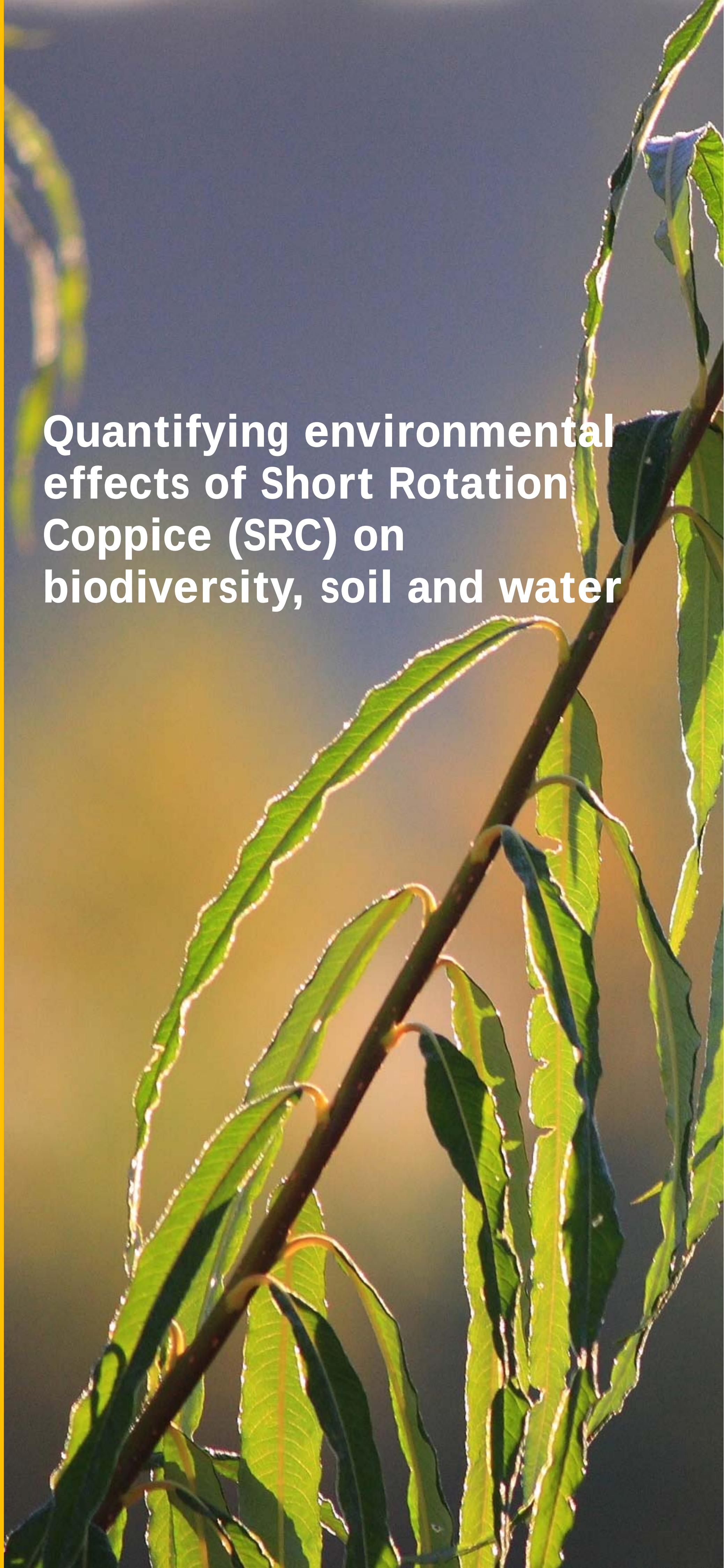
Quantifying environmental effects of Short Rotation Coppice (SRC) on biodiversity, soil and water



This report was prepared as a joint effort by the Swedish University of Agricultural Sciences (SLU), Johann Heinrich von Thünen-Institute (vTI), University of Rostock, Chalmers University of Technology, Georg-August University of Goettingen, University of Applied Sciences Eberswalde, Biop Institut and Buro for Applied Landscape Ecology and Scenario Analysis. The report addresses environmental effects of short rotation coppice production. The purpose of the report was to produce an unbiased, authoritative statement on this topic based on a review of relevant scientific literature.

