations between SRC plantations were greater than between landscape species pools.

Table 4 Relative goodness-of-fit-test of the multiple Poisson regression models explaining the γ-diversity: only models with significant variables are shown

Number in model	AIC	SBC	Variables in model	Significance
1	58.4212	58.5801	c	sig
2	51.4753	51.7136	cd	c sig
2	51.8684	52.1067	ce	c sig
2	51.4586	51.6969	cf	c sig
3	45.9765	46.2942	cde	c sig
3	45.2899	45.6077	cdf	c sig
3	44.6970	45.0147	cef	c sig
4	39.2852	39.6824	c d e f	c sig

Response variable: γ-diversity (species number)

AIC Akaike information criterion, *SBC* Schwarz criterion, *c* number of habitat types, *d* perimeter–area ratio, *e* percentage area CLC class 2, *f* percentage area CLC class 3, *Sig.* significant

Discussion

High Landscape SRC Diversity Effect on γ-Diversity

The results show that α-diversity of small-scale (<10 ha) SRC plantations (1,600 m^2 in area) can contribute considerably to plant species richness in larger landscapes (γ-diversity, 225 km^2) accounting for a share of 6.9 % (±1.7 % SD, Table 3) on average. This is in line with Kroiher et al. [31] who found an 8 to 12 % contribution to landscape species richness when comparing similar-sized SRC stands with landscape units nine times smaller (25 km^2). For other land uses (arable land, forests, fallow and grassland), Simmering et al. [11] also found a similar mean share of 10 % of α-diversity of different sized patches to γ-diversity, although these findings related to a considerably smaller agricultural area (0.2 km^2 area). The species–area relationship (cf. Fig. 2) indicated a study size of 1,600 m^2 per SRC plantation is representative for this type of analysis. In accordance with our results, Kroiher et al. [31] showed the increase in species slowed down rapidly above 200–400 m^2 sample area for a poplar SRC plantation in central Germany. We conclude that larger SRC plantations of several hectares on homogenous sites will not result in any further increase in plant species richness and their 'diversity effect' over smaller SRC plantations, and probably rather decrease diversity. Therefore, we recommend planting several smaller SRC plantations instead of one large one, i.e. larger than 10 ha, the maximum plantation size studied here. SRC plantations of different ages, rotation regimes and tree species enhance structural diversity providing habitats for species with different requirements and are thus beneficial for species diversity [50, 51].

Less Species and Habitats in a Landscape Increase the Importance of SRC Plantations for γ-Diversity

Our study is the first report to show a clear relationship between landscape structure (number of habitat types), γ-diversity and the contribution of SRC plantations to γ-diversity across two European landscapes (Fig. 3, Table 7): In accordance with the mosaic concept [35, 36], the species number for the landscapes increased with increasing number

Table 5 Multiple Poisson regression analysis: results of the effect of landscape matrix variables on γ-diversity

Analysis of maximum likelihood parameter estimates

Parameter	DF	Estimate	Standard error	Wald 95 % confidence limits		Wald χ^2	Pr>χ^2
Intercept	1	5.9413	0.4992	4.9629	6.9197	141.65	<.0001
Number habitat types	1	0.0820	0.0130	0.0565	0.1074	39.82	<.0001
P/A ratio	1	−0.0069	0.0143	−0.0350	0.0212	0.23	0.6295
(%) CLC 2	1	−0.0011	0.0025	−0.0059	0.0038	0.18	0.6695
(%) CLC 3	1	0.0022	0.0072	−0.0118	0.0162	0.09	0.7596
Scale	0	1.6182	0.0000	1.6182	1.6182		

The scale parameter was estimated by the square root of Pearson's χ^2/DOF

P/A ratio perimeter–area ratio, *(%) CLC* percentage surface on landscape area covered by CLC class, *CLC class 2* agricultural areas, *CLC class 3* forest and semi-natural areas

of habitat types. The more diverse the landscapes and the higher the number of habitat types, the lower was the share of SRC plantations on vascular plant γ-diversity. This indicates that SRC plantations are most beneficial for flora diversity in rural areas with low habitat type heterogeneity, by providing habitats suitable for many species.

Unlike Poggio et al. [52], who analysed the relationship between the quotient perimeter/area and γ-diversity in cropped fields and edges, we found no increasing diversity with increasing landscape complexity expressed by the perimeter-to-area ratio. Edges between biotope types often contain a rich flora and fauna [13, 36], so that smaller mosaic patches with their comparatively longer ecotones enhance biodiversity of a landscape [36]. Wagner and Edwards [53] showed edges of arable fields and narrow habitats contributing more to species richness than the interior of arable fields and meadows. However, the species present at the edges are intermixed subsets of the adjacent plant communities, and only few species are expected to be present only at edges [13]. We speculate that land cover data on a greater scale than CORINE land cover could provide further information on the relationships between diversity and patch sizes as well as edge lengths. Our results do not confer with one hypothesis of the mosaic concept which claimed the surface proportions of natural, semi-natural and intensely cultivated areas influenced biodiversity, which was also confirmed by Simmering et al. [11]. The landscapes studied here were all dominated by non-irrigated arable land and coniferous forests; all other habitat types comprised only very small percentages of land cover. Thus, the landscapes we analysed may be unsuitable for sound exploration of this hypothesis because only few habitat types dominated the landscapes and their land cover percentages were similar for all landscapes.

SRC Plantations Increase Habitat Variability on Landscape Scale

Due to our study design we were not able to identify plant species that are exclusively found in SRC plantations, since they were also included in the assessments on landscape scale. However, it could be demonstrated that the SRC stands provide a large habitat variability suitable for species of many different plant communities. This becomes apparent particularly when considering the large difference in area between SRC plantations and the landscapes regarded (cf. Fig. 4): three plant communities each contained more than 10 % of the species present (19 % species had no real affinity for a particular community), whereas, in the landscape species pools, the percentage species of four communities accounted for more than 10 %. The SRC plantations species composition differs greatly from other land uses common in agricultural landscapes. This was shown by Baum et al. [54] who compared species diversity of arable lands, forests and grasslands and found that species composition of SRC plantations differed especially from arable lands and coniferous forests. SRC

Table 6 Relative goodness-of-fit of the multiple regression models explaining the 'landscape SRC diversity effect': only models with significant variables are shown

Number in model	R^2	AIC	SBC	Variables in model	p model
1	0.60	5.403	5.56185	SRC shoot age	0.0242
2	0.71	4.8601	5.09839	SRC shoot age, number of habitat types	0.0459

AIC Akaike information criterion, *SBC* Schwarz criterion

Table 7 Parameter estimates of multiple regression analysis modelling the influence of the number of habitat types and the SRC shoot age on the 'landscape SRC diversity effect'

Variable	Estimate	Standard error	Pr>|t|
Intercept	16.347	2.846	0.0022
Number habitat types	−0.646	0.213	0.0291
SRC shoot age	−0.513	0.375	0.2296

Overall model: R^2 =0.71, p=0.0459

plantations can contribute to landscape diversity by creating new habitats with species composition different from other land uses. Even though SRC plantations are an extensive land use, they contributed mainly to plant diversity by contributing species of disturbed and anthropogenic environments. The proportion of species assigned to plant communities of frequently disturbed and anthropo-zoogenic habitats was higher in SRC plantations than in the landscape species pools. SRC plantations contain predominantly common species and only few studies report the presence of rare species [cf. 25]. Analyses of Baum et al. [54] have shown that SRC plantation age does not affect species number, but species composition. They found a positive relationship between SRC plantation age and SRC tree cover along with a decrease in grassland species proportion and an increase in woodland species proportion. Considering this temporal habitat heterogeneity promoting light-demanding and ruderal species after SRC establishment and rotation cuttings and woodland species later on, SRC plantations can host many different species groups in comparably small areas. The SRC plantations contain a subset of the landscape species pool that comprises on average a share of 6.9 %, and by creating new habitats with species composition different from other land uses, these plantations have a high value for landscape diversity.

Our results and those of many other authors (cf. introduction) have shown landscape heterogeneity as beneficial for biodiversity. The expected increase in bioenergy crop production in coming years may have negative effects on biodiversity if it results in the establishment of large monocultures [18, 55]. But, by avoiding large monocultures, planting bioenergy crops can also be an opportunity for increasing structural landscape heterogeneity and creating new habitats which enhance biodiversity in current agricultural landscapes, whereby woodland and SRC plantations are especially beneficial [15].

Conclusion

Our results show that SRC plantations provide habitats for plants with different requirements and thereby have a significant share on γ-diversity. Therefore, these plantations positively affect species diversity on the landscape scale, in particular in landscapes with lower habitat diversity. The number of habitat types and the species number in a landscape can be used to predict the contribution of SRC plantations to vascular plant diversity in fragmented agricultural landscapes. Especially in rural areas with low habitat type heterogeneity, SRC plantations are beneficial for plant diversity, where plant diversity enrichment is mainly due to the occurrence of additional species present in disturbed and anthropogenic environments.

CORINE land cover data can be used for landscape structure analyses on higher landscape scales. However, on lower scales, restrictions due to low scale of land-use data must be considered in landscape structure analysis in relation to the mosaic concept: edge effects may be neglected of habitats not distinguished by CLC. Further analyses using consistent land cover information in both Sweden and Germany will be useful

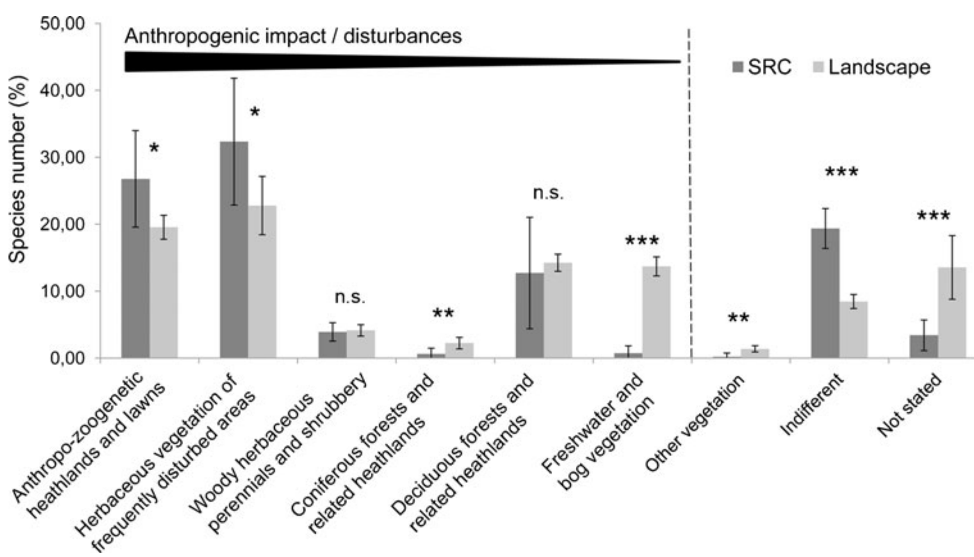

Fig. 4 Mean percentage species proportion assigned to plant communities and standard deviation of the landscapes (225 km^2, N=8) and SRC plantations (1,600 m^2, N=8). Species proportions were not significantly different between landscape and SRC plantation for 'Woody herbaceous perennials and shrubbery' (p=0.7213) and 'Deciduous forests and related heathlands' (p=0.6017). Significances: *p<0.05; **p<0.01; ***p<0.001

for further detailed landscape structure analyses of SRC plantation effects on landscapes.

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GIS-based Tools for Regional Assessments and Planning Processes Regarding Potential Environmental Effects of Poplar SRC

Gerald Busch

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Abstract Based on regional stakeholder preferences and planning guidelines as allocation criteria for SRC, this study aims at providing a transparent approach to evaluate multiple environmental effects and the regional significance of SRC systems. Using the example of two poplar SRC-systems (4-year rotation, 9-year rotation) the potential effects on ground water supply, wind erosion, and biodiversity aspects are evaluated in comparison to arable land for two selected municipalities in the district of Uelzen, Germany. Building on fuzzy membership functions and simple fuzzy-logic rules, the qualitative multi-criteria assessment is transparent and easily to adapt. This approach is transferable to other regions and spatial levels, since it derives from commonly available data and scientific evidence. Results show that implementation of SRC could provide multiple beneficial environmental effects, especially in areas with low landscape heterogeneity. The tools provided allow for a multi-criteria evaluation of environmental effects, and reveal the sensitivity to distinct allocation patterns. Physiographical conditions of the study area implicate a preference for mini-SRC systems. This is supported by smaller decline of annual deep percolation water compared to maxi-SRC. On average, decline in groundwater recharge of mini-SRC (92mm a^{-1}) is comparable to irrigated arable land (80mm a^{-1}), which is common practice in the study area. Currently, the utilization of beneficial environmental SRC effects is quite limited, since only 3 % of arable land is suitable for SRC implementation regarding farmers' preferences for SRC allocation. Allocation preferences could however change substantially with increasing incentives for SRC, e.g., due to regional bioenergy schemes or "Greening" initiatives within the European Common Agricultural Policy, which is to be reformed by 2013.

Keywords Multi-criteria evaluation · Linguistic variable · Poplar SRC · Annual deep percolation water · Stakeholder preferences · Wind erosion · Landscape heterogeneity

Introduction

Due to increasing fossil energy prices, European/national government incentives and regional bioenergy initiatives [1–3], bioenergy crops are currently cultivated on 2.28 Mio ha (19 %) of German arable land [4]. Within this highly dynamic process, the interest in planting fast-growing tree species on agricultural land is increasing in Germany. Typically, hybrid poplars and willow species are being planted in short rotation coppice (SRC) systems. Up to now, the established area of SRC is rather small and comprises about 5,000 ha — which is less than 0.5 ‰ of the arable land in Germany [5]. SRC area, however, doubled each year within the last 5-year period, and thus shows a dynamic growth rate [5]. Moreover, as a result of limited forest resources [6, 7] and increasing bioenergy demand [8], several national studies identify a substantial demand for SRC area in Germany. The numbers vary between 0.55 Mio ha and 0.9 Mio ha until 2020 or 2050 respectively [9, 10].

Despite the uncertainty of projections, demand for biomass-based energy will grow in the next decades, and energy wood from SRC will play an increasing role. Current and future cultural landscapes in Germany are being shaped by various factors ranging from global energy and commodity prices over European subsidies to European, national or regional incentives

G. Busch (✉)
Balsa - Buro for applied landscape ecology and scenario analysis,
Am Weissen Steine 4,
37085 Goettingen, Germany
e-mail: welcome.balsa@email.de

to grow annual or perennial crops (such as SRC). Since agricultural area is limited and the ecological reasonable maximum area for biomass crops in Germany is considered to be around 4 Mio ha [11], economic and ecological interests have to be balanced.

The (scientific) discussion on SRC during recent years has focused on three major issues: (1) power of SRC to compete with annual crops [12, 13], (2) management practice [14, 15], and (3) impacts on and options for nature conservation goals and areas respectively [16–18]. Several research projects (e.g., NOVALIS [18], AGROWOOD [19] DENDROM [20], and ELKE [21]) have been established dealing with these various topics. They, however, all have in common that they do not — or only marginally — address regionally specific dynamics on a landscape scale. Apart from scientific research, bioenergy/biomass production itself and its effects on the cultural landscape are not stipulated at any level of spatial planning in Germany.

Therefore, disciplines like landscape ecology or spatial planning, have to address this subject by working on 1) how to adapt existing instruments of spatial planning, and 2) how to shape cultural landscapes by implementing actor-oriented approaches effectively. In any case, it is crucial to have hands on visualization and assessment tools to allow both institutions and regional stakeholders to catch the effects of broad-scale SRC implementation on a landscape scale.

Objective

Based on regional stakeholder preferences for allocating SRC, this study aims at providing a transparent approach to visualize multiple potential environmental effects and the regional significance of poplar SRC systems on arable land with the goal to support regional planning/communication processes. Referring to fuzzy membership functions as a mean to translate and visualize both, quantitative and qualitative systems knowledge, this approach allows for discussing environmental assessments with regional stakeholders. A common scaling of the addressed environmental effects, enables stakeholders to assess potential trade-offs or win–win situations of distinct SRC allocation in the study area.

Material and Methods

Study Area

Municipalities Raetzlingen and Oetzen Within the District of Uelzen

For visualisation of the results, the two municipalities Oetzen (~31 km^2), and Raetzlingen (~9 km^2) within the district of Uelzen (~1500 km^2) were selected. Landscape composition as well as physiographical processes in the district of Uelzen are quite typical for several regions in northern parts of Germany. The undulating glacial landscape is dominated by sandy soils which are associated with residual patches of degraded bogs and fens. In ground moraine areas, soil texture changes, since the proportion of silt and/or clay is increasing. The climate is humid temperate oceanic, with mean annual precipitation ranging from 620 to 690 mm (average: 651 mm a^{-1}) and an average annual temperature around 8.3 °C in the two municipalities [22]. Agricultural land use is dominated by arable farming. Pasture is restricted to smaller linear structures in floodplain areas. Forest area dominates the edge-areas of the so-called Uelzen basin (Fig. 1).

A typical crop mix in the study area comprises potatoes, barley, sugar beet, and wheat. Due to an increasing implementation of biogas plants, maize becomes more and more important [23]. The biogas plant near Oetzen (0.9MW$_{el}$), for example, influences an area of around 1,300 ha with increased cultivation of maize (assuming 0.35 ha per kw$_{el}$ of capacity, typical substrate mix and crop rotation with a 25 % share of maize). For the study area, this is equivalent to a radius of 2.5 km (Fig. 1). With the exception of floodplain areas and some lower valley sites, the ground water table is low. Hence, cropland irrigation is common practice — about 81 % of the arable land is currently being irrigated [24].

Framework for a Multi-criteria Environmental Assessment in this Study

The reference area for this assessment is arable land without any restrictions to implementing SRC, i.e., arable land outside of designated flooding areas. Pasture land was not evaluated, since it is not readily available due to EU regulations (Cross Compliance). The assessment comprises three major elements: (1) the implementation of allocation criteria based on stakeholder perspectives, regional planning guidelines, legislative regulations, and competing biomass use, (2) an evaluation of potential environmental effects of poplar SRC implementation compared to arable land, and (3) the comparative assessment of the addressed environmental effects and their regional significance. The potential environmental effects of SRC dealt with in this study are: (a) improvement of the landscape structure, (b) benefits for biodiversity, (c) reduction of wind erosion, and (d) impact on deep percolation water. Both the evaluation of environmental effects and the subsequent assessment build on so-called "fuzzy membership functions" [25, 26] as a means to translate and visualize both quantitative and qualitative systems knowledge. A membership function ranges between 0 and 1, representing absence of membership or full membership to the corresponding linguistic variable, respectively. The linguistic variable, in turn, reflects an interpretation of the underlying indicator.

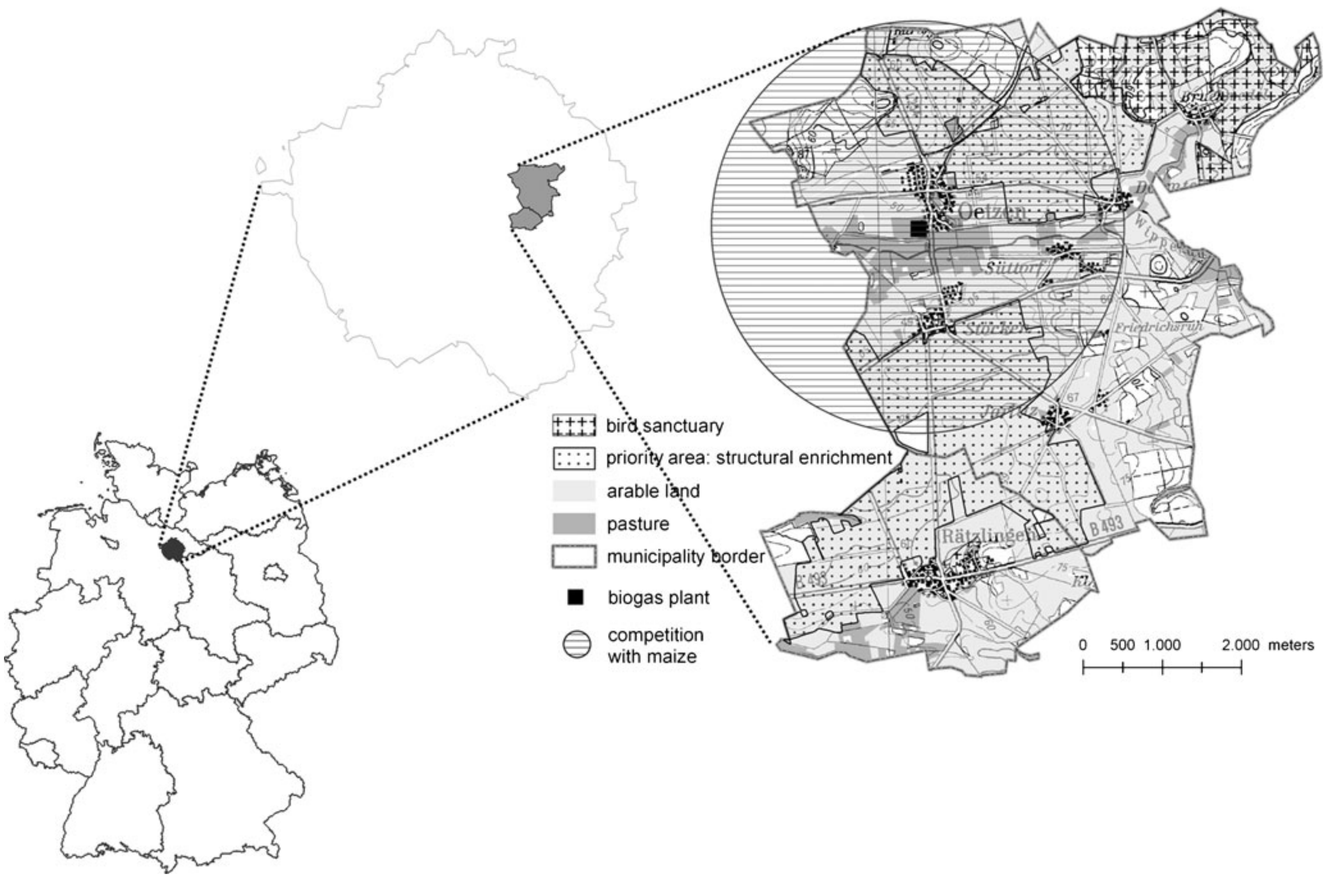


Fig. 1 Study area Oetzen/Raetzlingen in the district of Uelzen, Lower Saxony, Germany

Linguistic variables could be combined with various operators and thus flexibly reflect systems or expert knowledge concerning the underlying indicators. This approach is much more transparent than classifying, data and using a matrix approach allows for a better representation of qualitative assessments, and more complex options for combining qualitative or quantitative data as input information.

Implementation of Allocation Criteria

Stakeholder Perspectives and Legislative Regulations

Fifty-five stakeholders with various backgrounds, ranging from farmers, foresters over spatial planners, regional developers, to members of different government agencies, were questioned [18, 27] about both environmental and economical aspects of SRC implementation in the district of Uelzen. All farmers (n=20) stated that they could imagine planting up to 10 % of their area of holding with SRC. Their preference for SRC allocation on arable land would be on lower productivity sites and smaller fields (less than 3 ha) with unprofitable geometries. Further, there was a broad consensus about the maximum spatial extension of SRC in the agricultural landscape (Fig. 2).

Derived Allocation Rules from Stakeholder Preferences

SRC Preference Areas for Farmers

Smaller sites (less than 3 ha) with unprofitable geometries were addressed by using arable field size and its patch complexity as determining indicators. Arable field size was

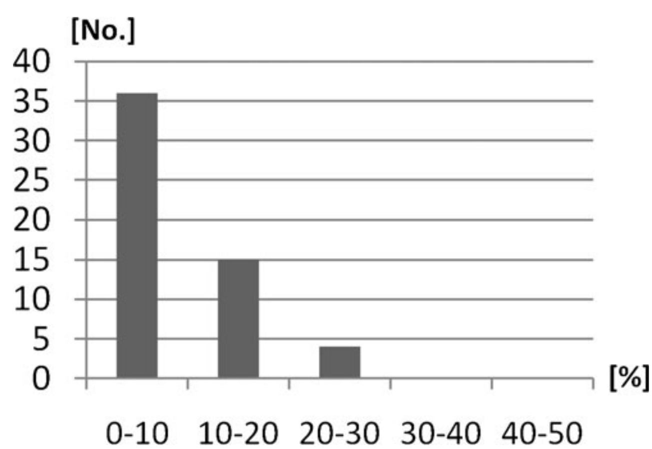


Fig. 2 Number of answers concerning the question:"What is the maximum share of SRC in the landscape that you can think of as economically and ecologically feasible?"

derived from the ATKIS land cover information system [28], and manually modified on the basis of aerial photos [29], since the digital ATKIS mapping only covers land parcels — which are not necessarily congruent with arable fields. The median field size (6.3 ha) of the study area was used to determine the medium value [0.5] of the membership function describing the linguistic variable μ_1 - "Large arable fields" (Fig. 3a). A zero membership value was assigned to a field size <1 ha (first percentile), and a field size >11 ha (ninth percentile) is equivalent to a membership value of 1.

Patch complexity [30] expressed as area/edge ratio (m^2/m) reflects both form and size of arable fields. An increasing ratio indicates a large field size with a form approximating to a rectangle. The area/edge ratio is the underlying indicator for the linguistic variable μ_2 — "Low patch complexity". A medium (0.5) membership value was assigned to the region-specific median of the area/edge ratio (i.e., 50 m^2/m) of arable fields (Fig. 3b). A ratio higher than 70 was assigned a full membership value (1), representing fields larger than 10 ha. A ratio lower than 35 represents small rectangles with distinct side length or polyangular patches. The corresponding membership values of these ratios were set to zero.

Farmers' preference for lower productivity sites was derived from existing maps on "site-specific potential arable productivity" provided by NIBIS [31]. In this approach, potential productivity was assigned to seven classes ranging from "very low potential productivity" to "very high potential productivity". The classified data were related to a fuzzy membership function [0-1] of the linguistic variable μ_3 — "High potential productivity of arable land" (Fig. 3c). A zero membership value was assigned to the two lower classes, while the two upper classes were appointed to a full [1] membership value. The membership values in between the endpoints of the membership function were derived by linear interpolation.

The linguistic variables were combined iteratively to gain "SRC preference areas for farmers" (μ^{pref}) by using the fuzzy $\gamma_{0.5}$-operator (Fuzzy Min-Max-AND). This operator reflects the "human thinking" about "and" by calculating the geometric mean between minimum and maximum values for both variables.

$$\mu^{\text{field}} = \mu(\mu_1 \hat{\wedge} \mu_2) = \sqrt{\mu_1 * \mu_2} \tag{1}$$

$$\mu^{\text{pref}} = \mu(\mu_{\text{field}} \hat{\wedge} \mu_3) = \sqrt{\mu_{\text{field}} * \mu_3} \tag{2}$$

Stakeholders' Preferences for a Maximum Share of SRC in the Agricultural Landscape

To allow for the selection of a "maximum share of SRC" in the agricultural landscape, the study area was intersected with hexagons (area of 1 km^2) as a spatial reference system. To address farmers' preferences, arable fields were chosen to a maximum (15 %) share of arable land by ranking the values of μ^{pref}. Adjacent polygons were only allowed to a maximum area of 10 ha, since this is the restriction of the "Renewable Energy Sources Act" [2] to gain full compensation for electricity fed into the grid.

GIS Implementation of Regional Planning Guidelines

Priority Areas for Improving Landscape Structure

Regional planning provides spatial development objectives by mapping large priority areas for improving landscape structure which cover around 40 % of the cropland area [32] in the two municipalities Raetzlingen and Oetzen (Fig. 1).

In these areas SRC could be implemented as enriching element, since it provides different vertical structures compared to annual crops, and increases landscape heterogeneity

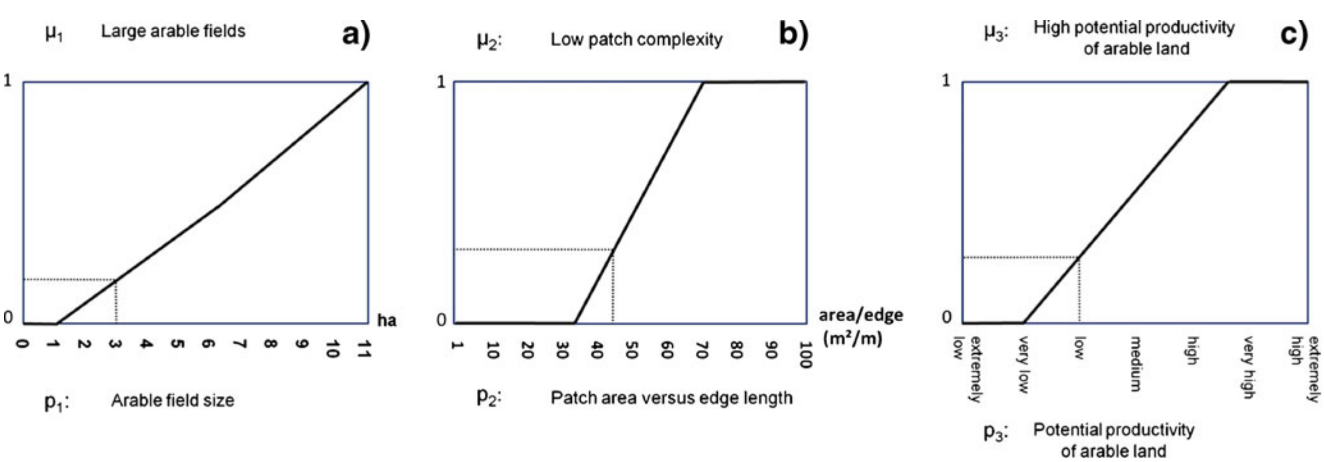


Fig. 3 Input variables and related membership functions to derive the linguistic variable "SRC preference areas for farmers". Dashed lines show the selection criteria and the corresponding membership values according to farmers' preferences

in two ways: (1) as an additional land cover type, and (2) by lowering the evenness of an agricultural landscape.

Following this rationale, improving landscape structure is interpreted as increasing agricultural landscape heterogeneity. Two indicators were calculated to catch agricultural landscape heterogeneity: (1) "Shannon diversity", and (2) “Agricultural conformity”. The "Shannon diversity” indicator refers to the number of land cover types in the reference area compared to the number of land cover types in the study area. The land cover information was derived from the ATKIS land cover information system [28]. In total, 15 land cover classes were part of the analysis. Again, hexagons with an area of 1 km^2 were used as a spatial reference.

“Agricultural conformity” was calculated per hexagon area by taking into account three parameters (1) share of arable land (2) field size of arable land, and (3) variation of field size (percentiles). Combining the two indicators catches both, the landscape mosaic of the hexagon and the structure of arable land within the hexagon.

Then, parameter values of both indicators were translated into fuzzy membership functions of linguistic variables (Fig. 4). The membership function of the linguistic variable “Low diversity” — μ_{ldiv} — was derived by taking the median value over all hexagons for "Shannon diversity" as a medium membership to μ_{ldiv}. The endpoints of the membership function [0, 1] were determined by taking the upper/lower 10 % of the input values as reference (see Fig. 4a).

To gain the linguistic goal-variable μ_{hconf} — “High agricultural conformity”, the associated membership functions (μ_4 and $\mu5$) for the parameters “Share of arable land per hexagon area”, and “Arable field size per hexagon” were derived by taking the median indicator values per hexagon as a 0.5 membership value, with the first percentile (P1) and the ninth percentile (P9) as endpoints [0, 1].

The membership function (μ_6) referring to “Variation of field size per hexagon” was derived by taking the difference between P1 and P9 of the field size variation values (interpercentile range between P1 and P10) as endpoints (Fig. 4b–d), and expressing this difference relative to the median variation. The median of the variation (0.77) in turn, was defined as the 0.5 membership value.

Finally, the linguistic variables μ_{ldiv} — “Low diversity” — and μ_{hconf} — “High agricultural conformity” — were combined to gain “Low agricultural landscape heterogeneity“ ($\mu^{heterogeneity}$) by using the fuzzy $\gamma_{0.5}$-operator (Fuzzy Min-Max-AND). The resulting map was then validated with the mapping of priority areas for improving landscape structure stemming from regional planning guidelines [31].

$$\mu^{\mathrm{heterogeneity}} = \mu\left(\mu_{\mathrm{ldiv}} \hat{\wedge} \mu_{\mathrm{hconf}}\right) = \sqrt{\mu_{\mathrm{ldiv}} * \mu_{\mathrm{hconf}}} \quad (3)$$

Potential Environmental Effects of SRC Implementation

Using Fuzzy Heterogeneity Mapping to Derive Beneficial Biodiversity Effects by Implementing SRC

Many scientific papers state that structural heterogeneity in agricultural landscapes is beneficial for biodiversity [e.g., 33, 34, and 35]. A matrix of large patches of plant communities supplemented with small patches scattered throughout the landscape characterizes a structural optimum because small patches provide different benefits for biodiversity than large patches [e.g., 36]. Baum et al. stated [37] that SRC plantation species compositions enrich landscape flora significantly, especially when habitat types are few. SRC plantations are most beneficial for flora diversity in rural areas with low habitat type heterogeneity, by providing habitats suitable for many species. In agricultural landscapes, SRC could either significantly decrease evenness or in turn increase diversity [38]. Further, implementation of SRC on arable patches promotes avifaunal ecotone species [39] and could be used as stepping-stones for habitat corridors [40]. Note that the stated biodiversity effects are especially valid for small-scale (<10 ha) SRC plantations. The fuzzy heterogeneity map, as introduced in the previous section, was used to identify areas with highest beneficial biodiversity effects, since the underlying membership function reflects a qualitative evaluation of structural heterogeneity in agricultural landscapes.

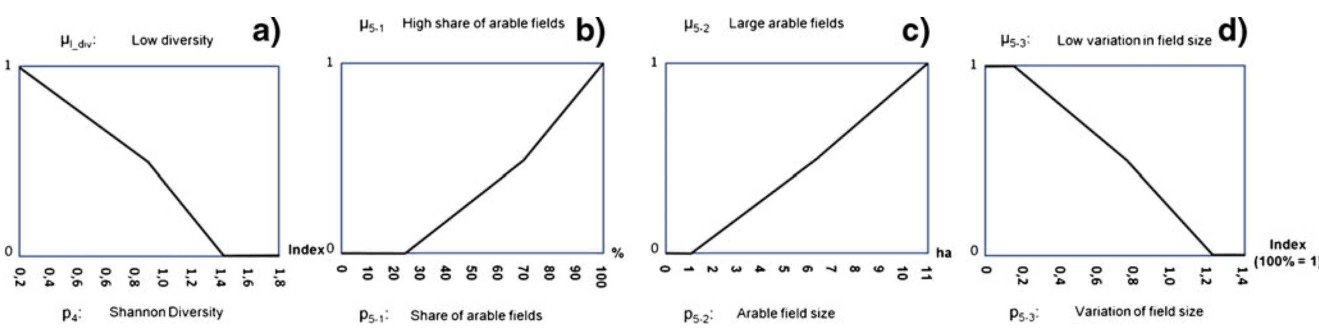


Fig. 4 Input variables and related membership functions to derive the linguistic variable “Low agricultural landscape heterogeneity”

Using Patch Complexity to Generate Ecotone Effects and Support Faunal Diversity

Several studies show [e.g., 39, 40] that species diversity (e.g., birds, butterflies, cicada, and grasshoppers) increases with rising edge length of SRC. Thus, the potential benefit of an enriched landscape structure could be amplified when allocation of SRC focuses on arable sites with a high patch complexity. With an adapted rotation management, these structures could be further diversified in both dimensions, horizontally and vertically. Patch complexity already has been calculated to address farmers' preferred sites for SRC, and could be used to derive priority areas for ecotone effects this time.

Reduced Groundwater Due to Poplar SRC on Arable Land

Rapid growth and substantial transpiration lead to a higher water consumption of SRC compared to annual crops, and could thus reduce groundwater recharge [e.g., 41, 42, and 43]. Modeling results, however, are sparse, and currently there are no extensive data available that make it possible to calculate deep percolation rates of SRC on a landscape scale.

For this study, a first approach was developed and validated that allows for evaluating annual deep percolation water of poplar SRC with respect to annual precipitation and soil texture. The basis for this approach was the empirical approach by Beims and Gutt [44]. In this approach, annual deep percolation is a non-linear function of soil texture (a, b), annual precipitation (P_y), and vegetation cover (c).

$$w_p(annual\ deep\ percolation\ water) = a^*\left(P_y - b\right)^c \qquad (4)$$

Beims and Gutt provide a matrix with predefined soil types and factors to calculate annual deep percolation water for related annual precipitation as input (Table 1).

The soil types were substituted by soil type-specific texture classes, and a multiple regression was carried out by taking the resulting w_p for all four soil types from the empirical equation and relating it to soil texture classes and annual precipitation. Soil texture was derived according to the German manual of soil mapping [45].

$$w_p = a^*\mathrm{Sand}(\%) + \mathrm{b}^*\mathrm{Silt}(\%) + \mathrm{c}^*\mathrm{Clay}(\%) + \mathrm{d}^*\left(P_y\right) \qquad (5)$$

The resulting coefficient of determination (R^2) for arable land ranges between 0.93 and 0.96 for $P_y \leq 500$ mm a^{-1} and < 1000 mm a^{-1}.

With this approach it is possible to describe annual deep percolation water according to annual precipitation over a broad range of soil textures.

To adapt this approach to the specifics of a) a 9-year-old poplar SRC, b) a 3-year-old poplar SRC, and c) the initial state (1st year) according to the data given in Table 2, an iterative fitting procedure was carried out.

Modeling results of deep percolation water (Table 2) determined the fitting procedure of the matrix factors (Table 3). The related regression function was applied to the soil texture of the modeling sites. The iteration of this procedure was carried out until the regression function produced a tolerable approximation (divergence less than 6 %). As a result, a set of regression functions are at hand that cover a wide range of annual precipitation and a broad variety of soil textures.

In a subsequent step, the findings were validated with a second data set showing modeling results of deep percolation water over two rotation periods (9 years and 3 years respectively) of poplar SRC in Brandenburg, Germany (Table 4). To validate the results of specific years with data referring to rotation periods, a two-step approach was carried out. Firstly, a function for annual seepage water was derived by using age 1, age 3 and age 9 as interpolation points. Secondly, numeric integration was carried out according to Simpson's rule to derive the mean annual deep percolation water over a specific rotation period.

Validation results (Table 4) show that the regression-based annual deep percolation water values are slightly higher than the modeling results for the two sites in Brandenburg. The maximum deviation is, however, less than 10 % und confirms the approach.

Table 1 Factors for different soil types to calculate deep percolation for arable crops

Matrix according to Beims and Gutt for arable crops						
				Soil type-specific texture		
Factor a	Factor b	Factor c	Soil Type	Clay (%)	Silt (%)	Sand (%)
0.55	250	0.96	S	3.5	4	92.5
0.52	300	0.96	Uls	12	58	30
0.49	350	0.96	Ts2	52	10	38
0.45	400	0.96	Lt3	40	40	20

Table 2 Annual deep percolation of poplar SRC on various German sites with the age (shoots) of a) 1 year, b) 3 years, c) 8–9 years

Source	clone	Soil texture (DIN 4220)			Site	Precipitation	Annual seepage water (Model)	Annual seepage water (Regression)	Deviation (%)
		Sand	Silt	Clay					
a) Poplar 8–9 years old (age 9)									
1	Max1	14	70	16	Methau	752	66	66	+0
2	Max1	15	60	25	Methau	758	62	64	+3.2
2	Max1	23	54	23	Pommritz	685	55	53	−3.6
2	Max1	23	54	23	Köllitzsch	526	37	39	+5.4
2	Max1	55	25	20	Tharandt	957	185	178	−3.8
b) Poplar 3 years old (age 3)									
4	Max1	65	25	10	Gülzow	782	228	216	−5.3
4	Max1	65	25	10	Gülzow	555	78	83	+5.1
4	Max1	65	25	10	Gülzow	635	137	139	+1.4
4	Max1	65	25	10	Gülzow	692	187	180	−3.8
3	Beaupre	75	5	20	Welzow Süd	729	194	188	−3.1
3	Beaupre	75	5	20	Welzow Süd	977	319	311	−2.6
c) Poplar 1 years old (age 1)									
2	wheat	15	60	25	Methau	758	180	173	−3.9
2	wheat	23	54	23	Pommritz	685	160	154	−3.7
2	wheat	23	54	23	Köllitzsch	526	45	43	−4.5
2	wheat	55	25	20	Tharandt	957	350	334	−4.6
3	Beaupre	75	5	20	Welzow Süd	811	354	340	−2.6

1 [41], 2 [46], 3 [47], 4 [48]

Mapping Changes in Annual Deep Percolation Water of Poplar SRC Compared to Irrigated Cropland

To derive GIS maps of SRC-related groundwater supply, arable land geometries taken from ATKIS [28] were intersected with annual precipitation numbers (long-term measurements, 1961–1990) provided by NIBIS [22] and the soil data base by LBEG [50]. The resulting geometries show a minimum resolution of 200×200m, reflecting the resolution of the climate data. Annual deep percolation water of arable land was taken from maps (scale of 1:50.000), provided by LBEG [50], and calculated with a well established rule-set supplied by NIBIS [31].

Since cropland irrigation is common practice in the study area — about 81 % of the arable land is currently being irrigated [24] — irrigated cropland serves as reference for the comparison to SRC. The amount of irrigation water is governed by the Chamber of Agriculture for Lower Saxony for a 7-year period, and adds up to around 80 mm of irrigation water annually [51]. The amount of irrigation water is interpreted as decline in annual deep percolation water, a parameter which in turn was used to shape the membership function to the linguistic variable μ_{ldecl} — "Large decline in annual deep percolation water" (Fig. 5). A decline of 80 mm a^{-1} was assigned a medium (0.5) membership value. The lower boundary (0) is equivalent

Table 3 Factors for different soil types after fitting to 9-year-old poplar conditions

Modified matrix for 9-year-old poplar with soil type-related texture classes

Factor a	Factor b	Factor c	Soil texture class Clay (%)	Silt (%)	Sand (%)
0.56	280	0.89	3.5	4	92.5
0.52	450	0.89	12	58	30
0.48	500	0.89	52	10	38
0.45	530	0.89	40	40	20

Table 4 Validation data

Source	Clone	Soil texture (DIN 4220)			Site	Precipitation	Annual seepage water (Model)	Annual seepage water (Regression)	Deviation (%)
		Sand (%)	Silt (%)	Clay (%)					
Poplar 9-year rotation period									
1	275	75	5	20	Neuruppin	590	70	74	5.7
1	275	65	25	10	Neuruppin	590	65	68	4.6
1	275	75	5	20	Lindenberg	634	87	92	5.7
1	275	65	25	10	Lindenberg	634	82	89	8.5
Poplar 3-year rotation period									
1	275	75	5	20	Neuruppin	590	127	131	3.1
1	275	65	25	10	Neuruppin	590	114	124	8.7

[1] [49]

to no change in deep percolation water. The upper boundary of the membership function results from linear interpolation, and corresponds to a decline of 160 mm a^{-1}.

Fuzzy SRC Suitability Mapping: Sites with High Yield Potential

From the scientific literature ([17, 52–54]), a set of appropriate site conditions for poplar SRC is known that covers various parameters of climate, soil, and landform conditions. These criteria, however, only allow for a binary distinction between suitable and unsuitable sites, and many of them are not relevant for this study. The only potential limiting factors are (1) an ample water supply during the vegetation without water logging effects, and (2) steeper slopes (>7°).

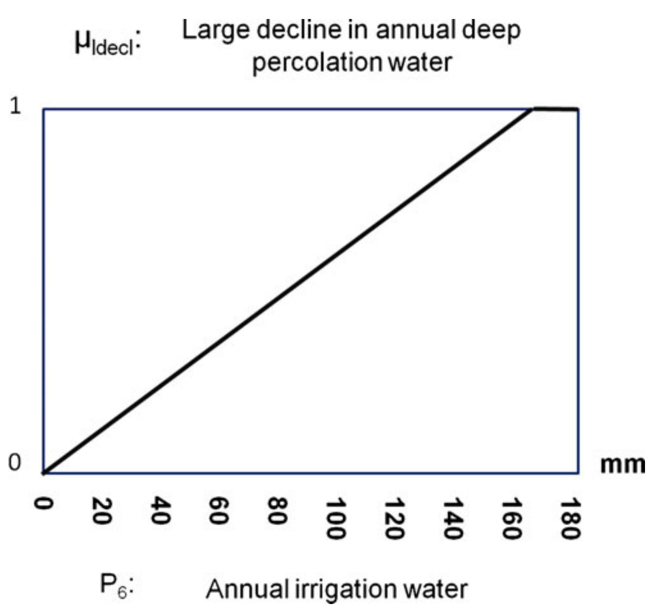


Fig. 5 Input variable and related membership function to derive the linguistic variable "Large decline in annual deep percolation water"

This issue was reflected by addressing two input parameters for the qualitative evaluation of site suitability: (1) "available transpiration water", and (2) slope.

"Available transpiration water" (ATW) is a function of precipitation, actual evapotranspiration, rooting depth, "available soil water capacity" (ASWC), and capillary rise. Rooting depth, in turn, is determined by soil texture, precipitation, bulk density, water logging, and groundwater table. The overall approach follows a methodology described by Busch [42], and opposes SRC water demand to water availability from soil and precipitation during the vegetation period. Actual evapotranspiration determines SRC water demand, and was derived from the balance of annual precipitation and deep percolation water (as described in the section on the calculation of annual deep percolation water). Water supply, in turn, is determined by precipitation ASWC and capillary rise. Over the rotation period, surface runoff could be neglected, since there is a permanent soil vegetation cover. The balance of water supply and water demand during the vegetation period determines ATW.

Again, NIBIS soil and climate data were used to derive precipitation, ASWC, and capillary rise. Based on NIBIS soil information, rooting depth was calculated, assuming an exponential decline in root distribution down to 1.20 m of soil depth, following data provided, for example, by Raissi [55], Crow & Houston [56], and Petzold et al. [57]. Calculations for the study area turned out that a surplus of at least 125 mm of water is necessary to sustain optimal growth conditions during the vegetation period. When defining the linguistic variable μ_7 — "Sufficient productive transpiration water" to reflect ATW, this value was set as a threshold for a full membership (Fig. 6a). A zero membership was assigned to a balanced demand–supply equation, revealing the fact that this is accompanied by lowered productivity due to decreased available soil water.

The parameter “slope” was derived from a digital elevation model with a mesh size of 25 m [58]. For the qualitative evaluation, slope parameter values were associated to a membership function referring to the linguistic variable μ_8 — "Chipper suitability". A slope steeper than 7° defines the lower [0] boundary of the membership function, while slopes flatter than 2° determine the upper [1] boundary. All values between the endpoints were interpolated linearly (Fig. 6b).

Again, the fuzzy γ-operator (Fuzzy compensatory-AND) was used to represent the membership function of the linguistic variable μ^{hp} — "Potential high yield site" when combining $\mu_7$with μ_8.

$$\mu^{\mathrm{hp}} = \mu(\mu_7 \hat{\wedge} \mu_8) = \sqrt{\mu_7 * \mu_8} \tag{6}$$

Potential Impact of Climate Change on Annual Deep Percolation Water and SRC Site Suitability

Since SRC is a perennial culture with a cultivation horizon of at least 20 years, it is important to consider potential climate changes — especially changes in level and distribution of precipitation. It is outside the scope of this study to provide an extended analysis on impacts of precipitation changes on SRC. However, WETTREG2010 data [59] were analyzed to check both dimension and orientation of potential precipitation changes up to the decade of 2030 to 2040. WETTREG2010 represents a statistical regionalization method based on global climate modeling results (ECHAM5) for underlying IPCC A1B scenario conditions [59]. For this study, regionalized precipitation data of WETTREG2010 weather station data in a rectangle of 900 km^2 around the study area were used as an input to calculate and compare annual precipitation and precipitation during the vegetation period (1 April to 31 October) as decadal averages for 1961–1990 (P1), and 2030–2040 (P2) respectively. Station data were interpolated by using inverse distance weight (IDW) interpolation. IDW interpolation is a common routine — here applied in an algorithm using six stations with a search radius of 40 km influencing the interpolation.

Beneficial Effects of Poplar SRC on Areas with High Disposition to Wind Erosion

Apart from the initial phase of plantation establishment, SRC fields show almost no on-site wind erosion, since the soil is covered with herbaceous plants and leaves. Thus, areas prone to wind erosion could substantially benefit from SRC implementation.

To evaluate areas with a high disposition to wind erosion, two major aspects have to be addressed: (1) soil sensitivity to wind erosion, and (2) potential impacts from land use under given climatic conditions. Soil sensitivity to wind erosion was adopted from the NIBIS soil information system [31]. The input information — classified parameter values ranging from very low to very high soil sensitivity — were transformed into a membership function that determines the linguistic variable $\mu_{soilsens}$ — “High soil sensitivity to wind erosion” (Fig. 7a). Class values “Low risk of wind erosion [0], and “High risk of wind erosion” [1] were set as lower and upper boundary of the membership function.

The potential impact of land use on soil erosion is determined by crop type, crop rotation, crop management, field size, and exposition to wind. The crop-specific (including crop management) impact on soil erosion was taken from an assessment carried out by NLÖ [60]. Information on crop rotation was obtained from municipal statistics by calculating the average crop rotation (area-weighted) over the last decade. Again, these crop-specific input parameters were translated into a linguistic variable "Protective crops" (μ_{cprot}), and the parameter values were transformed into a membership function (Fig. 7b). Again, the classified values provided by NLÖ were linearly interpolated with determining class values “1” and “9” as lower and upper boundaries respectively of the associated membership function.