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Quantifying environmental effects of Short Rotation Coppice (SRC) on biodiversity, soil and water

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KEY MESSAGES

The term Short Rotation Coppice (SRC) refers to cultivation systems using fast-growing tree species with the ability to resprout from the stump after harvest. Harvest occurs in short intervals, 2-6 years, and management practices (soil preparation, weed control, planting, fertilisation, harvest, etc.) are more similar to those of agricultural annual crops than to forestry, despite the fact that the species currently used in commercial SRC plantations in Europe are fast-growing species with good coppice ability that achieve high biomass yields, such as willows (*Salix sp.*) and poplars (*Populus sp.*).

SRC is considered a promising means to meet the different targets set in Europe to increase renewable energy. The current areas with SRC are very few compared to other agricultural crops but a series of predictions suggest for a rapid increase. SRC is a perennial crop that differs from agricultural crops with respect to a number of physical traits and is managed quite differently; a potential large-scale shift from currently-grown agricultural crops to SRC will undoubtedly have implications, both positive and negative, on a range of environmental issues.

Cover Picture: Courtesy of Pär Aronsson

INTRODUCTION

This report reviews the potential effects of Short Rotation Coppice (SRC) cultivation on water issues, such as groundwater quality and hydrology, on phytodiversity and animal diversity, and on soil issues concerning soil quality, soil organisms, and soil carbon. This report addresses the biomass production systems cultivated for energy purposes usually referred to as “SRC”. This term refers to systems using fast-growing tree species with the ability to resprout from the stump after harvest, in which harvest occurs in short intervals, 2-6 years. The management practices for SRC (soil preparation, weed control, planting, fertilisation, harvest, etc.) are more similar to those of agricultural annual crops than to forestry practices, despite the fact that the species currently used in commercial SRC plantations in Europe are tree or bush-formed species such as willows (*Salix sp.*) and poplars (*Populus sp.*), i.e., fast-growing species with good coppice ability that achieve high biomass yields even under very short harvest intervals. In this report, we consider SRC a variety of Short Rotation Forestry (SRF). SRF is a broader term describing forest systems for biomass production (for energy purposes among others) using fast-growing tree species grown at denser spacing and elevated maintenance than in traditional forestry, typically harvested after 2 to 25 years depending on the desired end-product. Therefore, SRC falls within SRF and represents a more specialised and intense practice of SRF dedicated mainly for energy purposes. In this report, we explicitly describe the effects of SRC on the environment, referring in a few parts also to effects of single stem trees used in SRF.

Figure 1. Pictures of SRC cultivation at different stages. Top left: SRC poplar field (Germany); Top right: Planting willow SRC (Sweden); Bottom left: Inside a willow SRC of two-year growth (Sweden); Bottom right: Poplar SRC field with one-year growth (Spain).

SRC for production of biomass for heat and electricity is considered a very promising means to meet the different targets set in Europe to increase the amount of renewable energy (EEA, 2006). SRC has also been identified as the most energy efficient carbon conversion technology in the field of bioenergy for reducing greenhouses gas emissions (Styles and Jones, 2007); additionally, larger-scale SRC cultivation could help contribute to the social and economic targets of other EU policies (e.g., EU Rural Development, CAP reform).

Such technological and political drivers have stimulated the interest in growing and processing biomass crops as a source of renewable energy; different incentives for growing SRC have been introduced in several European countries. Currently, there are c. 14,000 ha willow SRC cultivations in Sweden, mostly on productive agricultural land, and smaller areas of SRC in Italy (c. 6,000 ha, mostly poplar), Poland (c. 3,000, mostly willow), the UK (c. 3,000 ha, mostly willow), Germany (c. 1,500 ha, mostly poplar), and other European countries. These areas are not as extensive as for other agricultural crops (for instance, in Sweden the SRC area is only 0.5% of the total agricultural land in the country), but a rapid increase of SRC has been predicted in the short-term in several European countries. For example, in Sweden the Swedish Board of Agriculture predicts a short-term increase of SRC to 30,000 ha (Jordbruksverket, 2006); the UK Biomass Strategy predicts that perennial energy crops will occupy some 350,000 ha by 2020 (DEFRA, 2007), and in Germany, SRC cultivation area may also increase markedly during coming years due to a changing subsidy policy and the identification of high cultivation potentials for certain areas (e.g., 200,000 ha for the federal state of Brandenburg; Murach et al., 2008). Such predictions have failed in the past, but they represent the expected development of SRC in Europe.