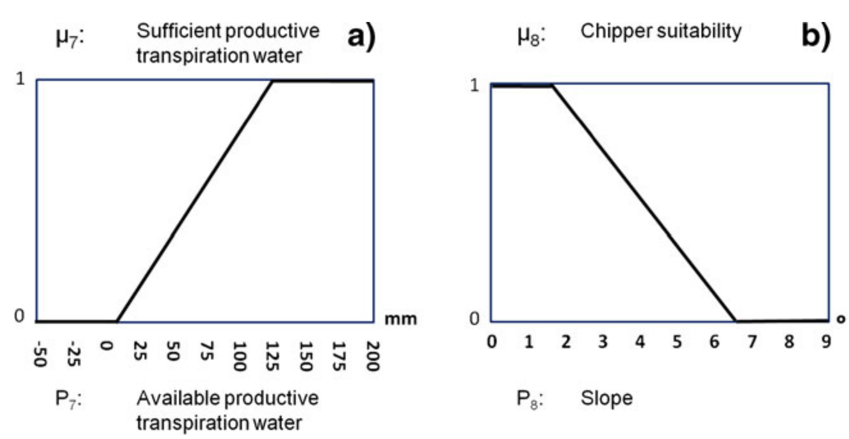


Fig. 6 Input variables and related membership functions to derive the linguistic variable “Potential high yield site”

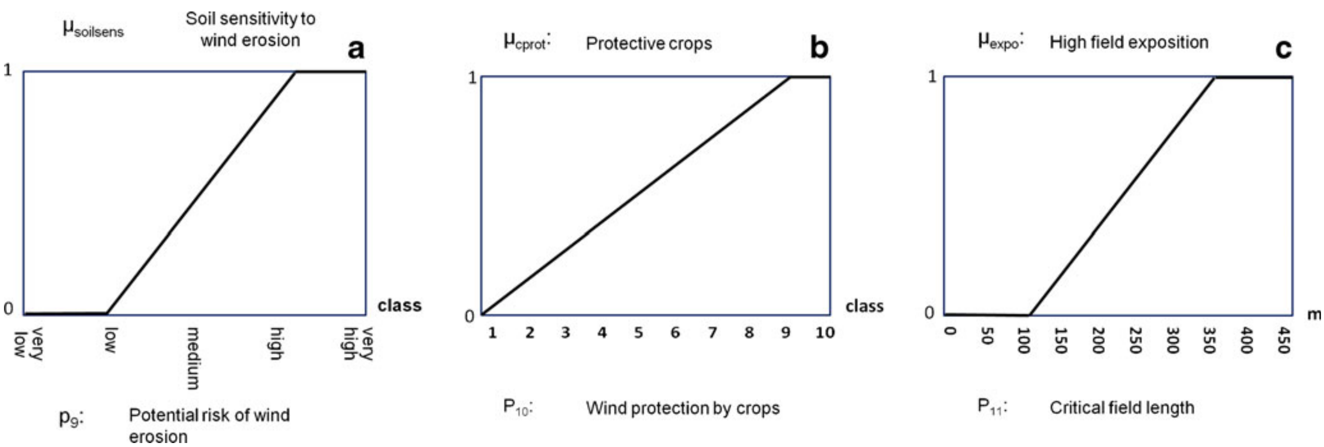


Fig. 7 Input variables and related membership functions to derive the linguistic variable "Field disposition to wind erosion"

Dominant wind direction and wind speed >8 ms^{-1} were derived from weather-station data (Fassberg) nearby the study area. Field size and exposition to wind determine whether wind erosion could become critical or not. Thiermann et al. [61] provided an assessment of critical field length exposed to major wind directions. This information was linked to agricultural field geometries (derived from aerial photos and ATKIS land cover maps) and crop information via GIS. The resulting values were transformed to obtain the membership function of the linguistic variable (μ_{expo}) — "Exposition" (Fig. 7c). The two variables were combined to the linguistic variable μ^{f_expo} — "Field exposition to wind erosion" — by using the $\gamma_{0.5}$-operator (Fuzzy Min-Max-AND). The protective effect of existing perennial vegetation structures (e.g., hedgerows, forests) was calculated by implementing a GIS algorithm according to the information provided by Müller et al. [62], and Hennings [63].

$$\mu^{\mathrm{f\_expo}} = \mu\left(\mu_{\mathrm{cprot}} \hat{\wedge} \mu_{\mathrm{expo}}\right) = \sqrt{\mu_{\mathrm{cpro}} {}^{*} \mu_{\mathrm{expo}}} \qquad (7)$$

Finally, the resulting linguistic variable "Field disposition to wind erosion" — μ^{dispo} — was derived by combining the two linguistic variables $\mu^{soilsens}$ and μ^{f_expo}. The outcome of this calculation represents fields prone to wind erosion due to a combination of land use and natural site conditions.

$$\mu^{\mathrm{dispo}} = \mu\left(\mu^{\mathrm{f\_expo}} \hat{\wedge} \mu^{\mathrm{soilsens}}\right) = \sqrt{\mu^{\mathrm{f\_expo}} * \mu^{\mathrm{soilsens}}} \qquad (8)$$

Results

Suitable Areas for Poplar SRC

Precipitation during the vegetation period (1 April to 31 October) was calculated with 397 mm as average value for the study area (range of 372–421 mm). Mean actual evapotranspiration during the vegetation period was calculated with an average of 430 mm for Mini-SRC and 460 mm for Maxi-SRC respectively. Thus, an average deficit of around 30 mm for Mini-SRC or 60 mm for Maxi-SRC has to be compensated by ASWC. The qualitative evaluation in terms of "Potential high yield sites" reveals that the SRC water demand could not adequately be balanced on all sites by showing distinct regional differences in suitability for Mini-SRC, and mapping a considerably lower site suitability for Maxi-SRC (Fig. 8). Consequently, the focus should be on Mini-SRC, since only 15 % of the arable fields show major constraints for productivity (i.e. membership values < 0.5).

The most productive sites are situated in southwestern areas of the study region, which is largely congruent with priority areas for structural enrichment. Thus, if structural enrichment is an option, the focus should be on this region, since SRC faces both productivity constraints and additional competition with maize in the northern parts of the study area. For Maxi-SRC, due to higher demand of transpiration water, "site suitability" is considerably lower. Taking a value of 0.5 as threshold for suitability, only 50 % of the arable sites could be designated as adequate sites. The most productive sites (values >0.85) are restricted to more silty soils at lower slope or valley areas on around 15 % of the arable land (Fig. 8).

Reduced Annual Deep Percolation Water Due to Poplar SRC on Arable Fields

Due to vast areas with sandy soil texture, annual groundwater recharge from arable land is high. Ranging between 88 mm a^{-1} and 265 mm a^{-1}, the average annual groundwater supply is about 223 mm. On 2,817 ha of arable land this adds up to an amount of 6,244 Mio m³ a^{-1}, or around 34 % of annual precipitation (Table 5).

Irrigation measures of around 80 mm a^{-1} reduce the annual amount of groundwater recharge by about 36 %, or 2,259 Mio m^2 for the study area. For Mini-SRC systems,

Fig. 8 Membership value for the linguistic variable "Potential high yield sites" for (**a**) Mini-SRC and (**b**) Maxi-SRC in the study area

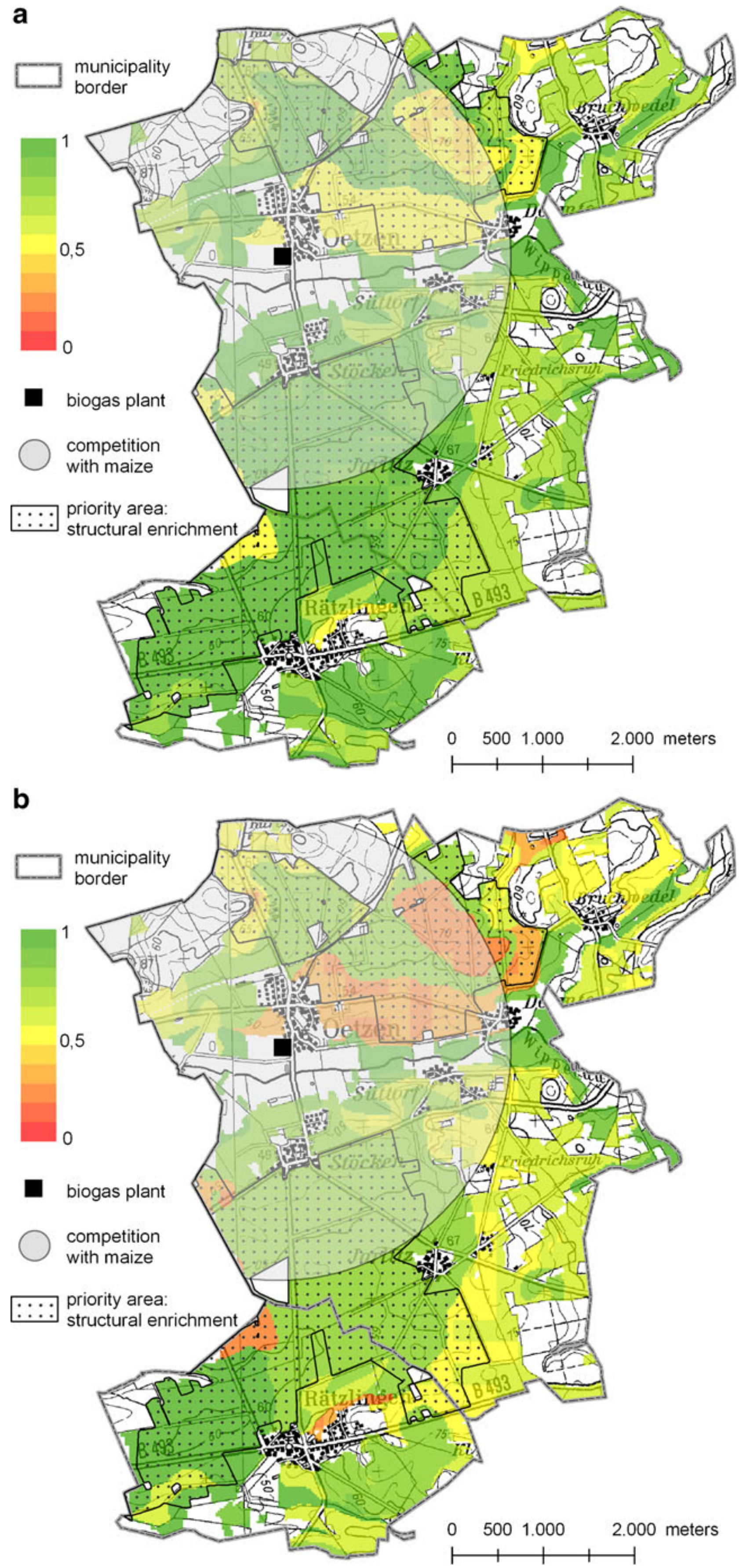


Table 5 Characteristics of groundwater recharge (GWR) in the study area. A comparison of arable land with (a) irrigated crops, (b) Mini-SRC and (c) Maxi-SRC

	Arable land	Irrigated crops	Mini-SRC (% change to irrigated crops)	Maxi-SRC (% change to irrigated crops)
Annual GWR (mm)	223	143	131 (−8 %)	100 (−30 %)
Change in annual GWR (mm)	-	−80	−92	−123
% change to non-irrigated situation	-	−36 %	−41 %	−55 %
Areal sum(Mio m³ a^{-1})	6,244	3,996	3,684	2,810
Change in areal sum	-	−2,248	−2,560	−3,434

average annual groundwater supply lessens to 131 mm, which is equivalent to a further reduction of 8 % compared to irrigated arable crops. However, when taking regional stakeholder preferences for a maximum of 10–15 % of SRC area into account, annual decline of deep percolation water is only around 1 % compared to irrigated crops. Even the considerably higher water demand of Maxi-SRC (Table 5) modifies to a 5 % decline in groundwater recharge when assuming this spatial restriction.

The higher the productivity (suitability), the higher the decline in annual deep percolation water is the overall picture for sandy soil textures (Fig. 9). With increasing silt and clay content, deep percolation rates are reduced, and the difference between SRC and irrigated crops diminishes. Therefore, if maximum SRC productivity should be combined with minimum decline of annual deep percolation water, the focus should be on sites with a lower sand fraction. Implemented as elements of agroforestry systems, SRC could reduce evapotranspiration in adjacent arable fields [64], and thus eventually compensate for the own increased water use.

To summarize: implementation of both Mini-SRC and Maxi-SRC diminishes annual groundwater recharge compared to annual crops. In areas where groundwater protection (quantity) is an issue, the focus of SRC implementation should be on mini-rotation systems. On a landscape level, groundwater reduction due to SRC plays a minor role, when its spatial extent is restricted to less than 20 %.

The Potential Impact of Climate Change on SRC Suitability

Interpolation results of precipitation in the 1961–1990 period (WETTREG2010 data) are very similar to the reference data base of this study (Table 6). Thus, precipitation changes derived from the WETTREG2010 data analysis could be assigned to the reference data of this study. The analysis of the WETTREG2010 data revealed (Table 6) that both precipitation (+2.5 %) during the vegetation period and annual precipitation (+5 %) are expected to increase up to 2030–2040. The higher precipitation increase during the non-vegetation period (Table 6) is in line with other sources [e.g., 65, 66] which suppose that autumn and winter precipitation will increase. The small increase of precipitation during the vegetation period is supported by the findings of Meinke et al. visualized in the regional climate atlas of Germany [66]. Since these different regionalization approaches based on the same global climate model (ECHAM5-OM) come to similar outcomes for this particular region, it could be concluded for this study that suitability and thus productivity of poplar SRC is not negatively affected by climate change within the next 2 decades. Moreover, increasing precipitation during wintertime could result in slightly higher groundwater recharge compared to the current situation.

Disposition to Soil Erosion as a Potential for SRC Implementation

Sandy soils in combination with region-specific crop rotation result in a considerable "field exposition" to wind erosion. More than 60 % of the arable sites show membership values higher than 0.8 (Fig. 10a). Present structures, e.g., hedgerows but mainly the arable patch characteristics (field length and exposition to major wind directions), substantially decrease the risk of wind erosion on many sites (see Fig. 10b). There are, however, 900 ha, or 32 % of arable land, with a high disposition to wind erosion (membership value >0.85) — a circumstance that reveals a substantial need for wind-protection measures. In terms of erosion protection, these are priority areas for SRC implementation. Especially in priority areas for landscape structure, or in the radius of the biogas plant, an implementation of SRC could generate additional win–win effects by providing structural elements and compensating for the erosion-promoting effects of maize. Another positive win–win effect by implementing SRC on erosion-prone sites could be the protection from topsoil loss, and thus reduced organic carbon. In contrast, SRC sites are even expected to accumulate carbon over time [67]. Apart from implementing SRC on arable fields with disposition to wind erosion, an effective erosion

Fig. 9 Membership values of "Large decline in annual deep percolation water" for (**a**) Mini-SRC, and (**b**) Maxi-SRC (a decline of 80 mm a^{-1} as reference is equivalent to a 0.5 membership value)

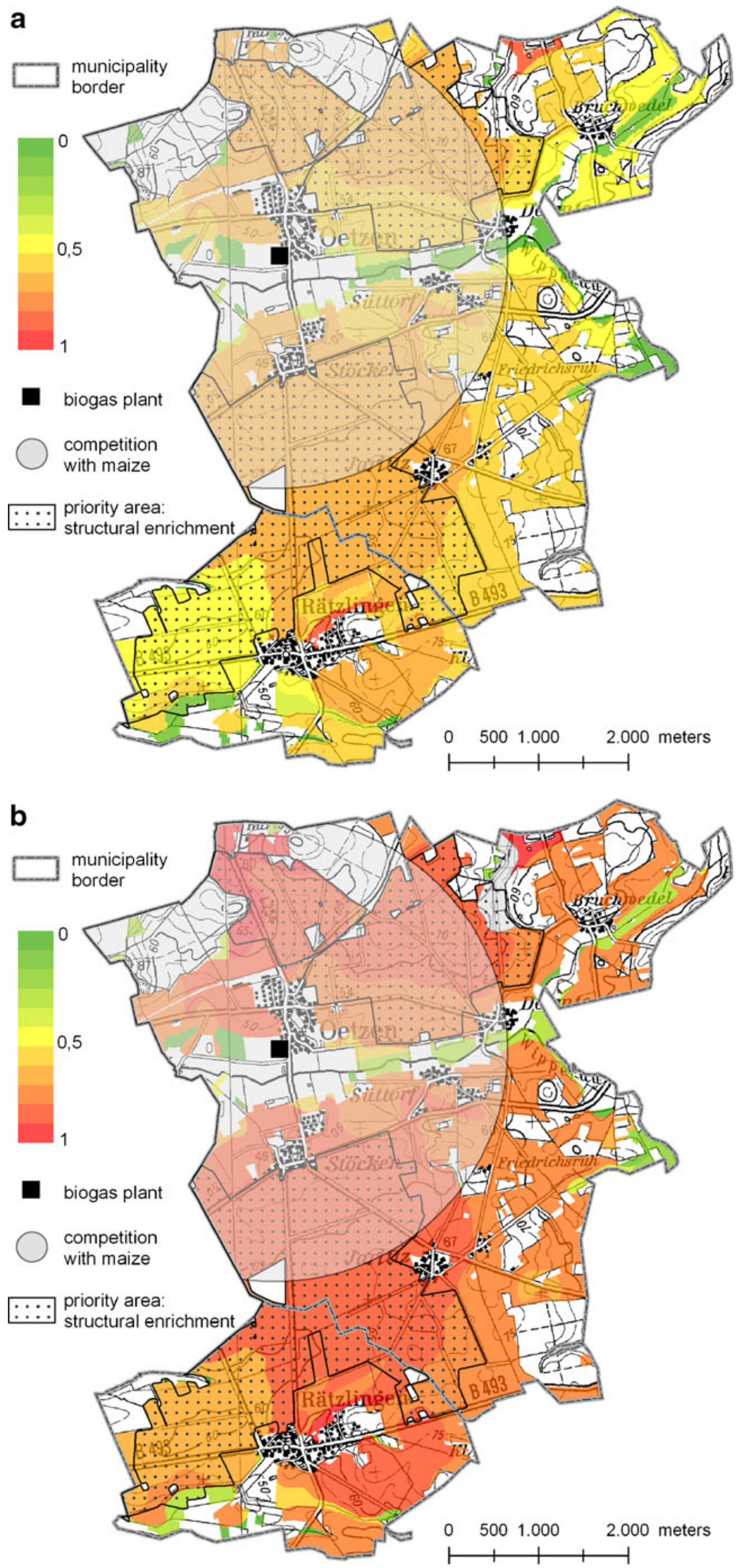


Table 6 Precipitation changes according to a WETTREG 2010 data analysis for the study

	Long-term average 1961–1990						Decadal average 2030–2040					
	Year			Vegetation period			Year			Vegetation period		
Precipitation (mm)	min	avg	max	min	avg	max	min	avg	max	min	avg	max
WETTREG	618	654	690	381	402	423	652	686	721	390	412	429
NIBIS (reference)	620	651	686	380	400	420						

protection program should focus on establishing an adapted hedgerow system or, alternatively, promote agroforestry systems containing SRC strips.

Agricultural Landscape Heterogeneity and Arable Field Complexity as Indicators for Promoting Biodiversity Aspects Through SRC Allocation

The linguistic variable "Low agricultural landscape heterogeneity" provides valid results, since its high membership values obviously cover core areas with deficits in landscape structure (Fig. 11a). Further, the qualitative mapping allows for a specific selection of areas to promote flora diversity. The empirical relationship found by Baum et al. [38] states beneficial flora biodiversity effects of SRC compared to other land cover types (e.g., grassland, deciduous forest) in agricultural landscapes with low heterogeneity. As a consequence, SRC implementation in areas with high membership values to "Low agricultural landscape heterogeneity" could endorse both structural heterogeneity and flora diversity. Note, however, that the results provided by Baum et al. [38] address species quantity and diversity, which is not necessarily congruent with regional nature conservation goals.

Building on the aforementioned criteria, a selection of complex field forms (Fig. 11b) could enhance the biodiversity value of implemented SRC for flora as well as for (ecotone) fauna. In terms of environmentally-oriented land use, SRC implementation could generate notably win–win effects in the overlapping areas of increased maize cultivation and structural deficits.

Planning Guidelines Meet Stakeholders' Perspectives and Farmers' Preferences — Comparative Assessment of the Addressed Environmental Effects and their Regional Significance

The results presented so far dealt with environmental effects of a potential SRC implementation and visualised options for an environmental-oriented SRC allocation. In this section, both the farmers' perspective (addressed by the linguistic variable "SRC preference areas for farmers" — see Materials and methods section) and regional stakeholders' opinion ("Maximum share of SRC" — related to priority areas for structural enrichment) were used to allocate potential SRC sites. The resulting allocation patterns, in turn, built the ground for a comparative assessment of the addressed environmental effects.

"SRC Preference Areas for Farmers"

Farmers' current preferences for SRC implementation on less productive sites with small field sizes and an unprofitable layout led to a selection of 95 ha of arable fields with a mean field size of 1.3 ha (Fig. 12a). With an average membership value of 0.9, the resulting field patches are pooled outside the priority areas for structural enrichment on around 3 % of arable land. This is not surprising, since deficits in landscape structure are generally correlated with high-yield sites. However, one potential benefit from SRC (i.e., promoting biodiversity by providing structural heterogeneity), is substantially reduced. In areas with a high existing heterogeneity, SRC implementation has to be carefully evaluated. Comparably high patch complexity, in turn, endorses ecotone-related biodiversity aspects.

Site suitability for Mini-SRC is considerably high, with a median value of 0.69, and 75 % of the sites showing values higher than 0.65 (Fig. 12b). The risk of declined yields is substantially higher for Maxi-SRC, since only 50 % of the sites show membership values higher than 0.5. Thus, suitability for Maxi-SRC has to be carefully considered when implementing SRC. Decline in annual deep percolation water shows a small variation, with a median value for Mini-SRC that is congruent with irrigated annual crops. Erosion protection could not be tackled as a side-effect of implanting SRC preference sites. However, with a specific site selection, areas with high disposition to wind erosion could be addressed. This could be of particular importance when implementing SRC on sites within the radius of the biogas plant. In summary, one can conclude that several environmental win–win effects could be generated when implementing SRC on farmers' preference sites. With only 95 ha, the spatial significance of these protective effects is, however, comparably low. For a broader dispersion of

Fig. 10 Membership values for (**a**) the linguistic variable "Field exposition to wind erosion", and (**b**) "Field disposition to wind erosion" shown for arable sites in the study area

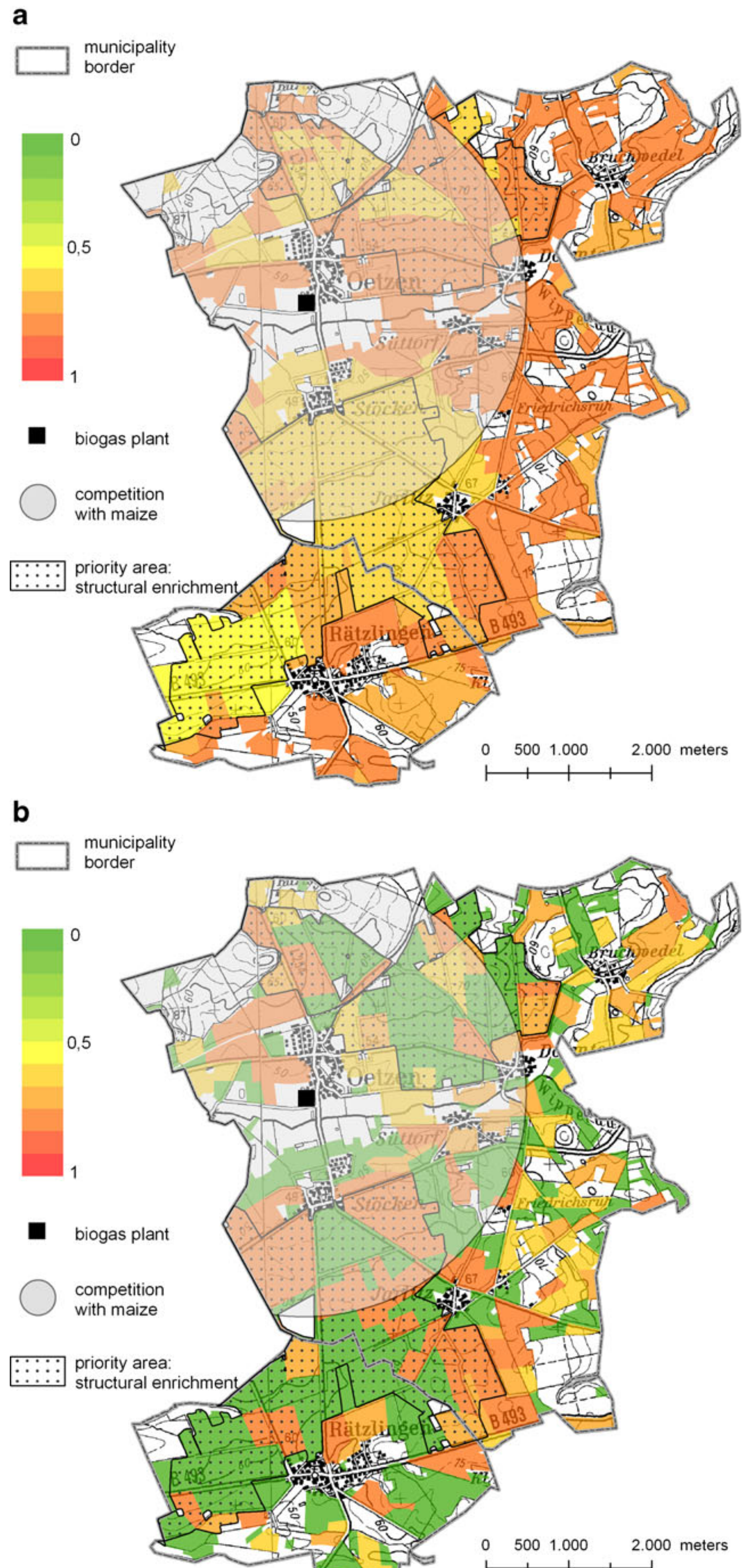
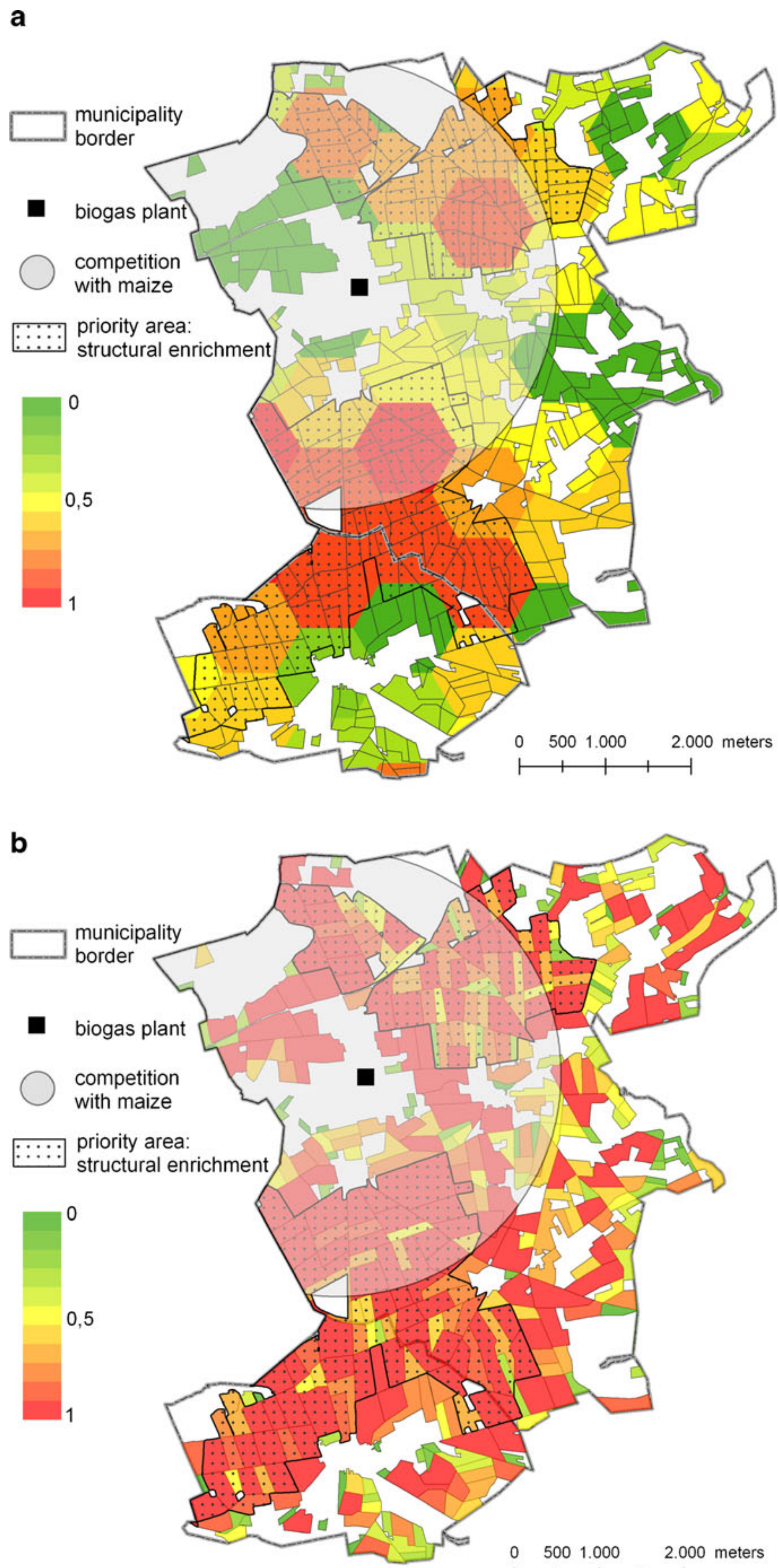


Fig. 11 Membership values for (**a**) "the linguistic variable “Low agricultural landscape heterogeneity", and (**b**) "Low patch complexity"



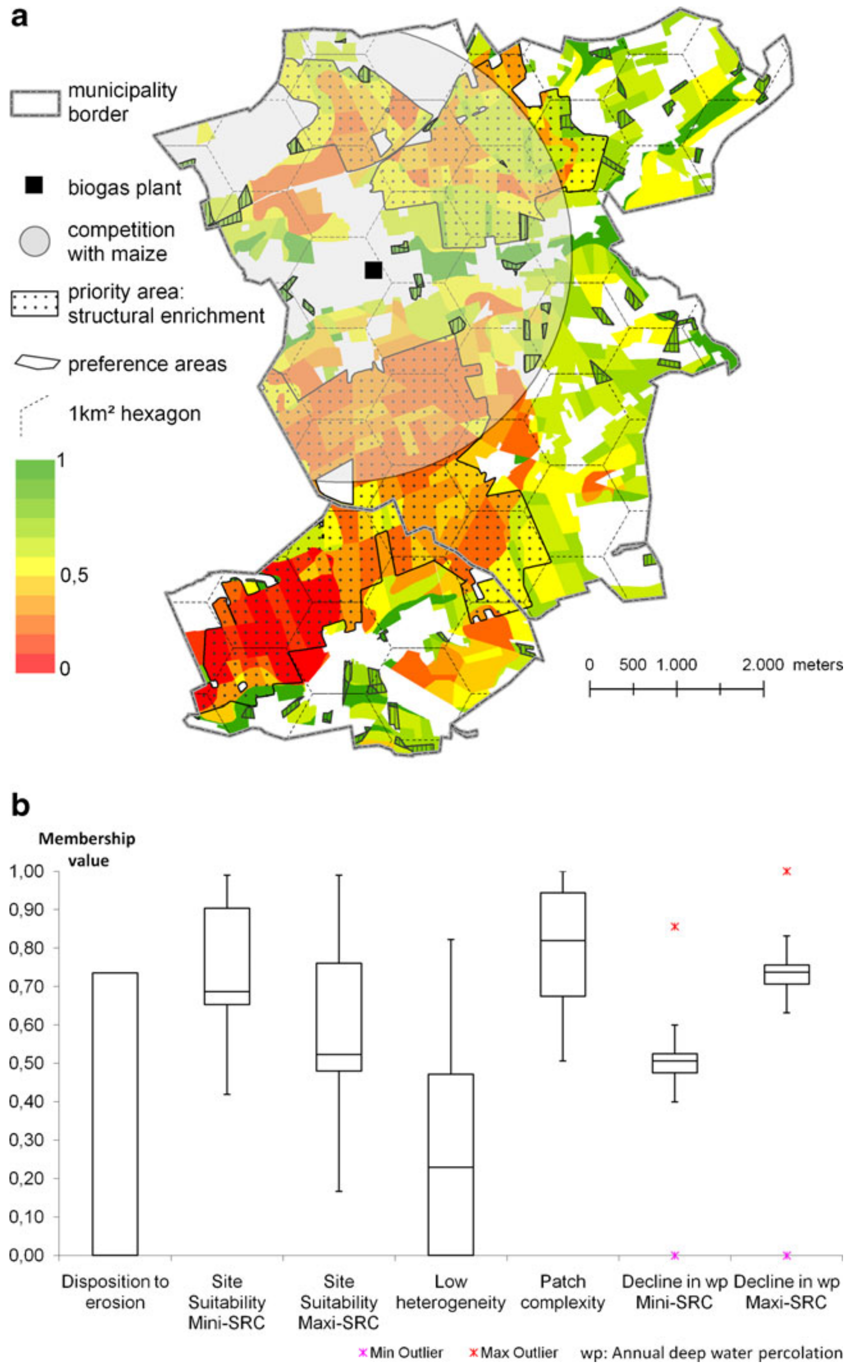


Fig. 12 (**a**) Membership values of the linguistic variable “SRC preference areas for farmers” and selected sites according to farmers’ preference thresholds, and (**b**) site-related boxplots (quartiles) for the environmental linguistic variables addressed in this study

potential SRC areas, farmers have to give consideration to fields with lower patch complexity, since this turned out to be the most restricting factor for SRC allocation.

"Maximum Share of SRC" in Priority Areas for Structural Enrichment

Potential SRC allocation covers around 129 ha or 10 % of priority areas for structural enrichment. The sites are concentrated in the northern region of the priority areas, mainly because of the comparably lower productivity for arable crops (Fig. 13a). This is reflected by the average membership value of 0.7, stating that the vast majority of sites address farmers’ preferences to allocate SRC on sites with lower productivity. Moreover, the average field size of 1.9 ha is considerably lower than the average of 6.3 ha for arable fields in the study area. Albeit, site suitability is high, showing membership values higher than 0.7 for more than 75 % of Mini-SRC potential sites (Fig. 13b). Even for Maxi-SRC, 50 % of the sites are of medium to good site quality. Erosion protection could be addressed only partially, since half of the areas show no disposition to wind erosion. Again,

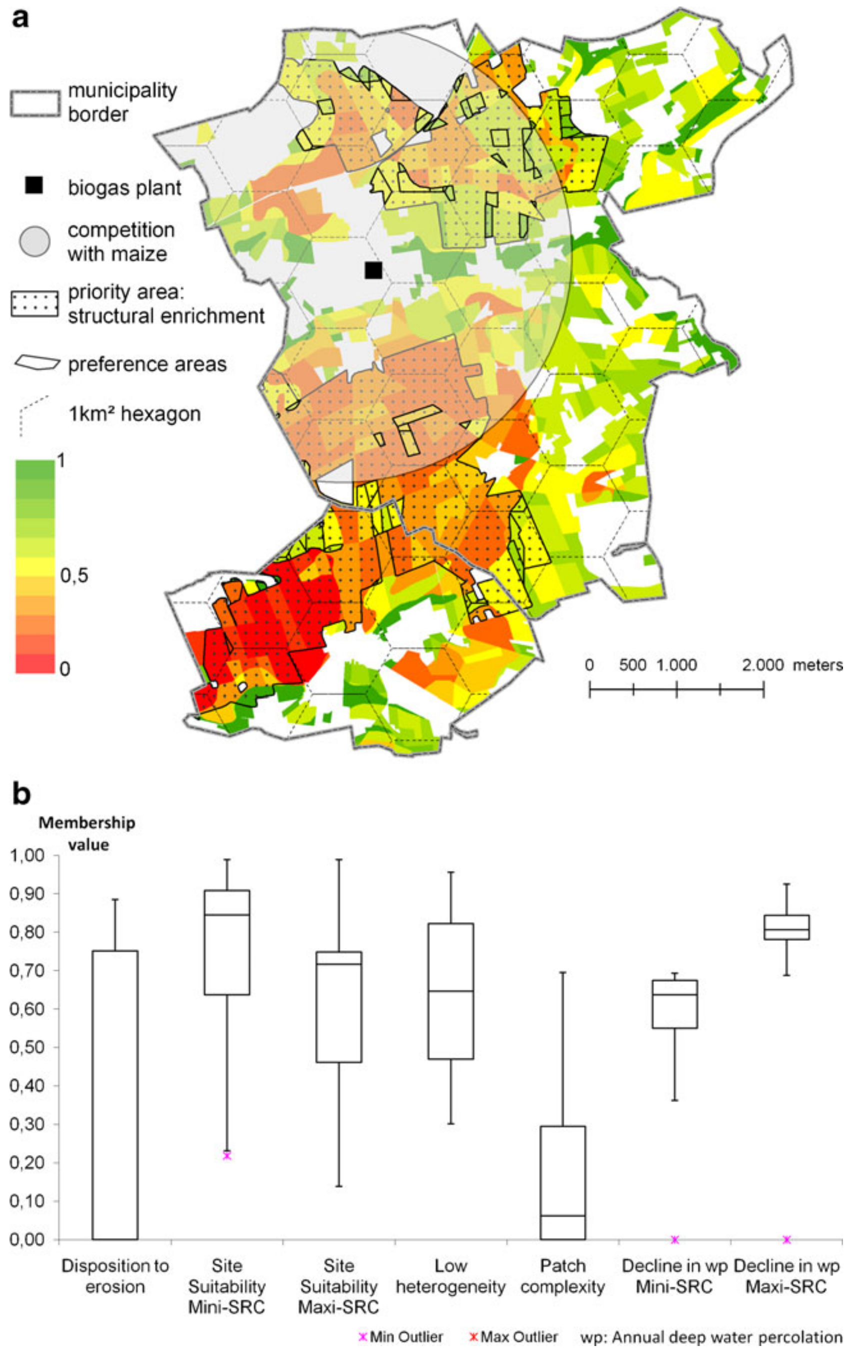


Fig. 13 **a** Membership values of the linguistic variable "SRC preference areas for farmers" and selected sites according to stakeholders' perception of "SRC maximum share", and **b** site-related boxplots (quartiles) for the environmental linguistic variables addressed in this study

it could be of particular interest to identify potential areas prone to wind erosion within the radius of the biogas plant. Higher membership values for "Low heterogeneity" reflect the SRC potential to increase structural richness and biodiversity. The broad range of values, however, illustrates the importance of site selection. Low membership values for "Patch complexity" seem to impede the promotion of ecotone effects at first sight. SRC implementation on subareas of arable fields, though, could enhance edge length significantly. As a side-effect of higher SRC productivity, the decline in annual deep percolation water is rising as well. As an example, more than 75 % of potential Mini-SRC areas demonstrate an increased decline of deep percolation water compared to irrigated crops (Fig. 13b).

Discussion and Conclusion

Results for the exemplary study area show that the presented approach provides tools that are capable of visualizing various environmental effects resulting from spatial allocation patterns. The implemented fuzzy routines as means for a qualitative environmental assessment show robust results. The underlying membership functions are based on general scientific knowledge, and could easily be adapted to regional expert knowledge. The tools could be used for adapted planning, e.g., by comparing environmental effects for various sub-regions, defining targets for specific environmental services, or meeting multi-criteria decisions when allocating SRC.

SRC-suitability mapping supports a spatial explicit analysis of priority areas with respect to soil and climate conditions as well as rotation periods. Physiographical conditions, such as sandy soils with high percolation rates, mean annual precipitation averaging around 650 mm, and a comparably low availability of transpiration water lead to a preference for Mini-SRC. There are, however, smaller patches available for the implementation of highly productive Maxi-SRC. Quantitative analysis of annual groundwater recharge revealed, by comparing arable sites with SRC, that Mini-SRC systems are similar to irrigated annual crops. Depending on site productivity, the annual groundwater recharge of Mini-SRC alternates around the annual groundwater recharge of irrigated crops. The combination of soil-related assessment routines, suitability mapping, and quantification of SRC-dependent groundwater recharge revealed that the focus of SRC implementation should be on Mini-SRC. The brief analysis of potential precipitation changes due to climate change impacts showed that SRC systems don’t have to face additional site suitability restrictions from reduced precipitation.

Analysis of disposition to wind erosion allowed for a site-specific assessment of actual erosion risk. In this context, SRC could play an integrating role in wind-erosion protection schemes, e.g., SRC areas could intermediate between existing hedgerow structures or could be a component in agroforestry areas. However, especially in regions with an area-wide disposition to wind erosion, the focus has to be on linear structures. Implementing SRC as an element of agroforestry systems could be an option, especially when scientific results provide further results that SRC strips could promote yields and reduce water consumption on adjacent arable fields.

The qualitative mapping makes it possible to address biodiversity aspects by identifying priority areas for adding horizontal and vertical structures to agricultural landscapes with low heterogeneity. Further specification according to regional conservation goals or planning guidelines could be integrated into this approach, which is, however, out of the scope of this study.

SRC implementation based on current allocation preferences of farmers is quite restricted, and therefore does not meet priority areas of structural enrichment stemming from regional planning guidelines in the study area. Farmers, even those generally interested in SRC as an option to diversify their income, go for the least attractive arable sites, since the implementation of SRC faces several problems, ranging from sparse machinery, and uncertainties of economic return, to lack of attractive supply contracts. Here, the further development of regional bioenergy policies promoting decentralized supply of heat and electricity could be a stepping-stone for SRC as an additional biomass source. Further, if SRC would be accepted as one land use option within the "Greening" initiative [68] for the 2013 reform of the European Common Agriculture Policy (CAP) , a substantial increase of SRC area could be anticipated. For this option, it is even more important to provide tools permitting rapid regional assessments as a basis for more detailed investigations.

Long-term field experience with SRC systems is still limited in Germany. Hence, quantitative data on several environmental parameters (e.g., growth of different clones under various conditions, transpiration data, root development, nutrient balance, and biodiversity aspects) are sparse or still lacking. However, to fill in a niche for both regional energy supply and agricultural landscapes with a low heterogeneity could be a strong point for SRC. This implies that SRC is not to be treated as a surrogate for maize, particularly in light of the considerably lower energy density, which it is especially important to note because the potential beneficial effects of SRC could be substantially diminished or even foiled when established as large-scale monocultures.

In spite of all gaps and uncertainties, it is essential to build on existing knowledge and to develop transparent and flexible assessment tools. For practical application, it is crucial to refer to commonly available data and to develop transferable routines. The approach at hand builds on extensively available data on soil, climate, relief, and land cover. In principle, the implemented algorithms are applicable for the whole of Lower Saxony (~48,000 km^2). The focus is, however on small to medium scale assessments (e.g., maps with a map scale of 1:25,000–100,000). Regional aspects (e.g., regional-planning targets, bio-energy schemes, conservation aspects) could be flexibly implemented. Since co-operative participation models are increasingly promoted to generate biomass-based energy, regional land use is gaining growing interest. SRC as a long-term land-use option needs a substantial initial investment and, thus, acceptance plays a key role. Here, transparent assessment tools could help to communicate and visualize various land use alternatives. As an element of a participatory planning approach, an implementation of SRC could be analyzed beforehand with regard to win–win effects, or the potential of conflict, respectively.

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Meeting Sustainability Requirements for SRC Bioenergy: Usefulness of Existing Tools, Responsibilities of Involved Stakeholders, and Recommendations for Further Developments

Oskar Englund · Göran Berndes · Fredrik Fredrikson · Ioannis Dimitriou

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Abstract Short rotation coppice (SRC) is considered an important biomass supply option for meeting the European renewable energy targets. This paper presents an overview of existing and prospective sustainability requirements, Member State reporting obligations and parts of the methodology for calculating GHG emissions savings within the EU Renewable Energy Directive (RED), and shows how these *RED-associated sustainability criteria* may affect different stakeholders along SRC bioenergy supply chains. Existing and prospective tools are assessed on their usefulness in ensuring that SRC bioenergy is produced with sufficient consideration given to the RED-associated criteria. A sustainability framework is outlined that aims at (1) facilitating the development of SRC production systems that are attractive from the perspectives of all stakeholders, and (2) ensuring that the SRC production is RED eligible. Producer manuals, EIAs, and voluntary certification schemes can all be useful for ensuring RED eligibility. However, they are currently not sufficiently comprehensive, neither individually nor combined, and suggestions for how they can be more complementary are given. Geographical information systems offer opportunities for administrative authorities to provide stakeholders with maps or databases over areas/fields suitable for RED-eligible SRC cultivation. However, proper consideration of all relevant aspects requires that all stakeholders in the SRC supply chain become engaged in the development of SRC production systems and that a landscape perspective is used.

Keywords Short rotation coppice · EU · Producer manuals · EIA · Certification schemes · GIS

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O. Englund (✉) · G. Berndes
Division of Physical Resource Theory, Department of Energy and Environment, Chalmers University of Technology, 41296 Gothenburg, Sweden
e-mail: oskar.englund@chalmers.se

F. Fredrikson
County Administrative Board of Västra Götaland, Department of Rural Development, BOX 496, 50313 Borås, Sweden

I. Dimitriou
Department of Crop Production Ecology, Swedish University of Agricultural Sciences (SLU), P.O. Box 7016, 75007 Uppsala, Sweden

Background

Bioenergy has been put forward as a potential option for improving energy security and mitigating climate change [1–3]. It offers a new market for farmers and bioenergy production has, particularly in developing countries, been proposed as a possible driver of rural development with capacity to improve energy access, increase employment, and stimulate productivity growth in agriculture. Over recent years, however, concerns have arisen regarding the true environmental, social, and economic viability of bioenergy systems, and the bioenergy sector has been put under pressure to verify the sustainability of its operations.

In response to concerns about unintended consequences of biomass production and use for energy, producers of biomass feedstock in the private sector, as well as governmental and non-governmental organizations, have taken initiatives to develop criteria and indicators for sustainable bioenergy supply chains, as a means for regulating the bioenergy sector. The sustainability certification schemes that are being developed or implemented by a variety of private and public organizations can be applicable for different feedstock production sectors (notably forest and agriculture sectors) and for different bioenergy products, ranging from relatively unprocessed forest and agriculture residues to electricity and refined fuels, such as ethanol and biodiesel. They can be applicable for entire supply chains or certain segments of a supply chain [4–7].

The heterogeneity of sustainability certification schemes, which are developed largely without coordination, might present a challenge for the stakeholders along the bioenergy supply chains that must comply with these systems to maintain market access or to comply with legislative mandates. Also, consumers who prefer to purchase certified sustainable bioenergy, and regulatory agencies and governments involved in enforcing sustainability standards, may find it difficult to manage a wide range of systems that use different criteria and indicators for the sustainability certification.

Stakeholders involved with bioenergy that is used within the European Union (EU) have to specifically consider the EU Renewable Energy Directive (RED), which mandates levels of renewable energy use within the EU and also includes a sustainability scheme for biofuels for transport as well as for bioliquids used in other sectors. It sets out criteria and provisions to ensure sustainable production and use of biofuels and bioliquids [8]. These, or similar, sustainability criteria may later be applied also for solid and gaseous biofuels; in its 2010 report on sustainability requirements for the use of solid and gaseous biomass sources in electricity, heating, and cooling, the European Commission (EC) did not propose legally binding requirements at the time, but recommended that "Member States that either have, or who introduce, national sustainability schemes for solid and gaseous biomass used in electricity, heating, and cooling, ensure that these in almost all respects are the same as those laid down in the EU RED" [9]. There is a desire to ensure greater consistency and avoid unwarranted discrimination in the use of raw materials. It can therefore be assumed that the differentiation between different types of bioenergy will be lessened in future revisions, making the RED sustainability requirements legally binding also for solid and gaseous biofuels. It should be noted that the existing sustainability requirements in RED are limited compared to most certification standards and relates only to GHG emissions and biodiversity.

Short rotation coppice (SRC) (e.g., willow or poplar) is considered an important biomass supply option for meeting the European renewable energy targets [10]. A rapid expansion of SRC, especially in agricultural areas near the end user of biomass (e.g., heat and electricity plants for direct biomass combustion), is expected in several European countries [11]. It is important to note that cultivation of SRC, although using tree species, is an agricultural practice. It should therefore be regulated through the EU-wide Common Agricultural Policy (CAP), which cross-compliance requirements include considerations on, e.g., preservation of habitats, biodiversity, water management and use, and mitigation of climate change.[1]

Despite the similarities in management practices between SRC cultivation and conventional crop production, there are two principal differences; SRC plants are perennial and the species cultivated are trees. Consequently, there are several differences in how the cultivation affects the biophysical environment. Results from experiments reported in this special issue suggest that water quality in terms of N concentration in the groundwater is significantly improved when SRC is cultivated instead of cereals, but similar positive effects in terms of P are not evident [12]. However, Baum et al. [13] reports that less erosion is to be expected when SRC is cultivated instead of other arable crops, which probably leads to less P losses associated with surface runoff. Moreover, Dimitriou et al. [12] compared total carbon content and trace elements in the soil of a number of long-term commercial willow SRC fields in Sweden with adjacent, conventionally managed arable fields. Results showed that total carbon concentrations in the topsoil and subsoil of SRC fields were significantly higher (9.4 %) than in the respective reference fields. The respective average relative increase when SRC was compared with cereals was 10.5 % in the topsoil and 26 % in the subsoil, respectively. Regarding concentration of cadmium (Cd), an average relative reduction of 12 % in the topsoil of SRC compared to cereals was found. Sewage sludge, which is commonly applied to SRC fields for nutrient recycling and additional compensation to the farmers, had no effect on the evaluated soil quality parameters. Concerning phytodiversity, positive impacts from SRC plantations can be expected with regards to species richness [14], and concerning zoodiversity and breeding birds abundance in particular, positive or negative impacts seem to be site and SRC-age specific [15].