SRC is a perennial crop that differs from agricultural crops with respect to a number of physical traits and is managed quite differently, as well. In particular, it is anticipated that SRC plantations will remain in place for a number of years (10-25 years depending on national regulation and market issues), taking the land out of agricultural crop rotation. In northern Europe, harvest normally occurs in winter or early spring, when the soil is frozen to avoid soil compaction and when the need for fuel is the greatest. The plants are deeper-rooted and generally have high water consumption compared with conventional crops. SRC is much taller (c. 5-8 m at harvest) than other arable crops. In addition, once established, SRC requires no annual soil cultivation and considerably less agrochemical input. Typically, much less nitrogen fertiliser is applied to SRC compared with agricultural crops (Gustafsson et al., 2007). In fact, the vast majority of Swedish and UK SRC fields are not supplied with inorganic fertilizer at all. Minimal or no fungicide and insecticide are applied, although herbicides are needed during the establishment phase.

Figure 2. Different stages of willow SRC cultivation; harvested field; regrowth during the same year; after one growing season; harvest after 4 years. The first three pictures show the same SRC willow field (in Uppsala, Sweden).

The lower intensity of SRC cultivation, particularly the lower nitrogen fertiliser application, results in a much lower carbon footprint compared with food or biofuel production based on annual agriculture food crops (Heller et al., 2004). A potential large-scale shift from currently-grown conventional agricultural crops to SRC will undoubtedly have implications, potentially both positive and negative, on a range of environmental issues. A concentrated increase of SRC grown on agricultural land is anticipated in areas neighbouring biomass-fueled heat and/or power plants (with approximate radius from power stations of 50 km). In such areas, SRC might need to be cultivated on a substantial fraction of all available agricultural land to economically and energy-efficiently meet the biomass needs for fuel. Where SRC is grown to supply small local heat and/or power stations, plantations will be much smaller scale, although they may still be concentrated. This, coupled with the above-mentioned special features of SRC will surely alter the appearance of the landscape and have potential implications for the local water and soil quality, hydrology, carbon storage in soil, and biodiversity. The following parts of this report include information about the potential effect of SRC on all these issues, and discuss the expected positive or negative impact of SRC cultivation at the micro- (field) and macro- (catchment and larger) scale based on the existing literature. Such implications need to be known or indicated if decision-makers implement incentives toward a rapid increase of SRC in certain areas. Therefore, this report focuses on research results related to the above topics, identifies possible gaps in knowledge, and presents qualified assumptions for potential impacts of SRC cultivation.

SRC IMPACT ON WATER (WATER BALANCES AND WATER QUALITY)

Salix and *Populus* are used for SRC for production of biomass for energy mainly because they are fast-growing and produce more biomass, compared to other tree species. Especially in central and northern Europe such biomass systems were initiated and adapted commercially (Christersson and SennerbyForsse, 1994). Increased biomass yields have been linked with high water use, and willows and poplars are commonly adapted to grow in places with high water availability such as river banks. Therefore concerns about the effects on local hydrological balances and flow to neighbouring streams/rivers have been expressed in several reports predicting future biomass supply from agriculture (EEA, 2006; Eppler et al., 2007; EEA, 2008). Fast canopy development and high leaf area index during the growing season are the special features of willow and poplar significantly affecting transpiration rates from leaves and interception evaporation from the canopy. Potential deeper rooting of SRC species compared to annual crops may also favour higher rates of water consumption.

From the several estimates for evapotranspiration for poplar and willow presented in Table 1, it seems there are variations in the figures reported depending on the location of SRC field (soil type), the local climatic conditions but also between years, the species or clones/genotypes used, and the age of the plantation. Variations can also be attributed to the methods used for estimation of evapotranspiration. Persson and Lindroth (1994) simulated seasonal evapotranspiration (May to November) to be 360 to 404 mm for irrigated and fertilized willow SRC grown on clay soil in southern Sweden for four years. Persson (1995) estimated that the average seasonal evapotranspiration (May to October) from six fields in different locations in southern Sweden was 435 mm. For poplars at the Lusatian mining region in Germany evapotranspiration was calculated to be between 404 and 373 mm during 1996-2002, for two different poplar clones, respectively (Beaupré, *Populus trichocarpa* x *P. deltoides* and *Androscoggin*, *P. maximowiczii*), respectively. Annual evapotranspiration was calculated by Knur et al. (2007) to be 356 mm and 359 mm for a 3- and a 9-year old SRC poplar plantation, respectively, located in Neuruppin, Germany. Another 9-year old poplar plantation was estimated to transpire 480 mm during the vegetation period (April to November) in middle Saxony, Germany (Petzold et al., 2008). In the UK, Hall (2003) estimated that approximately 600 mm water per year is used by SRC willow grown on a clay soil which annually receives precipitation of 700 mm, i.e., higher than the figures estimated in Sweden and Germany. To add to the uncertainty about which species is a higher “consumer” of water, Linderson et al. (2007) found that the estimated transpiration rate of a willow stand with different clones in southern Sweden from April to October was between 100 and 325 mm (markedly lower than the previous findings for poplar), whereas the Penman-Monteith transpiration in that area for willow reached 400 to 450 mm for that period. This was attributed to the relatively high temperatures in the summer when the measurements took place.