The establishment of SRC plantations not only affects criteria of sustainability found in the RED scheme, but also the agricultural landscape as such. As discussed later in this paper, a wide range of stakeholders are either affected by, or expected to influence, the establishment of SRC plantations. Proper consideration of all relevant aspects therefore requires that all stakeholders in the SRC supply chain are engaged in the development of SRC production systems and that a landscape perspective is used. A multi-stakeholder landscape level process would facilitate linking with the

[1] Forest management is regulated on a national level, with policy guidance through the EU Forestry Strategy and international processes such as the Ministerial Conference for the Protection of Forests in Europe.

European Landscape Convention (ELC), which promotes the protection, management, and planning of European landscapes and organizes European cooperation on landscape issues. The ELC also promotes the public involvement in matters concerning the landscape. It is the first international treaty to be exclusively concerned with all dimensions of European landscapes [16].

This paper presents an overview of existing and prospective sustainability requirements as well as of Member State (MS) reporting obligations in the EU RED, and shows how these *RED-associated criteria* may affect different stakeholders along the SRC bioenergy supply chain—from feedstock producers to energy consumers. Based on this, the extent to which three different types of tools (producer manuals, environmental impact assessments, and sustainability certification schemes) can be used to ensure that SRC bioenergy is produced with sufficient consideration given to RED-associated criteria is discussed. In a concluding section, a framework for engaging relevant stakeholders in the development of SRC within a landscape perspective is outlined. This framework has two purposes: (1) to facilitate the development of SRC production systems that are attractive from the perspectives of all stakeholders; and (2) to ensure that the SRC production is RED eligible.

Methodology

Analysis of RED

As stated earlier in this paper, stakeholders involved in production of solid and gaseous biofuels that are used within the EU have good reasons to consider RED sustainability requirements, despite the fact that they are currently only legally binding for bioliquids. It is also indicated in RED that additional legally binding sustainability requirements might be added in future revisions of the directive. For example, Article 18(9b) in RED states: "By 31 December 2012, the Commission shall report to the European Parliament and to the Council on whether it is feasible and appropriate to introduce mandatory requirements in relation to air, soil, or water protection". Identifying and considering such "potential requirements" in the development of sustainability frameworks reduces the risk of having to make future adjustments.

Besides the sustainability requirements for production of bioliquids, RED requires the EC and MS to monitor and report on certain sustainability aspects of bioenergy production and use. Such obligations typically concern impacts due to production and use of bioenergy in general, i.e., no distinctions are made between liquid, solid, or gaseous biofuels. Therefore, in order to fulfill the obligations in RED, sustainability aspects related to monitoring and reporting need to be addressed for SRC. Furthermore, specific sustainability considerations can be identified in the methodology for calculating GHG emissions savings. Considering these in a sustainability framework for SRC bioenergy would support the involved stakeholders in producing bioenergy with high GHG emissions savings. Finally, RED includes a number of sustainability considerations requiring no particular actions at present. Such considerations should be noted, as they may be subject to reporting and monitoring obligations in the future, or even become additional sustainability requirements.

The RED was reviewed using the above reasoning and RED-associated sustainability criteria were formulated. The criteria were then sorted under specific categories to put them into a correct context (see also Englund et al. [17] and Electronic Supplementary Material (ESM) 1). Finally, the criteria were evaluated on their relevance for SRC bioenergy on a national level.

Inventory and Categorization of Stakeholder Landscape

The stakeholder landscape was investigated using in-house experience and stakeholder consultation, to identify principal stakeholders involved in SRC bioenergy. A general SRC bioenergy supply chain was created and the stakeholders' roles in meeting RED-associated criteria were discussed.

Analysis of Producer Manuals

Ten producer manuals were collected and analyzed. The manuals all refer to willow and/or poplar coppice production, including site selection, planting, and harvesting. The RED-associated criteria can be both directly and indirectly covered in producer manuals and also at varying level of comprehensiveness; for each criterion, manuals were assigned as having *major*, *minor,* or *no coverage*. Based on the number of producer manuals that cover each criterion—and the extent (minor or major) of coverage—the overall coverage of producer manuals was determined.

Analysis of Environmental Impact Assessments

Nineteen EIAs were collected from bioenergy projects that include the establishment of plantations or large-scale agricultural operations, and/or construction of a biofuel processing plant.

Four approaches were used to collect EIAs: (1) email inquiries to researchers and experts; (2) email inquiries to EIA consultants, certification audit companies, and development banks; (3) internet searches; and (4) asking local consultants associated to Winrock International in 18 countries to "attach any Environmental Impact Assessments (EIAs), Strategic Environmental Assessments (SEAs), or Social Impact Assessments (SIAs) you encounter related to biofuels".

Depending on the nature of the assessed bioenergy projects, EIAs were sorted into three categories: *Plantations*, *Biofuel plant*, and *Plantations and biofuel plant*. The EIAs were then assessed on their coverage in relation to the RED-associated criteria. For each criterion, an EIA was assigned one of five levels of coverage, depending on how the criterion was considered in the EIA. These individual results were then combined to indicate the coverage of RED-associated criteria in EIAs in general. Coverage levels include: *overall low coverage*; *varying coverage*; and *overall high coverage*.[2]

Sustainability Certification Schemes

An overviewing review of international sustainability certification schemes relevant for SRC bioenergy was performed. Based on this, the role of certification in national SRC bioenergy sustainability frameworks was discussed.

Results

RED-Associated Sustainability Criteria Relevant for SRC and Corresponding Responsibilities for Principal Stakeholders

Thirty-one sustainability criteria were derived from RED. These include the described existing and prospective sustainability requirements, reporting and monitoring obligations for the EC and MS, and more general sustainability considerations. On a national level, 18 of these 31 criteria are relevant for national SRC bioenergy sustainability schemes to address (Table 1, see also ESM 1 for more details). These criteria are related to:

1. Existing and prospective legally binding sustainability requirements,
2. Reporting obligations for MS, and
3. The methodology for calculating GHG emissions savings.

Throughout this paper, the term “RED-associated sustainability criteria”, “RED-associated criteria”, “RED criteria”, or “criteria” refer to the criteria presented in Table 1. “Existing RED sustainability requirements” or “RED requirements” refer to the existing, legally binding, sustainability requirements for bioliquids laid out the RED.

Principal stakeholders involved in producing SRC bioenergy include *landowners*, *entrepreneurs*, *bioenergy producers*, *end users*, *administrators,* and *legislators*. These are defined in Table 2.

The principal stakeholders are involved at different stages in the SRC bioenergy supply chain (Fig. 1). *Planting* may be undertaken by the *landowner* or, more commonly, by an *entrepreneur. Cultivation* is most often the responsibility of the *landowner. Harvesting* is typically done by an *entrepreneur* but can also be done by the *landowner. Transportation* of the harvested biomass can be done by either the *landowner*, an *entrepreneur*, or the *bioenergy producer. Processing* of the biomass into bioenergy for sale on a market is done by the *bioenergy producer*. Finally, the *bioenergy use* stage involves *end users*. In addition, *administrators* and *legislators* are in different ways involved in regulating each stakeholder in each stage of the supply chain. Therefore, specific SRC bioenergy supply chains can have different structures depending on which stakeholders are involved at the different stages.

The *bioenergy producer* is responsible for demonstrating compliance with the legally binding RED sustainability requirements and for calculating specific GHG emissions savings (if not using default values). The l*egislator* on the other hand is responsible for meeting the MS reporting obligations. Even so, the RED-associated criteria apply at all stages in the supply chain, except at the final stage where the bioenergy is being used (Fig. 1). This means that all stakeholders besides the end user can be responsible for ensuring that the criteria are considered, depending on the specific structure of a supply chain. It is therefore difficult to assign stakeholder-specific responsibility for the individual criteria, implying that all criteria should be communicated to all stakeholders. In addition, effective consideration of some criteria may require interactions between several stakeholders depending on their respective involvement along the SRC bioenergy supply chain. Providing opportunities for such interactions may be a challenge, since experiences from Sweden show that coordination between different stakeholders involved with SRC can be poor [18] (see Fig. 2, fact box). It is therefore important that a sustainability framework is designed so as to facilitate stakeholder interaction to clarify the stakeholders' respective roles and responsibilities and to identify points where conflicts of interests may arise and where there are tradeoffs to be made between partly non-compatible goals and objectives.

Another important key to a successful sustainability framework is to provide the involved stakeholders with guidance on how to produce SRC bioenergy in compliance with the RED-associated criteria. In addition, there is a need for tools to provide verification and continuous monitoring of the RED eligibility. This is addressed in the following chapter.

Potentially Useful Tools for National SRC Bioenergy Sustainability Schemes

Producer manuals, environmental impact assessments (EIAs), and certification schemes can all provide guidance as well as contribute to the monitoring and verification of sustainable

[2] See also Englund et al. [17] for more information.

Table 1 RED sustainability categories and associated sustainability criteria of national relevance for SRC bioenergy production

RED categories	Associated sustainability criteria	Current status
Biodiversity	1.1 Preservation of natural forests	Existing requirement
	1.2 Preservation of areas designated for nature protection purposes or for the protection of rare, threatened, and endangered species	Existing requirement
	1.3 Preservation of highly biodiverse grasslands	Existing requirement
	1.4 Impacts on biodiversity	MS reporting obligation
GHG emissions	2.1 Preservation of peatlands	Existing requirement
	2.2 GHG emissions from extraction or cultivation of raw materials	GHG emissions savings calculation
	2.3 GHG emissions from processing	GHG emissions savings calculation
	2.4 GHG emissions from transport and distribution	GHG emissions savings calculation
	2.5 Carbon capture and replacement	GHG emissions savings calculation
	2.6 Co-generation of electricity, if producing bioliquids	GHG emissions savings calculation
Carbon stock	3.1 Preservation of wetlands	Existing requirement
	3.2 Preservation of continuously forested areas	Existing requirement
	3.3 Restoration of degraded land	GHG emissions savings calculation
	3.4 Restoration of contaminated land	GHG emissions savings calculation
Air, water and soil	4.1 Impacts on air quality	MS reporting obligation/prospective requirement
	4.2 Impacts on water quality	MS reporting obligation/prospective requirement
	4.3 Impacts on water availability	MS reporting obligation/prospective requirement
	4.4 Impacts on soil quality	MS reporting obligation/prospective requirement

biomass production. In order to determine whether these tools, individually or combined, can be useful for ensuring that SRC bioenergy is produced with sufficient consideration given to the RED-associated criteria, they have been assessed on their coverage in relation to the criteria in Table 1.

Producer Manuals

The purpose of producer manuals is to provide guidelines to support good management practices. In the case of SRC, manuals typically cover the feedstock production phase but can also cover other parts of the supply chain, e.g., transportation of harvested biomass, processing, etc. Ten producer manuals were assessed on their coverage of RED-associated criteria (Table 3). All manuals refer to willow and/or poplar coppice production, including site selection, planting, and harvesting. Most of the manuals also consider transportation and handling/storage of harvested material, but only three manuals cover the processing of biomass for bioenergy (i.e., heat, electricity, and bioliquids).

The general usefulness of producer manuals for ensuring that SRC bioenergy is produced with sufficient consideration given to the RED is assumed to coincide with their overall coverage of the specific RED-associated criteria. Given this connection, Table 3 can be interpreted as follows: Manuals are *likely to be useful* for ensuring avoidance of SRC production on peatlands and wetlands, as well as impacts on soil quality. Manuals are *potentially useful* for ensuring that impacts on air and water quality, water availability, and biodiversity, are avoided/acceptably low. They are also *potentially useful* for considering GHG emissions from cultivation, extraction, transport and distribution, and for avoiding SRC production on protected areas, i.e., areas designated for nature protection purposes or for the protection of rare, threatened, and endangered species. Manuals are however *unlikely to be useful* for ensuring that the remaining eight RED-associated criteria (i.e.,

Table 2 Principal stakeholders involved in SRC bioenergy supply chains

Stakeholders	Interpretation
Landowner	Farmer producing SRC
Entrepreneur	Responsible for planting/harvesting/transport (can be several entrepreneurs)
Bioenergy producer	Producer of electricity/heat/biofuels
End user	Consumer of electricity/heat/biofuels
Administrator	Municipality, county administrative board
Legislator	National government, European Commission, other bodies involved in developing sustainability frameworks (e.g., certification systems)

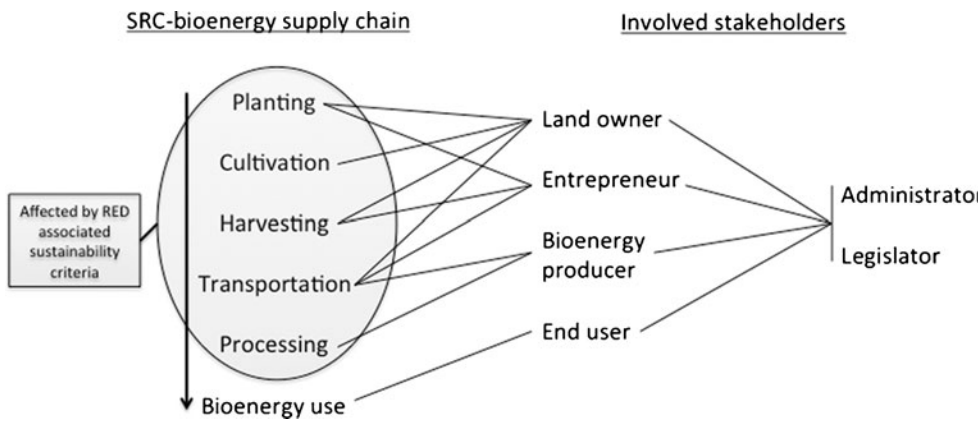

Fig. 1 A typical SRC bioenergy supply chain, with indication of involvement of principal stakeholders in the different supply chain segments

criteria 1.1, 1.3, 2.3, 2.5, 2.6, 3.2, 3.3, 3.4) are sufficiently considered.

The fact that the assessed manuals poorly cover eight of the 18 RED-associated criteria does not mean that SRC producers will automatically disregard these. For example: even though it is seldom discussed in manuals, SRC producers are less likely to convert forests into SRC plantations. Given that manuals typically describe only how to prepare existing cropland or grassland for SRC planting, it can be implicit that recently deforested land is unsuitable for SRC establishments. Even so, to make sure that the RED sustainability requirement of *no deforestation* is complied with, producer manuals should be complemented with the requirement that forested areas should not be converted to SRC plantations. This reasoning (i.e., not to exclude seemingly unnecessary information) should be applied when it comes to all RED-associated criteria, in order to safeguard that they are all considered.

The assessment shows that existing producer manuals are not sufficient for ensuring that SRC bioenergy is produced with sufficient consideration given to the RED-associated criteria. However, as manuals are typically consulted before starting new establishments and as they are commonly known and widely used by SRC and bioenergy producers, they could potentially be further developed to fulfill this purpose. It should however be taken into account that producers might not fully follow manuals. In such a case, it is of little use that the manual itself is perfectly comprehensive; the biomass might still not be RED eligible. Therefore, it should be clear which information in the manual refers to good management practice in general and which are connected to RED-associated criteria, so that the producers understand the consequences when deviating from the different advices given.

Even if SRC bioenergy is produced in compliance with future sustainability requirements, it would not automatically make it RED eligible, as compliance with the requirements has to be demonstrated. Therefore, advice on how the producers should monitor their activities in order to demonstrate compliance should also be provided in the manuals. The assessed manuals typically do not include such advice.

Environmental Impact Assessments

An EIA can be defined as “the process of identifying, predicting, evaluating, and mitigating biophysical, social, and other relevant effects of development proposals prior to major decisions being taken and commitments made” [19, 20]. Thus, the main purpose of an EIA is to help incorporate environmental considerations in decision making. This is achieved primarily by assembling and analyzing information, identifying potential environmental impacts from specific development proposals, and proposing measures to avoid or mitigate these impacts. An EIA for a proposed project should be conducted before major decisions are taken. In this sense, environmental considerations in an EIA can influence the whole decision-making process, from initial contemplation of a project to actual implementation [21].

The concept of “environment” in EIA originates from the initial focus on the biophysical environment, but has over time often been extended to include also physical–chemical, biological, visual, cultural, and socio-economic components of the total environment [20]. EIA systems may therefore use different definitions of the concept “environment”, including biophysical aspects only or also social, economic, and institutional aspects.

Typically, an EIA is made at a stage where projects are subject to consideration by authorities, but the EIA may also be used in earlier feasibility studies to guide decisions about how to proceed with a certain project idea. One example can be when stakeholders want to investigate whether a planned bioenergy project will have prospects for targeting the RED market [17]. Thus, besides serving a legal and institutional procedure, EIAs can also help stakeholders avoid or mitigate impacts from planned actions or unplanned events, e.g., natural disasters [20]. Specifically for the EU RED market, an advantage of using EIAs is that it can provide information needed to demonstrate that the produced biomass is RED eligible [17].

Table 4 shows the usefulness of 19 EIAs for this purpose (see also [17]). These EIAs relate to plantation projects or biofuel projects that either include both biofuel production

Administrative authorities involved in SRC production in Sweden

Profitable SRC production is currently taking place almost entirely on farmland and is therefore under the sectorial responsibility of the **Board of Agriculture** when it comes to rules for agricultural management, establishment support etc. Being a potential energy crop, research and other activities advancing SRC production for energy are supported by the **Energy Agency**.. Production of SRC on farmland affects the environmental quality goals, primarily governed by the **Environmental Protection Agency**. Production of perennial crops affects the landscape and the cultural environment, for which the **National Heritage Board** has the main responsibility. Even though SRC production on farmland is not directly the responsibility of the **Forest Agency**, the products (i.e. forest fuels) are. The Forest Agency (and other actors) offers the establishment of a *green plan* for forest holdings, which can be used as a basis for **FSC** and **PEFC** certification, although this system is currently only relevant for forest management and not SRC production on farmland. From the perspective of business administration, as well as recycling considerations it can be beneficial to use multifunctional production systems, for instance where it is possible to take advantage of the effective uptake of nutrients and heavy metals in fast growing trees. If SRC production is used for restoring contaminated land or the disposal of sewage sludge, the **Board of Housing, Building and Planning**, the **Chemical Agency,** the **Board of Health and Welfare** and the **Institute for Communicable Disease Control** all become involved. Finally there are aspects of e.g. nature, culture, agriculture and healthcare that require that **County Administrative Boards** and **Municipalities** become involved.

Thus, a large number of authorities at national, county and municipal levels have influence on SRC production. The coordination between different authorities' interests is poor. In addition, questions regarding the use of sewage sludge and ash in SRC production are sometimes circulated within authorities without anyone taking a clear overall responsibility. This can seriously hamper the process and there is also a risk of power struggles between authorities. In such situations, the holistic perspective gets lost and synergies between objectives are not realized.

Fig. 2 Fact box. Administrative authorities involved in SRC production in Sweden [18]

plants and plantations for feedstock supply or only the biofuel production plants. None of the 19 EIAs relate to SRC projects in EU, since EIAs are typically not required for SRC production.

As seen in Table 4, there are in many cases large variations in coverage between the individual EIA reports. Of the 18 RED-associated criteria, nine were considered in a sufficiently similar way in the EIAs to allow the general coverage to be estimated with an adequate accuracy. Of these nine criteria, five were typically well covered by the EIAs (impacts on biodiversity, impacts on air quality, impacts on water quality, impacts on water availability, and impacts on soil quality), while four were typically poorly covered (GHG emissions from extraction or cultivation of raw materials, GHG emissions from transport and distribution, restoration of degraded land and restoration of contaminated land).

Notable differences can be seen also between the project categories; EIAs for biofuel projects that include plantations for the feedstock supply had better coverage than EIAs for only plantations (that naturally do not consider criteria related to post harvest activities) or for biofuel projects that import their feedstock from external sources and therefore do not consider the feedstock production phase. This indicates that the entire supply-chain needs to be considered in EIA in order to ensure full coverage of the RED-associated criteria and consequently RED eligibility.

Only five RED-associated criteria are "highly covered" by the assessed EIAs, indicating that a typical EIA does not suffice for ensuring RED eligibility of a planned bioenergy project. However, *if* EIAs were extended to sufficiently consider all criteria, it should be possible to use it for assessing RED eligibility. One problem can be that such

Table 3 Coverage of 10 producer manuals in relation to the RED-associated criteria

RED category	Associated sustainability criteria	Coverage of producer manuals		
		Minor coverage	Major coverage	Overall coverage[a]
Biodiversity	1.1 Preservation of natural forests	0	3	–
	1.2 Preservation of areas designated for nature protection purposes or for the protection of rare, threatened, and endangered species	2	3	+/–
	1.3 Preservation of highly biodiverse grasslands	0	2	–
	1.4 Impacts on biodiversity	3	4	+/–
GHG emissions	2.1 Preservation of peatlands	6	3	+
	2.2 GHG emissions from extraction or cultivation of raw materials	7	2	+/–
	2.3 GHG emissions from processing	4	0	–
	2.4 GHG emissions from transport and distribution	3	4	+/–
	2.5 Carbon capture and replacement	0	0	–
	2.6 Co-generation of electricity, if producing bioliquids	1	0	–
Carbon stock	3.1 Preservation of wetlands	3	6	+
	3.2 Preservation of continuously forested areas	1	2	–
	3.3 Restoration of degraded land	0	0	–
	3.4 Restoration of contaminated land	3	0	–
Air, water and soil	4.1 Impacts on air quality	1	3	+/–
	4.2 Impacts on water quality	6	2	+/–
	4.3 Impacts on water availability	4	2	+/–
	4.4 Impacts on soil quality	4	6	+

[a] Coverage index (0–5) calculated with: (minor coverage × 0.5 + major coverage)

Interpretation: 0–1.5 = low coverage (−), 1.6–2.9 = varying coverage (+/−), 3–5 = high coverage (+)

EIAs may be too costly for smaller projects such as farm level SRC production. Thus, it would be required that EIAs are streamlined to become less time consuming and expensive.

Voluntary Certification Schemes

Several schemes exist for certification of sustainable production of biomass or biomass products, including bioenergy. Certification schemes require producers to comply with a set of sustainability criteria in order to become certified. In addition, certifiers require that producers monitor and document their operations to allow demonstration of compliance with the criteria. In cases where a product is certified (e.g., bioenergy), the full supply chain typically needs to be considered. Often, guidelines are provided in order to help producers to adjust their operations.

The second subparagraph of Article 18(4) in the RED [8] states that:

> "The Commission may decide that voluntary national or international schemes setting standards for the production of biomass products contain accurate data for the purposes of Article 17(2)[3] or demonstrate that consignments of biofuel comply with the sustainability criteria set out in Article 17(3) to (5).[4] The Commission may decide that those schemes contain accurate data for the purposes of information on measures taken for the conservation of areas that provide, in critical situations, basic ecosystem services (such as watershed protection and erosion control), for soil, water, and air protection,[5] the restoration of degraded land,[6] the avoidance of excessive water consumption in areas where water is scarce[7] and on the issues referred to in the second subparagraph of Article 17(7)."

The above citation from the RED refers to 22 of the 28 RED-associated criteria in [17] and to 16 of the 18 criteria identified in this paper as relevant for national sustainability frameworks (see footnotes 3–7). The Commission has thus acknowledged that certification schemes can play a role in

[3] Refers to RED criteria 2.2–6

[4] Refers to RED criteria 1.1–3, 2.1, 3.1–2

[5] Refers to RED criteria 4.1–2, 4.4

[6] Refers to RED criterion 3.3

[7] Refers to RED criterion 4.3

Table 4 Coverage of 19 EIAs for bioenergy projects in relation to the RED-associated criteria

RED category	Associated sustainability criteria	Coverage of EIAs [a,b]			
		Plantations	Biofuel plant	Plantations and biofuel plant	Overall coverage
Biodiversity	1.1 Preservation of natural forests	+	−	+	+/−
	1.2 Preservation of areas designated for nature protection purposes or for the protection of rare, threatened, and endangered species	+/−	+/−	+/−	+/−
	1.3 Preservation of highly biodiverse grasslands	+/−	+/−	+/−	+/−
	1.4 Impacts on biodiversity	+	+/−	+	+
GHG emissions	2.1 Preservation of peatlands	+/−	−	+/−	+/−
	2.2 GHG emissions from extraction or cultivation of raw materials	−	−	+/−	−
	2.3 GHG emissions from processing	−	+/−	+/−	+/−
	2.4 GHG emissions from transport and distribution	−	−	−	−
	2.5 Carbon capture and replacement	−	+/−	+/−	+/−
	2.6 Co-generation of electricity, if producing bioliquids	−	−	+	+/−
Carbon stock	3.1 Preservation of wetlands	+/−	−	+/−	+/−
	3.2 Preservation of continuously forested areas	+/−	−	+	+/−
	3.3 Restoration of degraded land	−	−	+/−	−
	3.4 Restoration of contaminated land	−	−	−	−
Air, water and soil	4.1 Impacts on air quality	+/−	+	+	+
	4.2 Impacts on water quality	+	+	+	+
	4.3 Impacts on water availability	+	+/−	+	+
	4.4 Impacts on soil quality	+	+/−	+	+

[a] Interpretation: Overall low coverage (−), varying coverage (+/−), overall high coverage (+).

[b] See also [17].

verifying that biofuel projects comply with existing and also possible future RED requirements. The EC has to date approved seven schemes for the purpose of verifying that bioliquids are produced in compliance with the existing RED requirements [22] (Table 5). Additional schemes are likely to be added as the benchmarking continues and rejected schemes reapply with revised standards.

Four out of seven RED-approved certification schemes can be relevant for SRC production; ISCC, RSB, 2BSvs, and RBSA. These schemes were assessed on their coverage in relation to the *existing* RED sustainability requirements only (all schemes also provide for calculation of GHG emissions savings). The extent to which they cover the RED-associated criteria related to reporting obligations has not been investigated. It should be noted that these schemes mainly focus on the production of liquid biofuels

The approved certification schemes represent an option for ensuring RED eligibility. However, as the approved schemes only have been proven to sufficiently cover the existing RED requirements, certification by an approved scheme may not ensure future RED eligibility. Other schemes may cover the RED-associated criteria equally well or better, although being better suited for SRC production in general or for specific local conditions. Information about which certification schemes have applied for, or shown an interest in, RED approval is unfortunately not available at present [23, 24]. It is therefore advised that separate assessments are initiated in parallel to the EC benchmark process, to clarify which specific schemes are best suited for verifying RED eligibility of SRC bioenergy production. Such assessments should include, but not be limited to, the RED-approved schemes and carefully monitor new outcomes from the EC benchmark process.

Examples of voluntary certification schemes potentially relevant for the entire, or parts of, the SRC bioenergy supply chain are presented in Table 6. These may or may not apply for RED approval and are likely to have varying coverage of the RED-associated criteria.

Tables 5 and 6 may give the impression that SRC bioenergy stakeholders interested in certification have a variety of options. However, in several cases it is uncertain if the schemes are suitable for, or even accept, SRC production. Even though SRC cultivation clearly has more in common with conventional agriculture than forest management, it is not clear whether schemes for certification of sustainable agricultural management (e.g., EU organic farming,

Table 5 Voluntary certification schemes approved by the EC for verifying that biofuels and –feedstock is produced in compliance with RED sustainability requirements

Certification scheme		Geographical coverage	Relevant for SRC
International Sustainability and Carbon Certification (ISCC)	www.iscc-system.org	Global	Yes, covers all types of feedstock
Bonsucro EU production standard	www.bonsucro.com	Global	No, only sugarcane
Roundtable on Responsible Soy (RTRS) (EU RED standard)	www.responsiblesoy.org	Global	No, only soybean
Roundtable on Sustainable Biofuels (RSB)	http://rsb.epfl.ch	Global	Yes, covers all types of feedstock
Biomass Biofuels voluntary scheme (2BSvs)	http://en.2bsvs.org	Global	Yes, covers all types of feedstock
Abengoa RED Bioenergy Sustainability Assurance (RBSA)	www.abengoa.com/corp/web/en	Global	Yes, covers all types of feedstock
Greenergy Brazilian Bioethanol verification programme	www.greenergy.com	Brazil	No, only sugarcane

GLOBALGAP, etc.) will accept to certify SRC; in fact, no such examples have been found. However, certification schemes for sustainable *forest* management have been identified as potentially accepting SRC plantations [25]. The ambiguity of SRC production (i.e., trees are cultivated with management practices similar to conventional agriculture) brings difficulties in evaluating whether or not a certification scheme is relevant for SRC or not, unless it is specified in the certification standards. Also, since some certification schemes (e.g., FSC and PEFC) have national variants of their certification standards, their relevance for SRC is likely to differ between countries. Therefore, national sustainability frameworks for SRC need to be designed so that the stakeholders can judge what certification options are available. As a consequence of nationally differing certification standards, it is difficult to provide a useful internationally valid assessment of the coverage of certain certification schemes in relation to the RED-associated criteria. Therefore, such assessments also need to be done on a country level within the SRC sustainability framework.

Conclusions and Discussion

Eighteen sustainability criteria associated to EU RED have been identified as relevant for stakeholders involved in SRC bioenergy (Table 1). These are related to (1) existing and prospective legally binding sustainability requirements, (2) reporting obligations for MS, and (3) the methodology for calculating GHG emissions savings. Even though specific stakeholders can be officially responsible for *demonstrating* compliance with certain RED-associated criteria, other

Table 6 Examples of voluntary certification schemes potentially relevant for the entire, or parts of, the SRC bioenergy supply chain

Certification scheme		Type of feedstock	Coverage
Forest Stewardship Council (FSC)	www.fsc.org	Forest based	Biomass production, global
Programme for the Endorsement of Forest Certification (PEFC)	www.pefc.org	Forest based	Biomass production, global
REDcert	www.redcert.org/index.php?lang=en	Not defined	Bioliquids, Germany (also other European countries)
NTA 8080	www.sustainable-biomass.org	Forest and agriculture based	Bioliquids/heat/ electricity, global
International Organization for Standardization (ISO)	www.iso.org/iso/home.htm (standard under development)	Forest and agriculture based	Bioliquids/heat/ electricity, global
Green Gold Label (GGL)	www.greengoldcertified.org/site/pagina.php?	Forest and agriculture based	Bioliquids/heat/ electricity, global
EKOenergy	www.ekoenergy.org	Forest and agriculture based	Heat/electricity, Finland (also Sweden, Norway, and Denmark)
Bra Miljöval	www.naturskyddsforeningen.se/bra-miljoval	Forest and agriculture based	Heat/electricity, Sweden (also other countries)
Green-e	www.green-e.org	Forest and agriculture based	Electricity, USA

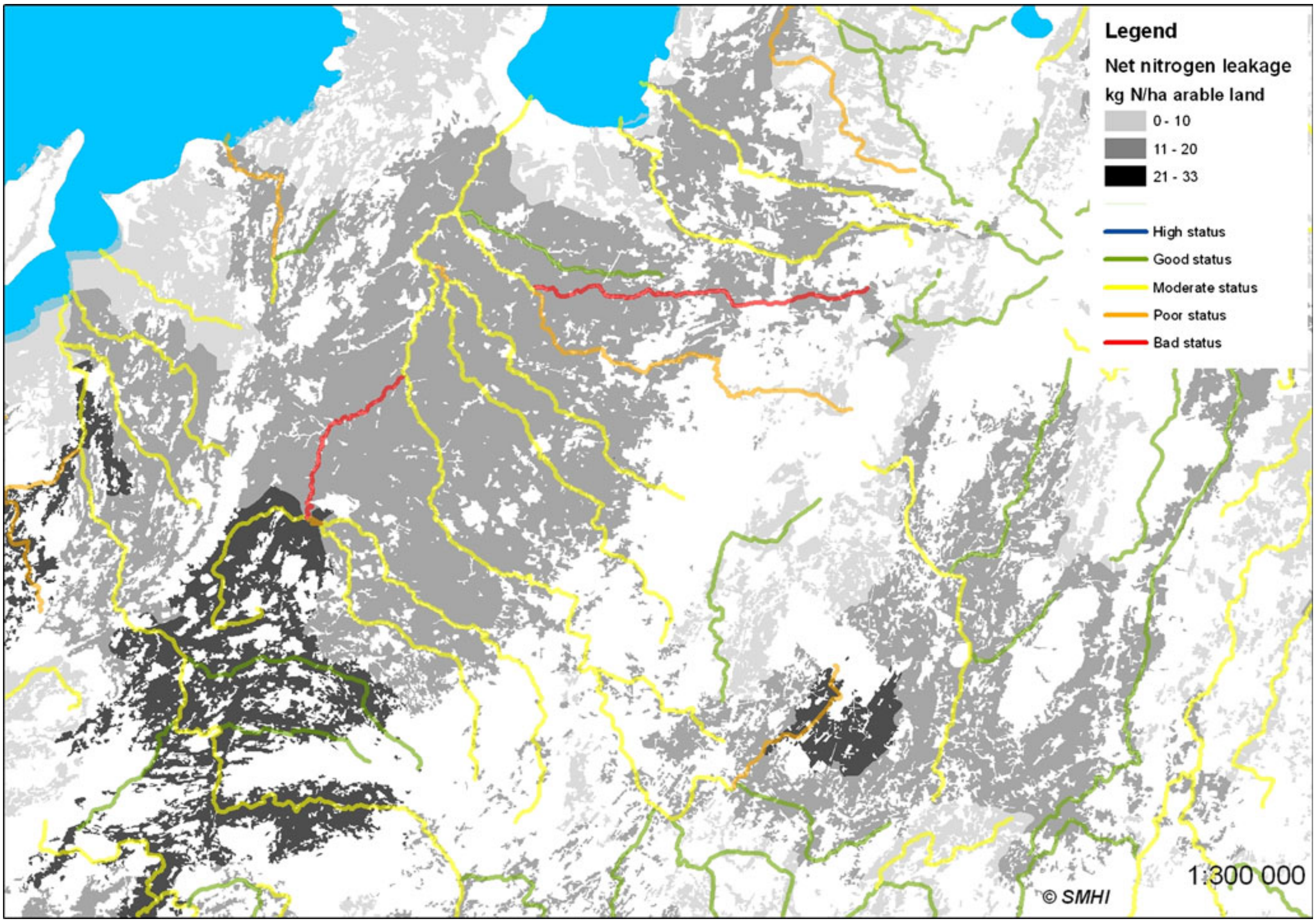

Fig. 3 Combined map of (1) water status in rivers and (2) net nitrogen leakage from arable land, in the Västra Götaland region, Sweden

stakeholders can have a responsibility in *ensuring* compliance. Given the varying structure of SRC bioenergy supply chains (Fig. 1), it is difficult to suggest stakeholder-specific responsibilities, implying that the RED-associated criteria should be considered by all stakeholders along SRC bioenergy supply chains. In addition, effective consideration of some criteria may require interactions between several stakeholders depending on their respective involvement along the SRC bioenergy supply chain. Providing opportunities for such interactions is important but may be challenging, as experiences from Sweden show that coordination between different stakeholders involved with SRC can be poor [18] (see Fig. 2, fact box). It is important that a sustainability framework is designed so as to facilitate stakeholder interaction to clarify the stakeholders' respective roles and responsibilities and to identify points where conflicts of interests may arise and where there are tradeoffs to be made between partly non-compatible goals and objectives. Proper consideration of all relevant aspects therefore requires all stakeholders in the SRC supply chain to be engaged in the development of SRC production systems and that a landscape perspective is used.

Producer manuals, EIAs, and voluntary certification schemes can all be useful for ensuring that SRC bioenergy is produced with sufficient consideration given to the RED-associated criteria. However, they currently do not suffice for this purpose, either individually or combined. *Producer manuals* need to be complemented to sufficiently cover the RED-associated criteria (Table 3), and advice on how producers should monitor their activities in order to demonstrate compliance should be provided. *EIAs* also need to be extended to sufficiently consider all criteria (Table 4), but they also need to be streamlined to become less time consuming and expensive. Regarding voluntary certification schemes, national sustainability frameworks for SRC need to be designed so that the producing stakeholders are well informed about the availability and relevance of certification options, which in most cases is likely to vary between countries. The coverage of certain certification schemes in relation to the RED-associated criteria also needs to be assessed on a country level, although continuously considering outcomes from the EC benchmarking process.

Thus, a sustainability framework for SRC bioenergy can include several components. Most importantly though—a

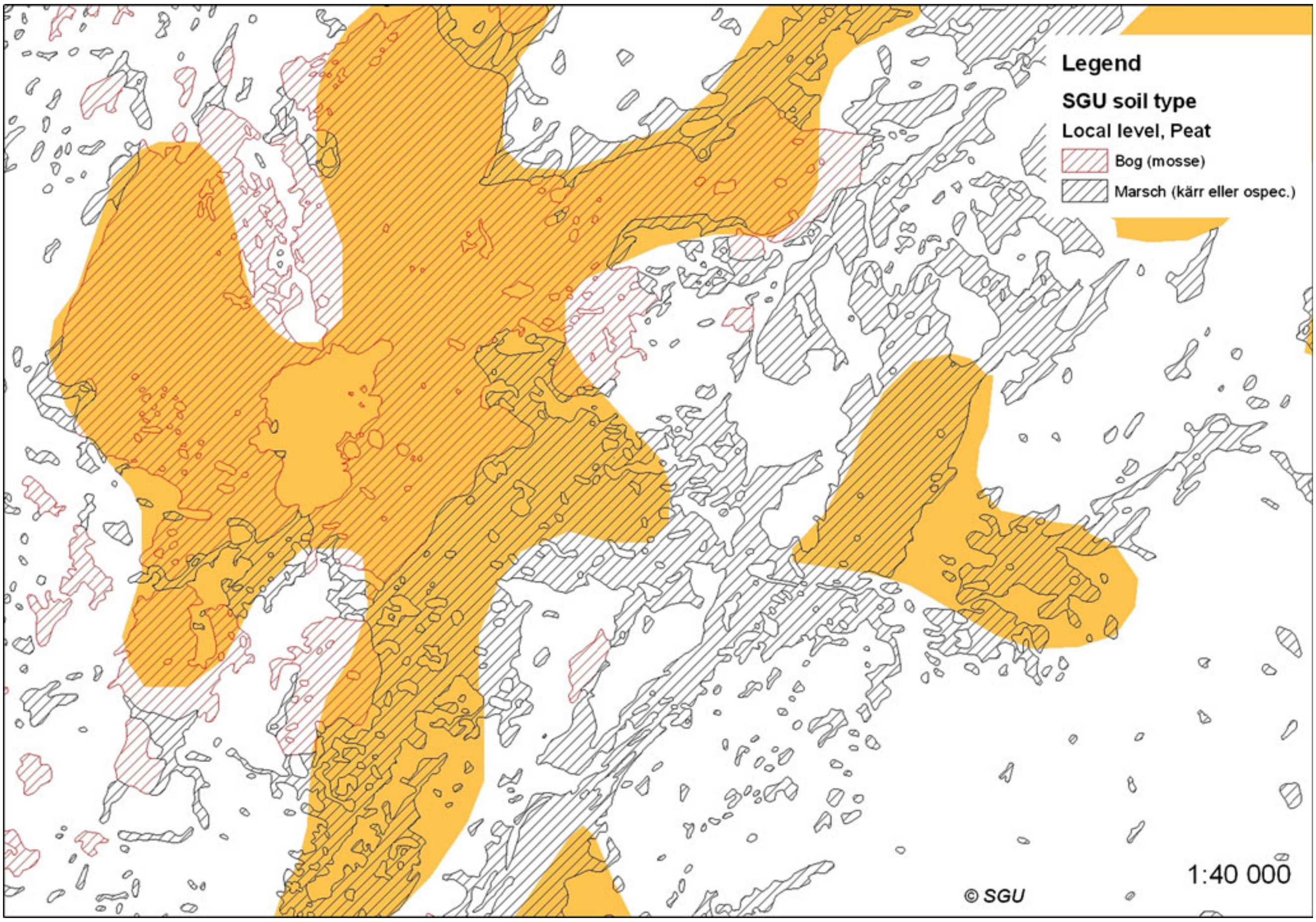

Fig. 4 Overlay of (1) a national soil class layer (peat) with low resolution and (2) a local soil class layer (peat) with high resolution, in an area close to Hornborgarsjön in the Västra Götaland region, Sweden. The *homogenous yellow area* shows peatlands according to (1), the *area with red stripes* shows bogs according to (2) and the *area with black stripes* shows marches according to (2).

sustainability framework needs to provide landscape level processes and engage all involved stakeholders. An appropriate institution should take a formal role in coordination, to ensure that developments are progressing in line with the interests of all stakeholders. From a Swedish perspective, county administrative boards may be best suited for this role since they are already involved in regulating SRC bioenergy stakeholders in different ways. In other countries, similar multi-sectoral administrative authorities involved in planning and governing rural development issues could be appropriate.

Multi-stakeholder, landscape level processes should include initiatives that allow a wide range of stakeholders to engage in dialog on collective issues. In Sweden, a recent initiative, "Salixdagen" (the Salix day), gathered several important stakeholder groups to discuss the potential of SRC in Sweden. Such initiatives should be realized also on sub-national levels. "Roundtable" sustainability certification initiatives, such as Roundtable on Sustainable Palm Oil and national FSC meetings, are other good examples that can be learnt from.

A Way Forward: Integrated Assessments of Landscape Level and Site-Specific Aspects

Consideration of values linked to biodiversity and cultural heritage, as well as esthetic and other landscape values, requires landscape level analyses. Other more site-specific aspects, such as soil quality, can be treated using suitable indicators. The use of geographic information systems (GIS) can facilitate an integrated assessment of both landscape level aspects and the more site-specific aspects. This is shown in Fig. 3 where a combined map of nitrogen leakage in cropland and water status in rivers in the Västra Götaland region of Sweden has been produced using GIS technology. Producing maps similar to Fig. 3 may be an appropriate strategy for identifying areas where SRC can be cultivated in compliance with the RED-associated criteria that restrict conversion of certain types of ecosystems/areas. Such maps can later be used by stakeholders in the SRC bioenergy supply chain for proving RED eligibility. In

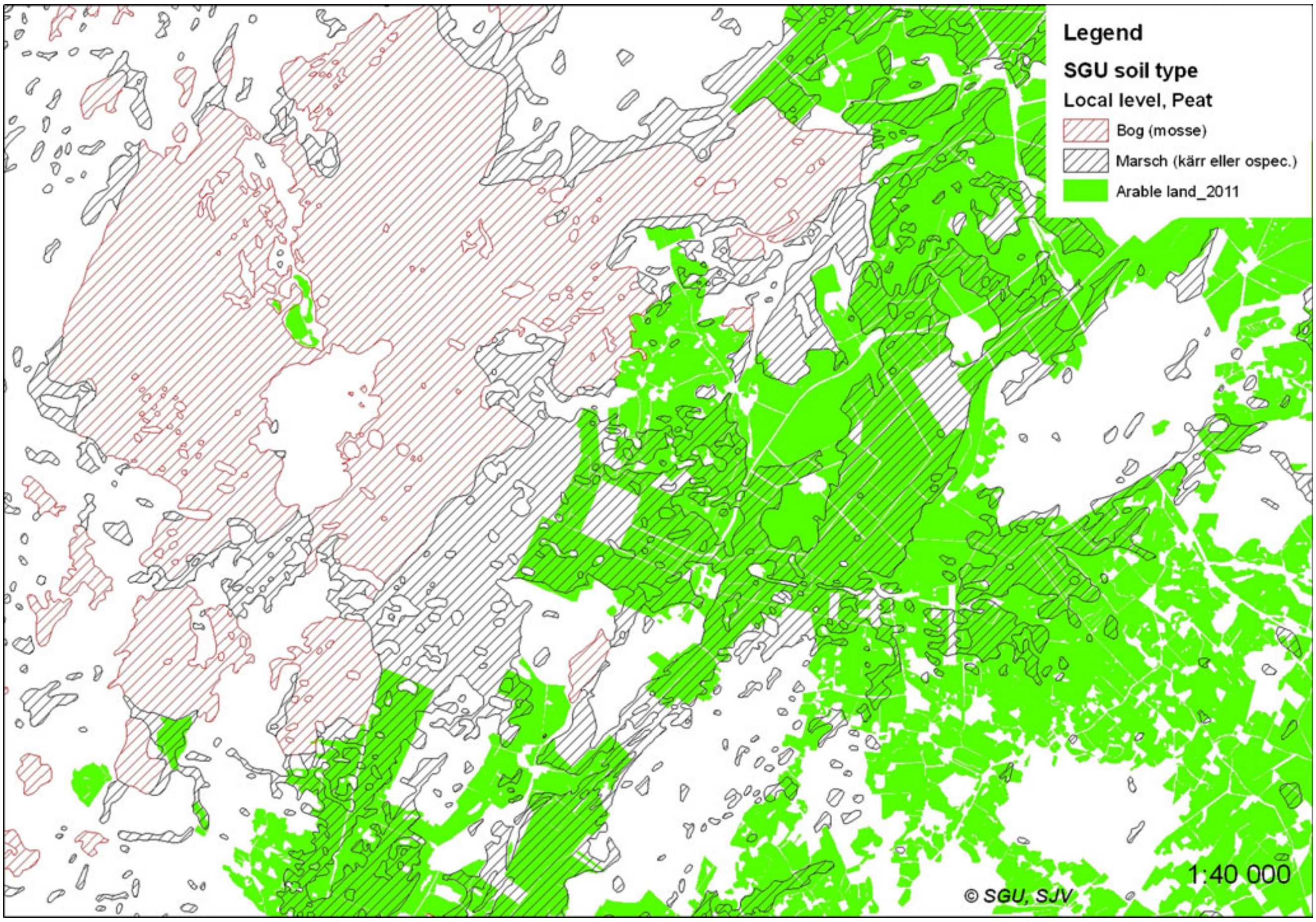

Fig. 5 Overlay of (1) a local-level soil class layer (peat) with high resolution and (2) a local-level land-cover layer (arable land) with high resolution, in an area close to Hornborgarsjön in the Västra Götaland region, Sweden (same area as in Fig. 4)

Sweden, each field of agricultural land has a unique identity,[8] which theoretically makes it possible to create a database of fields suitable for SRC cultivation with sufficient consideration given to the RED-associated criteria.

Since the baseline year in the RED is 2008,[9] datasets from 2008 or a few years earlier are required for GIS technology to be useful for the purpose described above. It is also important that definitions used in GIS datasets are comparable to definitions laid out in the RED. For example, "continuously forested areas" may be defined differently in a GIS dataset than in the RED, which is likely to cause difficulties if maps based on such datasets are to be used for proving compliance with RED sustainability requirements. It is also important to use datasets with sufficiently high resolution. If a particular field is investigated, a map based on a dataset with low resolution may not be sufficiently detailed. This is particularly the case where areas "protected" by legally binding RED requirements exist adjacent to an assessed field. This is shown in Fig. 4 where two different soil-type layers are shown, a national layer with low resolution and a local layer with high resolution. It is clear that the two layers do not entirely match. In this case, the national layer may not have a sufficiently high resolution for the map to prove that a particular field has been established on lands other than peatland. In Fig. 5, the local soil-type layer is shown with a layer of arable land (i.e., existing cropland). These layers have a similar resolution and the map may therefore be possible to use for identifying fields located on peat soils, where SRC cultivation should be avoided.

GIS can also be used for supporting the location, design, and management of SRC plantations to produce various environmental services, e.g., reduce nutrient leaching and prevent eutrophication [12, 26, 27], cadmium removal [12, 28], and promoting biodiversity [14, 15]. Such environmental services may not be explicitly relevant for the RED eligibility of SRC bioenergy, but can nevertheless be important to

[8] Other EU countries use similar systems.

[9] The status of a particular area in 2008 (e.g., natural forest, wetland etc.) is assessed when the RED-eligibility of a bioenergy project is determined.

consider when assessing the overall environmental performance of different production systems on a landscape level.

GIS technology may also be a useful tool for developing regional producer manuals, conducting EIAs[10], or demonstrating compliance to certification standards. It has however been scarcely used for such purposes in the past, much due to the needs of high-resolution datasets, which not always exist, and competent human capital, which can be too costly in case of smaller projects with limited financial capital. A centralized mapping of SRC suitability, as discussed above, may help to mitigate these constraints and thus make GIS more applicable also for these purposes.

Thus, by using GIS technology, *administrators* may be able to provide other stakeholders in the SRC bioenergy supply chain (particularly *landowners*, *entrepreneurs*, and *bioenergy producers*) with maps or databases over areas/fields suitable for SRC cultivation, with sufficient consideration given to the RED-associated criteria (see Fig. 5). For example, by combining datasets on soil and land-cover classes, maps, or field databases of no-go areas for SRC production in relation to the required preservation of peatlands and certain ecosystems, can be created—provided that regularly updated datasets of high accuracy and resolution exist and that definitions of land-cover or soil-type classes are comparable to the definitions laid out in the RED. Given that administrators typically regulate the producing stakeholders in different ways, it should also be possible for them to require that such maps or databases are consulted prior to the initiation of new SRC projects.

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Assessing Environmental Impacts of Short Rotation Coppice (SRC) Expansion: Model Definition and Preliminary Results

Hans Langeveld · Foluke Quist-Wessel · Ioannis Dimitriou · Pär Aronsson · Christel Baum · Ulrich Schulz · Andreas Bolte · Sarah Baum · Jörg Köhn · Martin Weih · Holger Gruss · Peter Leinweber · Norbert Lamersdorf · Paul Schmidt-Walter · Göran Berndes

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Abstract Short rotation coppice (SRC) systems can play a role as feedstock for bioenergy supply contributing to EU energy and climate policy targets. A scenario depicting intensive arable crop cultivation in a homogeneous landscape (lacking habitat structures) was compared to a scenario including SRC cultivation on 20 % of arable land. A range of indicators was selected to assess the consequences of SRC on soil, water and biodiversity, using data from the Rating-SRC project (Sweden and Germany). The results of the assessment were presented using spider diagrams. Establishment and use of SRC for bioenergy has both positive and negative effects. The former include increased carbon sequestration and reduced GHG emissions as well as reduced soil erosion, groundwater nitrate and surface runoff. SRC can be used in phytoremediation and improves plant and breeding bird biodiversity (exceptions: grassland and arable land species) but should not be applied in dry areas or on soils high in toxic trace elements (exception: cadmium). The scenario-based analysis was found useful for studying the consequences of SRC cultivation at larger scales. Limitations of the approach are related to data requirements and compatibility and its restricted ability to cover spatial diversity and dynamic processes. The findings should not be generalised beyond the representativeness of the data used.

H. Langeveld (✉) · F. Quist-Wessel
Biomass Research,
P.O. Box 247, 6700 AE Wageningen, The Netherlands
e-mail: hans@biomassresearch.eu

I. Dimitriou · P. Aronsson · M. Weih
Department of Crop Production Ecology,
Swedish University of Agricultural Sciences (SLU),
P.O. Box 7043, 750 07 Uppsala, Sweden

C. Baum · P. Leinweber
Chair of Soil Science, Faculty of Agricultural and Environmental Sciences, University of Rostock,
18051 Rostock, Germany

U. Schulz · H. Gruss
Faculty of Landscape Management and Nature Conservation,
University of Applied Sciences Eberswalde (HNEE),
Friedrich-Ebert-Straße 28,
16225 Eberswalde, Germany

A. Bolte · S. Baum
Institute for Forest Ecology and Forest Inventory,
Johann Heinrich von Thünen-Institute (vTI),
Alfred-Möller-Straße 1,
16225 Eberswalde, Germany

A. Bolte · S. Baum
Department of Silviculture and Forest Ecology of Temperate Zones, Georg-August-University Göttingen,
Büsgenweg 1,
37077 Göttingen, Germany

J. Köhn
Beckmann-Institute for Bio-Based Product Lines (BIOP),
Büdnerreihe 20a,
18239 Heiligenhagen, Germany

N. Lamersdorf · P. Schmidt-Walter
Soil Science of Temperate Ecosystems, Büsgen-Institute,
Georg-August-Universität Göttingen,
Büsgenweg 2,
D-37077 Göttingen, Germany

G. Berndes
Department of Energy and Environment,
Chalmers University of Technology,
Göteborg, Sweden

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Introduction

Ambitions for development of bioenergy as a means to improve energy security and reduce greenhouse gas (GHG) emissions in Europe are high. Most EU member states (MS) assign a crucial role to bioenergy in their strategies for meeting the EU Renewable Energy Directive (RED), which mandates levels of renewable energy use and also includes a sustainability scheme for liquid biofuels [1]. Countries that have considerable forest resources and a mature forest industry (e.g., Sweden and Finland) can obtain substantial amounts of biomass feedstock from their forest resources. In most European countries, however, forest biomass flows are relatively small compared to the biomass supply needed for reaching RED and longer-term targets [2]. Alternative sources are therefore needed and biomass imports, utilization of waste and residues and the cultivation of bioenergy feedstock are all considered as options—with varying importance depending on specific conditions in the different countries. In agriculture, bioenergy feedstock cultivation presently focuses on conventional food/feed crops that are used for the production of biofuels for transport and, to a limited extent, also for stationary energy purposes.

So far, the production of lignocellulosic biomass in European agriculture mainly consists of fibre crops for non-energy purposes and residue harvest in conventional food/feed crop production, straw being the most common residue. Crop residues are mainly collected for non-energy purposes such as animal feeding and bedding but in a few MS also for electricity and heat. Willow has been grown commercially for electricity and heat production in Sweden since the beginning of the 1990s, and the plantations presently amount to some 14,000 ha, about 0.5 % of the Swedish arable land. Thus, despite 20 years of cultivation experience, willow production is still an emerging agricultural activity with a small land claim [3]. Cultivation of lignocellulosic crops for energy in other MS is also limited, e.g., in Italy (about 6,000 ha, mostly poplar [4]), Poland (about 3,000 ha, mostly willow [4]), the UK (about 7,500 ha, mostly willow [5]) and Germany (about 5,000 ha, poplar and willow [6]).

Notwithstanding their slow start, willow and other short rotation coppice (SRC) plants are considered potentially important future sources of lignocellulosic biomass in Europe for heat and electricity generation—and for the production of so-called second generation biofuels when these technologies become commercially available. Studies indicate good prospects for production cost reductions and cost competitiveness relative to other renewable options and also point to a substantial potential for lignocellulosic crop cultivation on agricultural areas in the EU (see, e.g. [7, 8]).

Given that SRC cultivation differs significantly from the cultivation of conventional crops, one can expect that expansion of SRC cultivation will bring about changes in how agriculture affects the environment. Dimitriou et al. [4] present an overview of effects of SRC on biodiversity, soil and water, drawing on a literature review and studies made in the Rating-SRC project. Rowe et al. [9] present the outcome of an assessment of the evidence base for potential impacts of large-scale deployment of, primarily, dedicated lignocellulosic crops, principally within the UK context. Karp et al. [10], also focusing on the UK, present a broad-based assessment of increased cultivation of perennial energy crops, drawing on environmental, social and economic research. Londo et al. [11] studied the impact of willow on groundwater quality and biodiversity in The Netherlands. Börjesson [12] evaluated the role of willow and grass cultivation in Sweden in reducing GHG emissions, nutrient leaching and soil erosion as well as in providing phytoremediation, improving soil fertility and biodiversity and cleaning waste water. EEA [13] discusses a selection of impact categories (soil erosion, soil compaction, loss of nutrient and agro-chemicals to the surface and groundwater, water abstraction, fire risk, biodiversity and crop diversity) in relation to a range of crops including both the main arable crops (e.g. potato, cereals) and SRC (e.g. reed, eucalyptus, willow and poplar).

Examples of studies that focus on specific aspects of SRC cultivation include Dickinson and Pulford [14] and Berndes et al. [15] who evaluate willow systems for cadmium phytoextraction. Börjesson and Berndes [16], Dimitriou and Rosenqvist [17], and Dimitriou and Aronsson [18] assess the use of willow SRC systems as vegetation filters for capturing nutrients in pre-treated municipal wastewater and run-off water from intensively cultivated croplands. Studies on biodiversity in plantations of fast-growing trees arrive at contradictory conclusions, especially when different kinds of organisms are considered [19]. The contribution of willow and poplar SRC to biodiversity in agricultural areas is described in various studies [20–27].

The actual impact of SRC in Europe will depend on how SRC expansion influences current land use and on the impacts of new land use relative to earlier use. Thus, implications of SRC expansion will be determined by how farmers react when conditions (market, policy) for their different land-use options change. Equilibrium models have been used to study the land-use response to increased demand for bioenergy feedstock, so far mainly focusing on conventional crops suitable as feedstock for production of ethanol and biodiesel. The causes of land-use change are multiple, complex, interlinked and vary over time. Quantifications of these changes are therefore uncertain, and reported results vary substantially—especially concerning the indirect land-use changes (LUCs) that can arise as a consequence of more land being dedicated to the production of bioenergy feedstock [28]. Modelling studies require large amounts of data and calibration for real world agronomic, technology and economic conditions, which remains a challenge. The model results have been found to be sensitive to many factors that can develop in different directions—including land-use productivity, trade patterns, prices and elasticities and use of by-products associated with biofuel production. Achieving a consensus on the extent of the LUC effects is unlikely in the near future. The modelling of new types of crops, such as SRC, is further complicated by the lack of data and limited information on how farmers react to varying conditions for such crops [28].

The prospects for SRC expansion can be analysed based on considering various constraints with the objective of quantifying biomass supply potentials under the condition that specific requirements (e.g. food and fibre supply, soil and water protection) are prioritised. More data-intensive studies combine soil, climate and topography databases with information about non-biophysical constraints (e.g. legally protected areas) and production functions that generate achievable output of given agriculture products as a function of biophysical parameters and assumed agronomic inputs (see, e.g. [7, 8]). This type of study can also be combined with sustainability appraisal frameworks in which stakeholders engage with scientists to assess the performance of different SRC implementation plans in relation to specified objectives. For example, the constraint-mapping performed by Lovett et al. [5] has been used for this purpose, as described by Karp et al. [10]. Other approaches are more action-oriented, focussing on the discourse of sustainability, including consensus-building and other stakeholder-oriented processes (see, e.g. [29]).

These approaches are all valuable and yield important information. The study reported in this paper aims at a complementary approach to provide an integrated synthesis of potential implications that large-scale production of SRC may have in the European context. This integrated approach depends not only on the specific location (soil type, groundwater table and climate) but also on the cropping system and general crop management. This kind of analysis requires a comprehensive effort based on specific data provided by experts comparing the impact of SRC with that of other arable crops, on soil, water, biodiversity, economics and energy supply.

Programmes that could provide such data and expertise are rare. The Rating-SRC project, has been used in this study to generate data and expertise for a comprehensive, multi-disciplinary evaluation of the pros and cons of SRC. In this project, experiments in willow and poplar SRC fields were established in Sweden and Germany, looking into the effects on all of the above-mentioned environmental aspects (Table 1). In most cases, the SRC stands used in the various experiments under this project were commercial fields grown for a number of years with known management and were compared with adjacent arable crops. In some locations, sets of common investigation plots were established to evaluate the impact of more than one environmental aspect (Table 1).

The objective of this paper is to integrate some of the detailed technical project outcomes of the Rating-SRC project and to present a synthesis of the soil, water and biodiversity impacts of SRC expansion. Based primarily on results published elsewhere in this special issue, graphical representations of the multidimensional consequences—so-called radar or spider diagrams—are constructed. These are intended to provide an overview picture that allows quick assessment of the environmental consequences of SRC expansion. The synthesis builds on separate assessments of the consequences for soil, water and biodiversity, which are themselves summarized in the form of sub-diagrams showing the effects as measured using a number of specific indicators. Further details on the methodology, including the selection of the indicators and the assessment of SRC cultivation, are given in the text.

The paper is structured as follows: first, the methodology for assessing the effects of SRC expansion on soil, water and biodiversity is presented. The indicators selected for the assessment of effects are presented and a justification of their selection is given. Next, a scenario is presented for the cultivation of SRC in Europe in the light of increasing demand for bioenergy feedstocks under conditions of limited land and water availability. We present results of the assessments along the different dimensions using selected indicators and scenarios and discuss the sub-spider diagrams that have been constructed for this purpose. This is followed by a discussion and conclusion, including identified uncertainties and suggestions for further research.

Methodology

Table 1 provides an overview of the observation locations. Most plots are located in east central Sweden and north and central Germany. Field selection and data collection are described in Baum et al. [30] and Dimitriou et al. [31].

Scenario Design

Two scenarios—with and without SRC cultivation—are developed and compared. The first scenario, without SRC, is representative of the present situation in most agricultural areas, with local exceptions in countries like Sweden, Germany, Poland, Italy and The Netherlands. This scenario is

Table 1 Locations for observations in Sweden (S) and Germany (D)

	Name	Year planted	Species, clone	Reference field	Harvested[a]	Soil texture class (0–20 cm)	Previous use/before SRC	Observations
1	Billeberga I (S)	2002	W: Sven	Cereals	2008	Sandy loam	Sugarbeet	W
2	Billeberga II (S)	1994	W: Torhild	Cereals/ rapeseed	Annually	Loam	Cereals	W, S
3	Djurby (S)	1990	W: 78021	Cereals	2007/2011	Silty clay	Cereals	W, S, B
4	Forkarby (S)	1991	W: 78021	Cereals	2008	Silty clay	Cereals	W, S
5	French (S)	1994	W: 78021	Cereals (eco)	2007/2010	Clay loam	Cereals	W, S, B
6	Hacksta (S)	1994	W: Jorr, Rapp	Peas/ cereal	2008	Clay loam	Cereals	W, S
7	Hjulsta I (S)	1995	W: Jorr	Cereals	2008	Clay	Cereals	W, S
8	Hjulsta II (S)	1995	W: Jorr	Cereals	2008	Clay	Oil crops/cereals	W, S, B
9	Kurth (S)	1992	W: Ulv/Rapp	Cereals (eco)	2007/2010	Clay loam	Cereals	W
10	Lundby Gård I (S)	2000	W: Tora	Cereals	2005	Clay	Cereals/oil crops	W, S, B
11	Lundby Gård II (S)	1995	W: 78021	Cereals	2005	Clay	Cereals	W, S, B
12	Puckgården (S)	1992	W: 78112	Cereals	2008	Silty clay	Cereals	W, S
13	Skolsta (S)	1993	W: 78021, Orm	Cereals	2004	Silty clay	Cereals	W, S
14	Säva (S)	1993	W: Rapp, Orm	Grass	2007	Silty clay	Cereals	W, S
15	Teda I (S)	2000	W: Tora	Grass	2009	Silty clay loam	Cereals	W, S
16	Teda II (S)	1993	W: 78112	Grass	2007	Clay	Cereals/set aside	W, S
17	Åsby (S)	1996	W: Tora	Cereals	2008	Silty clay	Cereals	W, S, B
18	Georgenhof (D)	1996	P: N42, Max 4	Cereals	2008	Silty loam	Cereals	W
19	Thammenhain (D)	1999	P: Max 4, Graupa	Cereals	–	Sandy soil	Set aside/cereals	B
20	Cahnsdorf (D)	2004	P: Japan 104	Cereals	2008	Loamy sand	Cereals	W, S
21	Hamerstorf (D)	2006	P: Hybrid 275, Max 4,Weser 6;W: Tora, Tordis, Sven	Cereals	–	Sandy soil	P: grassland; W: cereals	B
22	Bohndorf I (D)	2006	W: Tordis, Inger	Cereals	2009	Sandy soil	Grassland	B
23	Bohndorf II (D)	2008	W: Tordis	Cereals	–	Sandy soil	Grassland	B
24	Bohndorf III (D)	2007	W:Tordis	Cereals	–	Sandy soil	Grassland	B
25	Fuhrberg (D)	1994	W: *Salix viminalis*, P: 18 clones	Fallow ground	2005, 2010, 2011	Sandy soil	Cropland	W, S
26	Fuhrberg (D)	2005	W: Tora	Fallow ground	2010, 2011	Sandy soil	Cropland	W, S
27	Fuhrberg (D)	2009	P: Androscoggin, Max 1–3, AF2	Fallow ground	–	Sandy soil	Fallow ground	W, S
28	Gülzow (D)	1993	W: Beaupré, 6, P	Cereals	2008	Sandy loam	Cereals	S

Source: Dimitriou et al. [71]

W willow, *P* poplar, *W* water, *S* soil, *B* biodiversity

[a] The given years refer to the last harvest occurred in spring

referred to as Business As Usual (BAU). In the second scenario, a 20 % share of the arable land is assumed to be used for SRC cultivation to provide biomass for bioenergy. The 20 % share is chosen as representing a future situation in which agriculture provides a significant part of the biomass supply for energy in the EU. Some resource assessments indicate that a 20 % share would not necessarily crowd out food production and is thus not directly ruled out on this basis alone. For instance, Fischer et al. [8] found that, by 2030, some 44–53 million ha (Mha) of cultivated land (of a total of 164 Mha) could be used for bioenergy feedstock production without putting food supply or nature conservation at risk. Other studies report both higher and lower estimates of land availability (e.g. EEA (2007): 20 Mha for EU25 by 2030 [13]; WBGU (2004): 22 Mha for EU25—no year given [32]).

Indications about the feasibility of a 20 % share can also be obtained from considering the demand side by looking at energy modelling studies that investigate how the EU can meet climate targets in the mid and longer term. For instance, ensuring a likely (>66 %) chance of achieving the goal of limiting global warming to <2 °C above pre-industrial temperatures (Copenhagen Accord) requires decreasing global emissions by 50–70 % of the 1990 level by 2050 provided global emissions peak by approximately 2015 and further emission reductions take place thereafter (see, e.g. [33]). This requires a drastic change of the EU energy and transport systems. Cultivation of crops to provide feedstock for biofuels for transport constitute a potential major land claim [34], but the stationary energy sector may also create a major biomass, and hence land, demand in the future.

For instance, modelling studies of the European power sector indicate that the demand for biomass for power could grow to very high levels (magnitudes corresponding to planting more than one third of EU cropland with plantations yielding on average 10 Mg DM ha^{-1} $year^{-1}$) in a few decades if emission trajectories compatible with the 2 °C target are aimed for [35]. The demand could become especially high in the absence of rapid development, deployment and expansion of fossil-fuel power generation technology with carbon capture and storage. Currently, there is no rapid large-scale progress in carbon capture and storage. The paying capacity for biomass in the stationary energy sector may also become high enough to make SRC production a highly competitive option for farmers [35].

Similarly, a recent review by the IPCC [36] of 164 long-term energy scenarios showed global bioenergy deployment levels in year 2050 ranging from 80 to 150 EJ $year^{-1}$ for 440–600 ppm CO_2eq concentration targets and from 118 to 190 EJ $year^{-1}$ for <440 ppm CO_2eq concentration targets (25th and 75th percentiles). For comparison, the energy content of the present global industrial roundwood production is around 15–20 EJ $year^{-1}$, and the energy content in the global harvest of major crops (cereals, oil crops, sugar crops, roots, tubers and pulses) corresponds to about 60 EJ $year^{-1}$. Organic post-consumer waste and residues and by-products from the agricultural and forest industries, which contribute a major portion of the biomass for energy today, will not suffice to meet the anticipated levels of longer-term biomass demand. Consequentially, much of the bioenergy feedstock would have to come from dedicated production.

Locally, in catchment areas or near bioenergy installations, higher shares of SRC may be found. Under the BAU scenario, arable crops are cultivated in a homogeneous, intensively managed, cleared agricultural landscape lacking any habitat structures (such as hedges, single trees, single forest patches, set asides/fallows, grass fields, margins, reeds, etc.). It is assumed that the arable land that has SRC plantations includes an effectual portion of each of the three main structure types of SRC (initial, shrub-like and tree-like stadiums), simultaneously, to establish adequate habitat qualities.

Selection of Indicators

Major impacts include how cultivation of SRC affects soil and water quality as well as biodiversity. These are discussed below. The order in which the impacts are discussed and individual indicators presented has been chosen at random.

Soil

Long-term effects of SRC on soil quality mostly focus on changes of soil organic carbon and of hazardous compounds, mainly trace elements [23]. The potential for storing carbon in agricultural land in relation to special features of the crop (leaf litter fall and fine root turnover, accumulation of and decomposition of roots and stumps and tillage) has been discussed by Hansen [37] and Makeschin [38]; others have provided empirical studies on SRC carbon storage [39–44]. Trace element (e.g. cadium) concentrations are another soil quality parameter [31, 45]. Removal of high amounts of cadmium from the top layers of the soil by willow shoots is discussed by Baum et al. [23], Klang-Westin and Eriksson [46] and Dimitriou et al. [47]. Uptake of copper, lead, zinc, chromium, nickel and arsenic has mainly been studied in experiments in potted plants [48–52]. The long-term impact of commercial willow SRC plantations on soil quality parameters such as pH, carbon, nitrogen and trace elements has not been investigated broadly on regular farmland in Sweden.

Water

The impact of SRC cultivation on water quality relates to the quality of ground and surface water, as well as surface runoff and soil erosion. Water quality studies [53, 54] mainly refer to application of nutrients and agro-chemicals (mainly herbicides) relative to prevailing application levels in common arable crops. Nitrate and phosphate in the groundwater are among the main water quality indicators for examining the impact of a certain crop (see [53, 54]). With respect to herbicide emissions, we concluded that the Rating-SRC project did not offer sufficient data for an evaluation of the SRC scenario. The groundwater table (or water balances) indicator shows differences in the water consumption patterns of different crops and has been used for biomass crops to show differences relative to "conventional" crops. Lower groundwater recharge can have a negative impact in areas where drought is evident but a positive impact in areas where flooding might occur [53]. Surface runoff is a common indicator of the risks of soil erosion, nutrient surface leaching to adjacent water bodies and flooding. Less surface runoff limits the risks of the above-mentioned effects [54–57].

Biodiversity

Biodiversity is evaluated along two dimensions that are common in SRC research: phytodiversity and breeding bird diversity. The SRC rating is based on the same scoring methods for both dimensions as it relies on species number differences as defined by the presence or absence of species, rather than measurements related to soil characteristics or water quality.

Phytodiversity SRC plantations enhance the structural diversity of the agricultural landscape. Thus, a positive effect on species richness is expected in accordance with the mosaic concept by Duelli [58, 59] that claims that the greater the number of different habitats within a landscape, the higher the species number, as each habitat has a characteristic flora (and fauna). This effect was confirmed by an analysis of the contribution of SRC plantations (sample area, 1,600 m^2) to species number of the higher landscape level (225 km^2) in five areas in Central Sweden and three areas in Northern Germany. These were areas where SRC plantations are a representative element. We found that the higher the number of habitat types, the higher the landscape species number and the lower the contribution of the SRC species number to the landscape species number [60]. The total number of plant species was chosen as an indicator to evaluate the influence of the 20 % SRC scenario on landscape diversity. Several authors have compared species diversity in SRC plantations and arable lands and reported higher species numbers in SRC plantations (cf. review by Baum et al. [23]). For a more differentiated evaluation and in order to analyse if all species communities are affected similarly, species communities typical for different habitats were chosen as indicators (the number of woodlands, ruderal, arable land and grassland species). Given the large impact of intensified agriculture on the loss of rare species with high intrinsic and possibly functional values in landscapes [27], the abundance of endangered species was also selected as a criterium for comparing both scenarios. The evaluation was done on the basis of vegetation data collected in 15 poplar and willow SRC plantations and adjacent arable lands in Central Sweden and Northern Germany. The sample size was 100 m^2 (cf. [30, 61]).

Breeding Bird Diversity Breeding birds have been chosen as model taxa whose specific space requirements and ecological amplitudes cover the dimensions of assessing the impact of SRC expansion in agricultural landscapes. Breeding birds are recognized bioindicators [61, 62] that may indicate both specific ecological parameters and complex habitat structures. Furthermore, breeding birds respond comparatively quickly to a changing environment. Regarding autecological habitat preferences, there are clear differences within the range of species occurring in SRC embedded in agricultural landscapes. Due to general habitat requirements, species were combined into ecological guilds [25, 63] to allow a more detailed assessment of the influence of SRC expansion on breeding bird species. Among other crucial habitat characteristics that have a major impact on the occurrence of breeding birds are height and coverage of vegetation, as well as structural composition and adjacent habitats. Regarding general habitat preferences, breeding bird species possibly affected by the establishment of SRC can be assigned to the ecological guilds of open land species, shrub species, forest species, tall ruderal/reed species and ecotone species [63, 64].

To assess the impact of SRC establishment on endangered species in terms of rating its conservational value, vulnerable species listed in the Red Lists of Germany [65] were considered separately. The rating was based on data collected in studies of breeding birds in several SRCs and adjacent arable lands in the German federal states of Brandenburg, Hesse and Saxony during the years 2007, 2008 and 2009 (Gruss and Schulz, in preparation). In total, 15 plots of poplar and 11 plots of willow were included in this study.

Scoring

No differentiation between willow and poplar was made in the spider diagrams. For some parameters, the impact can differ between the species from the two genera, but their relative impact compared to arable crops was considered more or less similar. In cases where the impact of SRC on some dimensions was not covered by the findings in the Rating-SRC project, the relevant information was obtained from the scientific literature. The rating of the impact of SRC on each dimension used a single standard based on a scale from 1 to 9. As a rule, the impact of the SRC scenario was compared to the BAU scenario not involving SRC. Situations that are considered positive (e.g. high carbon sequestration or reduced nitrogen loads in groundwater) are allocated higher scores (closer to 9); situations that are less desirable (e.g. enhanced mobility of toxic trace elements or reduced groundwater recharge) are allocated lower scores. Scoring is always relative, i.e. it refers to the difference between the SRC and BAU scenarios.

By definition, the situation in the BAU scenario is allocated a score of 5. Improvements are thus scored between 5 and 9, while less desirable situations are given scores below 5. Scoring of the SRC scenario is done by experts involved in the ERA-NET project. Based on field observations from their research (as described above), they provide an assessment how a given indicator would change under the SRC scenario as compared to a situation of pure non-SRC cultivation. Scoring of the changes has been synchronised as follows. A very significant deterioration (as compared to the BAU scenario) was assigned a score of 1 point, a significant deterioration a score of 2.5, a slight deterioration of 4, no change of 5, a slight improvement of 6, a significant improvement of 7.5 and a very significant improvement of 9 points. Intermediate results are given matching scores. For phytodiversity and breeding bird diversity, however, evaluation of the SRC scenario was done according to the scale given in Table 2. This approach was chosen to ensure maximum transparency in the assessments by the different scientists involved in the project.

We assumed that the establishment of 20 % SRC—regarding the spatial reference frame (500–5,000 ha)—did not affect the occurrence of species which appear commonly on arable land. Therefore, all species detected in our study on arable land without any functional influence of adjacent habitats, e.g. SRC, were supposed to be part of the "standard inventory".

Primary Results

Effects on Soil Quality

Seven dimensions were chosen to analyse the impacts of SRC cultivation on soil quality: (1) carbon (C) sequestration, (2) bulk density, (3) phosphorus (P) availability, (4) nitrogen (N) availability, (5) soil erosion, (6) pH and mobility of trace elements (excluding cadmium) and (7) concentration of cadmium (Cd) in the soil. Results of the SRC scenario are depicted in Fig. 1. Introduction of 20 % SRC cultivation leads to enhanced C sequestration and decreased soil N availability, which decreases N loss and decomposition rates and increases the stability of the organic matter in the soil [66]. Bulk density is slightly increased compared to tilled arable land, which may have some impacts on root and crop development of SRC [67]. Availability of soil P is decreased, which will eventually affect the P supply for crops [67]. Soil erosion is reduced. The impact on heavy metals is ambiguous. On the one hand, a decrease in soil pH is expected to increase metal mobility. On the other hand, increased mobility will lead to increased metal uptake by SRC plants, thus improving the soil remediation function [68]. Especially cadmium concentrations can be reduced significantly in the long run due to enhanced uptake and removal by SRC crops [69].

Effects on Water

Four dimensions were chosen to evaluate the impact of SRC cultivation on water quality: (1) nitrate–nitrogen concentrations in groundwater, (2) phosphate–phosphorus concentrations in groundwater, (3) surface runoff and (4) impact on groundwater recharge. Results of the SRC scenario are depicted in Fig. 2. Introduction of 20 % SRC cultivation leads to a large reduction of nitrate–nitrogen concentrations in the groundwater (reduction by a factor 20 according to Dimitriou et al. [71]; Schmidt-Walter and Lamersdorf [72]). However, the respective concentrations of phosphate-phosphorus increase slightly, leading to a small deterioration of water quality. Whole-year land cover and permanent root system of SRC leads to a reduction in surface runoff and soil erosion, thus improving surface water quality and limiting flooding risk for water bodies [54–58]. As water percolation is expected to decline, groundwater recharge may be limited, leading to a slight lowering of the groundwater table. Cultivating fallow land with willow, for example, may double rainfall interception and increase water transpiration by 30 % [72]. Based on this, it is assessed that introduction of 20 % SRC cultivation leads to a slight improvement in surface water quality and lower flooding risk, and slight lowering of the groundwater table, on the whole and at the catchment level.

Effects on Biodiversity

The impact of SRC introduction on biodiversity is assessed by separately defining expected changes in plant species (phytodiversity) and breeding bird species (zoodiversity). Results are depicted in Fig. 3a and b, respectively.

For phytodiversity, the following elements are evaluated based on results reported by Baum et al. [30]: (1) total number of plant species, (2) number of woodland species, (3) number of ruderal species, (4) number of arable species, (5) number of grassland species and (6) number of endangered species. In a homogenous landscape of pure arable crops, replacing 20 %

Table 2 Evaluation of phytodiversity and bird diversity based on differences in species numbers between the scenario 'BAU' and 20 % SRC integrated in cropland area: BAU was set 5 points

Change in species number	Score	Explanation
Decrease >66.7 %	1	Very significant deterioration from BAU
Decrease >33.3 %, ≤66.7 %	2.5	Significant deterioration from BAU
Decrease >5 %, ≤33.3 %	4	Slight deterioration
Around 5 %	5	No change to BAU
Increase >5 %, ≤33.3 %	6	Slight improvement from BAU
Increase >33.3 %, ≤66.7 %	7.5	Significant improvement from BAU
Increase >66.7 %	9	Very significant improvement from BAU

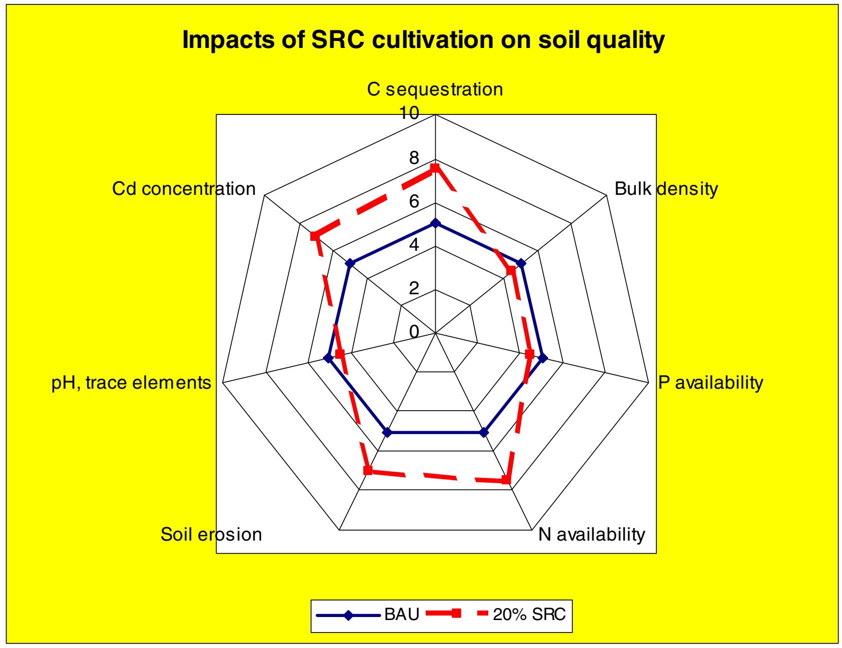

Fig. 1 Impacts of SRC cultivation on soil quality. Figure based on data collected and analysed in Dimitriou et al. [31] and Baum et al. [70]

of the crops by SRC will have a positive impact on phytodiversity (Fig. 3a). All elements are significantly improved, with the exception of the number of endangered species that were absent in both scenarios. Especially the number of grassland species improves (a 17-fold increase, with arable systems on average counting 0.6 grassland species versus 10.1 for arable land plus 20 % SRC). The total species number increases 5-fold from on average 6.2–32.9 species. The number of woodland species increases from 0 to 5.2 in the 20 % SRC scenario, the number of ruderal species increases from 2.7 to 8.1 and the number of arable field species increases from 2.4 to 3.8. The SRC plantations contain predominantly common species.

In general, the addition of SRC in homogeneous agricultural landscapes also increases breeding bird diversity (Fig. 3b) [73]. The exact extent of this effect depends on the age and structure of prevailing SRC trees, e.g. initial or recently harvested SRC, its shrub phase (height of the trees exceeding 1 m) and full grown trees (height >about 8 m). Each favours different bird species [63, 73–79].

Due to the structural enrichment, the establishment of SRC leads to higher numbers of breeding bird species in very poorly structured cropland. The total number increases almost 4-fold, from 10 to 37 species. Especially forest and shrub species benefit very significantly from the increased habitat availability offered by the addition of SRC. Due to the absence of suitable habitat structures such species are absent in poorly

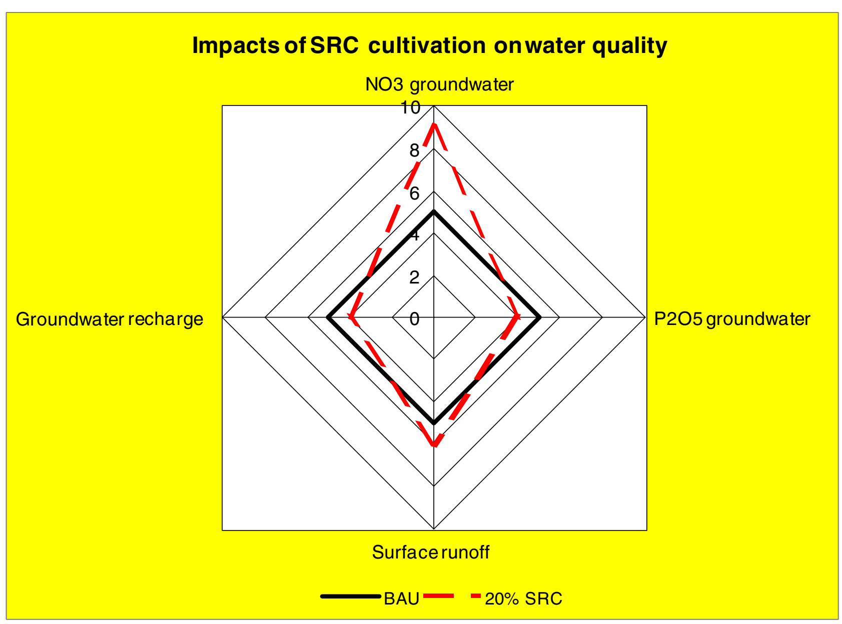

Fig. 2 Impacts of SRC cultivation on water quality. Figure based on data collected and analysed in Dimitriou et al. [71] and Schmidt-Walter and Lamersdorf [72]

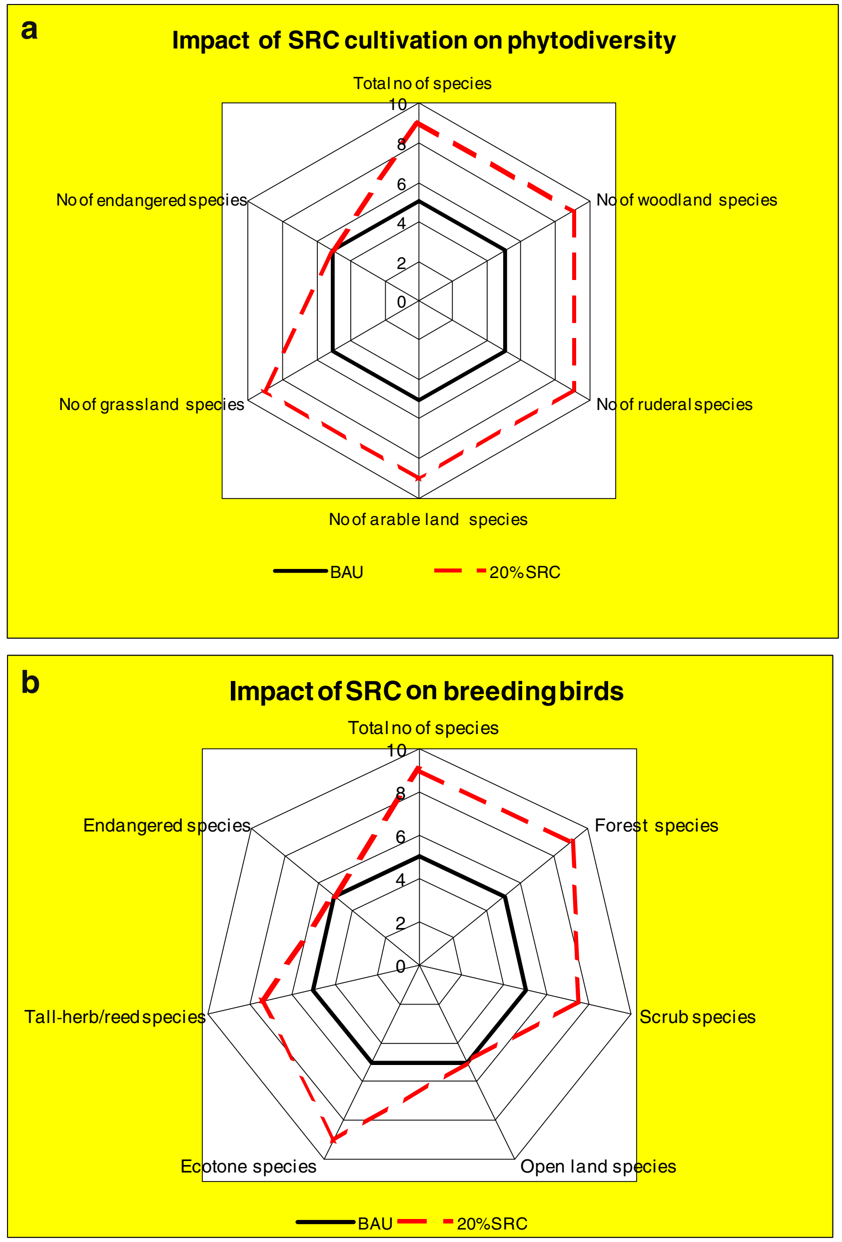

Fig. 3 **a** Impacts of SRC cultivation on phytodiversity. Figure based on data generated and analysed by Baum et al. [30]. **b** Impacts of SRC cultivation on breeding bird diversity. Figure based on data generated and analysed by (Gruss and Schulz, in preparation)

structured cropland, but in the scenario, the species numbers of each guild increases by up to 12 species. Furthermore, the numbers of ecotone species and species that inhabit ruderal habitats and reeds increase from three in poorly structured cropland to five species after the establishment of SRC. In terms of qualitative rating the increase is slighter than that of forest and shrub species, but the addition of SRC might especially favour the densities of ecotone and ruderal species compared to those in croplands [81]. Such species can benefit from the increase in margins and the enrichment with vertical structures by SRC.

The addition of SRC will not significantly increase the number of vulnerable species (3), in spite of the very significant improvement in species numbers. Almost all species attracted by SRC are common, less demanding in terms of habitat requirements, and thus not listed in the Red Lists [76, 77, 79]. However, many typical breeding bird species inhabiting cropland—and thus detected there in our studies—are currently endangered due to intensification and homogenization of agricultural landscapes [80, 81]. To conclude, the cultivation of SRC in homogeneous croplands may increase the breeding bird diversity very significantly but is of low conservational value.

Discussion and Conclusions

It is beyond the scope of this paper to compare the results of the individual analyses (soil, water and biodiversity) of the ERA-NET project to studies presented elsewhere. Instead, we will briefly summarize the outcome of the SRC scenario that has been defined above. Furthermore, we will reflect on the strength and weaknesses of the approach presented here and its ability to play a role in the evaluation of bioenergy policies.

The introduction of SRC in European arable cropping systems will affect soil quality in different ways. As per Dimitriou et al. [31], it may be good for carbon sequestration, which will help to reduce GHG emissions. Soil density will be somewhat compacted as the topsoil no longer is ploughed. Under certain conditions, however, increased soil carbon sequestration can in the long run lead to a decrease of the bulk density [41]. Availability of N improves, but P availability is reduced. Soil erosion is expected to decline. Due to lowering of soil pH, mobility of trace elements including toxic heavy metals will increase. This may enhance removal of cadmium from the soil.

SRC may further help to reduce the amount of nitrate in the groundwater, as nitrogen applications are generally lower than for arable crops. This can especially be valuable in areas with sandy soils under intensive agriculture (crops and grassland). Groundwater concentration of phosphates, on the other hand, will show a slight increase. SRC may also lower the groundwater table, as groundwater recharge is lower than for arable crops, and may therefore add to problems of limited water availability. Replacing arable crops by SRC, on the other hand, can be an effective way to reduce surface runoff and displacement of soil particles.

Phytodiversity generally is significantly enhanced under the SRC scenario, while SRC does not affect the number of endangered plant species. SRC is especially favourable for grassland species. As to breeding birds, SRC is beneficial for most species types, the total number of species and species heterogeneity (not included in the spider diagram). There is no effect, however, on open land species. Summarizing, the biodiversity impact of the 20 % SRC scenario is very positive, especially for phytodiversity. However, species encroach SRC plantations from the surrounding landscape (cf. [23, 70]). Therefore, it will take some time before they have been established in a scenario where 20 % SRC is found in an environment of 80 % cropland. Furthermore, SRC contribution to diversity will be lower in areas with additional land uses, for example grasslands or forests.

Concluding, the introduction of large-scale SRC cultivation in Europe may have both positive and negative effects, depending on the exact implementation. SRC can generate biomass for bioenergy production and thus help reduce GHG emissions. The exact reduction will obviously depend on the energy required for cultivation, harvesting, transport and conversion. Reductions may be accelerated by the option to store additional carbon in the soil, but actual sequestration will depend on land use history, the crops that are replaced and the prevailing soil and weather conditions. SRC can, further, help to reduce nitrate in groundwater and prevent soil erosion or surface runoff. It can be applied in phytoremediation, especially for cadmium and has a positive impact on biodiversity of plants and breeding birds (with the exception of grassland, arable and open land species). SRC should not be applied in dry areas or on soils high in toxic trace elements (with the exception of cadmium).

These results are in line with expectations and do not seem to contradict other findings reported in the literature. They present illustrative outcomes of detailed analysis supported by a range of measurements and observation programs covering 28 experimental plots presented elsewhere in this issue. The process of defining a scenario for large-scale SRC introduction in a real-world situation (as opposed to observations in relatively small but carefully selected SRC plots in parts of the study area) followed by a transparent process of scenario evaluation has been helpful in obtaining a more complete and realistic understanding of the potential impacts of SRC cultivation in practice. Insights from this process can be valuable for the design of policies affecting SRC.

It is emphasised here that the outcomes refer to a significant land use change from a monocultural landscape dominated by the cultivation of arable crops and lacking landscape elements, to a landscape characterized by evenly distributed plots with heterogeneous SRC crops covering in total 20 % of the arable area. This is especially relevant as today, locally, there are clear trends of increasingly monotone landscapes. One example is the growing dominance of silage maize in parts of Germany as steered by the expansion of biogas production in this country.

The outcome of the analysis is obviously influenced by the way environmental and ecological impacts of the SRC scenario have been assessed and presented. There is a range of indicators that could have been included in the analysis, but the chosen selection (and evaluation) of the indicators is corroborated by the literature as has been presented above. This holds especially for impacts on water and soil quality, although these clearly depend on the soil type and climate conditions [70]. As the evaluation is based on results from mineral soils under temperate climatic conditions, its validity is restricted to corresponding conditions.

The SRC impact on biodiversity could alternatively have been assessed by evaluating its effect on the prevalence of mammals. Willow, for example, is thought to increase the amount of wild animals/game for hunting (deer, reindeer and moose) and also other mammals, such as rabbits (see [4, 55]), but studies for SRC and mammals are rare, and resources in the project did not provide for an extensive observational mammal programme. Studies on the prevalence of invertebrates (e.g. earthworms, spiders, butterflies and beetles) in SRC are more common, but the impact of SRC on these animals is in some cases similar to the impact on breeding birds [4] (e.g. willow SRCs have a positive impact on honeybees but a negative impact on carabids [82]).

Other sources of uncertainty with respect to the outcomes presented here are related to the way SRC production may be distributed in the future. An even distribution of SRC, as evaluated in this paper, is not relevant for cases of regional specialisation, where specific regions concentrate on either specific arable (food) crop production or on dedicated energy crops including SRC. As a rule, the effects of the SRC scenario presented here are more likely to occur (and occur more strongly) where the existing landscape is dominated by arable crops. Impacts will be less evident when arable land alternates with intermittent landscape elements (trees, hedges, etc.).

Our approach presents results from the ERA-NET project in an integrative and coherent way. Spider diagrams offer a graphical representation of the evaluation that is both clear and condensed, allowing us to present field observations on environmental quality from different disciplines in a uniform and coherent way. Limitations of the current approach (i.e. a scenario analysis of SRC implementation using detailed technical, disciplinary information from scientific analysis) are mainly related to issues of data availability and—especially—data compatibility. This could be solved in our analysis by using data from a single (extensive) research project. A further limitation is the need to combine data from a wide range of observations and disciplines.

Restrictions of the method also refer to its limited ability to cover spatial diversity and dynamic processes. For example, evaluation results do not provide insight into how long it takes for a given result (e.g. a positive change in groundwater quality or prevalence of breeding birds) to be realised, nor does it explain how variations in local soil or climate conditions affect the results. Furthermore, it does not integrate stakeholder views or other discursive elements: It does not allow different groups of stakeholders to express specific views, e.g. on the importance of toxic trace element release, or on changes in the water cycle.

Selection of the indicators, their implementation in an assessment procedure, and (graphical) presentation of the results are exercises involving subjective judgments and decisions. The activities described in this paper have been carried out in a transparent, coherent and sincere way, but this does not guarantee that the outcomes are completely free of biases or that other scientists would come to the same conclusions. This is further complicated by the fact that some of the indicators relate to similar generic impacts (such as ‘soil quality’ or ‘nutrient management’). The number of indicators included in the analysis, and the order in which they are presented, may influence the way readers perceive SRC impacts. The influence of different subjective decisions made during the process could be assessed by presenting alternative approaches (e.g. presenting different indicators or presenting them in an alternative order), but this is beyond the scope of this paper, which is to present an overview of the ERA-NET project results.

Notwithstanding these limitations, the use of field observations and expert knowledge in the evaluation of an SRC scenario has provided useful information on how policies could affect the environmental and ecological characteristics of the rural landscape in Europe. In this way, our approach offers distinctive opportunities for policy evaluation that is integrative, land-based and clearly corroborated by field observations. Other approaches may provide better options when other qualities are required. GIS-based modelling work will be able to better define impacts of variations in soils, landscapes, climates and land use. Economic models could provide stronger representation of economic and market development, and of farm decision-making in crop cultivation, and process-oriented approaches will allow stronger and clearer stakeholder interaction. Any approach that is chosen will, however, represent a compromise among availability of data, expertise and modelling tools.

The need for comprehensive analytical frameworks, such as the approach presented here, may be growing, as the dimensions for policy analysis seem to be increasing. Issues related to (changes in) land use may be especially complex, requiring land-based information and analytical systems and affecting food and feed production as well as natural resources and nutrient (and other) cycles. Such frameworks may be used to support the work of international platforms and partnerships that have been developed to address sustainability issues. For example, the Global BioEnergy Partnership (GBEP), initiated in 2005 by the eight most important economic nations plus five emerging nations (Brazil, China, India, Mexico and South Africa), developed a list of 24 indicators to evaluate sustainability performance of bioenergy production systems (see Table 3 in Appendix). They provide a comprehensive set of key points of reference for the evaluation of bioenergy policies, programmes and projects. GBEP indicators cover issues evaluated in Fig. 1 (soil impacts; GBEP indicators 1 and 2) and Fig. 2 (water; GBEP indicator 5).

Spider diagrams—or similar methods that can provide a generic visual (quantified) representation of sustainability assessments—could be used to represent outcomes in platforms and organizations such as GBEP. They may be especially helpful when large numbers of indicators are used, as is the case of sustainability standards. One example of a large set of indicators [associated with the Roundtable for Sustainable Biofuels (RSB)], is presented in Table 4 in Appendix as an illustration of how complex sustainability assessments can be in practice. As demonstrated in Table 4 in Appendix, RSB has identified nine principles, close to 40 criteria, and more than 200 indicators of which (nearly) 30 deal with water and conservation and more than 45 are related to human and labour rights. Spider diagrams could be helpful in the generalisation and presentation of the results of evaluations made using such a framework.

Appendix: International Sustainability Indicators

Table 3 GBEP indicators

Environmental	Social	Economic
1. Life-cycle GHG emissions	9. Allocation and tenure of land to new bioenergy production	17. Productivity
2. Soil quality	10. Price and supply of national food basked	18. Net energy balance
3. Harvest levels of wood residues	11. Change in income	19. Gross value added
4. Emissions of non-GHG air pollutants including toxic components	12. Jobs in bioenergy	20. Change in consumption of fossil fuels and traditional biomass use
5. Water use and efficiency	13. Change in unpaid time spend in collecting biomass	21. Training and re-qualification of workforce
6. Water quality	14. Expansion of access to modern energy services	22. Energy diversity
7. Biological diversity in the landscape	15. Change in damage due to indoor smoke	23. Infrastructure and logistics for bioenergy distribution
8. Land use (change) related to bioenergy feedstock production	16. Incidence of occupational injury, illness, fatalities	24. Capacity and flexibility of bioenergy use

Source: GBEP [83]

Table 4 RSB indicators

Principles	Criteria	Indicators
1. Legality	(a) Comply with local laws and regulations and relevant international laws and agreements	3
2. Planning, monitoring, improvement	(a) Risk assessment, implement sustainability plans, (b) stakeholder consultation according to free, prior, informed consent, (c) business plan addressing long-term economic viability	17 (5, 8, 4)
3. Greenhouse Gases	(a) Comply with applicable LCA, policy regulations, (b) GHG calculations following well-to-wheel principles including indirect land use change, (c) on average >50 % GHG emission reduction	10 (3, 4, 3)
4. Human and labour rights	(a) Freedom of organization, (b) no slave or forced labour, (c) no or very limited child labour, (d) no discrimination amongst workers, (e) wages and working conditions respect international law, (f) maintain international health and safety standards, (g) ensure human rights and labour rights	47 (4, 5, 6, 3, 13, 14, 2)
5. Rural and local development	(i) In poor regions: improve the status of stakeholders, (ii) special measures to enhance participation of women, youth, indigenous peoples in poor regions	15 (12, 3)
6. Food security	(a) Assess risk food security, mitigate negative impacts, (b) enhancement of affected stakeholders in food insecure areas	8 (5, 3)
7.Conservation	(a) Conservation values maintained or enhanced, (b) maintenance of ecosystem functions, (c) protection of buffer zones, (4) protection of ecological corridors, (5) prevention of invasive species	29 (10, 3, 3, 6, 7)
8. Soil	(a) Maintain, enhance soil physical, chemical, biological conditions.	8
9. Water	(a) Respect water rights, (b) include water management plan, (c) no depletion of water resources, (4) enhance, maintain water resources	30 (8, 7, 8, 7)
10. Air	(a) Identify and minimize emissions, (b) avoid or eliminate open-air burning,	5 (2, 3)
11. Technology	(a) Full availability of technology, (b) minimize risk of damage from technology use, (c) containment of microorganisms, (d) good storage practices, (5) no damage caused by residues	29 (2, 8, 4, 8, 7)
12. Land rights	(a) Existing land rights are assessed and established, (b) interaction based on free, prior and informed consent	11 (4, 7)

Source: RSB [84]

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