Table 1. Indicative reported evapotranspiration (V_{ET}) rates from poplar (P) and willow (W) SRC stands in different countries (in Dimitriou et al., 2009; Busch, 2009).

Stand/ shoot age	Site	Soil	Species	Precipitation (mm)	V_{ET} (veg) (mm)	V_{ET} (year) (mm)	V_I (year) (mm)	Source	Country
9/9	Methau	loamy loess	P	690 (lta)	480			1	GER
3/3	Neuruppin	loamy sand	P	585		356	117	2	GER
9/9	Neuruppin	loamy sand	P	585		393	171	2	GER
Diverse	div	clay soil	W	700 (lta)		500	140	3	GB
8/8	Welzow	clay sand	P	749		404	138	4	GER
8/8	Welzow	clay sand	P	749		388	132	4	GER
2/2 to 5/2	Uppsala	loamy clay	W	352 precipitation +222 irrigation during vegetation period	435 (i, fer)	550 (e)	40	5	SE
3/3	Börringe	sandy loam		586 (lta)	360	439 (e)	30	6	SE
6/3	Alyckan	sandy loam	W	641 (lta)	440	550 (e)	40	6	SE
7/2	Brinkendal	sandy loam	W	641 (lta)	374	481	59	6	SE
x/2	Silsoe	Sandy clay loam	W	574 (lta)		441	125	7	GB
x/2	Selby	Sandy clay loam	W	643 (lta)		462	130	7	GB
x/2	Cirencester	Sandy clay loam	W	776 (lta)		594	140	7	GB

[1]Petzold et al., 2008; [2]Knur et al.,2007; [3]Hall, 2003; [4]Bungart et al., 2004; [5]Persson and Lindroth, 1994; [6]Persson, 1995; [7]Stephens et al., 2001. x - subsequent rotation period, e - extrapolated, lta - long-term average, I – Irrigated, Fert – Fertilized, Mean – mean value calculated from different sites, P – Poplar, W – Willow, V_{ET} – evapotranspiration, V_I – interception

The above estimations indicate that evapotranspiration levels from an SRC stand cannot be predicted with much certainty, since there is a series of factors that affect SRC evapotranspiration. However, a potential impact of SRC on the water use and balance in a certain area should be judged in comparison with the crops that will be replaced in a potential shift to SRC. Several authors report that evapotranspiration from SRC fields with willow and poplar is in most cases significantly higher than arable crops but lower than conventional forest (Persson, 1995; Stephens et al., 2001; Hall, 2003; Knur et al., 2007). Hall (2003) reported that on a clay site with 700 mm rainfall SRC is expected to use 600 mm compared to 400 mm for barley. Hall et al. (1998) indicated that in case of dry summers when there is a significant water deficit, the water use of poplar SRC will probably be considerably less than that of coniferous forest and closer to that of grassland. Therefore, the levels of water consumption of SRC in relation to other crops grown in the same area seem to depend on site-specific factors such as soil type, precipitation and others, and may vary from case to case. Although results obtained in Germany suggested that infiltration from poplar SRC fields was almost 3 times less than neighbouring agricultural fields (Knur et al., 2007), others suggest that such differences are significantly lower (Dimitriou et al., 2009; RELU, 2009) and that the water use by SRC in comparison to other crops largely depend on site-specific factors and on the methods chosen for calculation, and that the general perception that SRC “consumes” significantly more water than other crops should not be generalised to all cases. In this context, Finch (RELU, 2009) concluded from field experiments in Great Britain that water use of willow SRC is comparable to winter wheat and that the maximum rooting depth of willow SRC is about the same as for deep rooting conventional crops.

As a consequence of the higher evapotranspiration rates reported, assumptions concerning the effect of willow and poplar SRC were reported by Hall et al. (1998), Allen et al. (1999) and Perry et al. (2001) suggesting potential negative effects to water body enrichment from reduced percolation to groundwater due to willow and poplar SRC. In contrast, modeling exercises conducted by Stephens et al. (2001) indicated 10-15 % reductions of the hydrologically effective rainfall in SRC fields compared to arable crops in the UK. Despite this, the authors claimed that the effect on hydrology at the catchment level would be minimal, after extrapolations based on the model results obtained and the assumption that 2,500 ha SRC is planted within a 40 km radius from a biomass-driven power plant. This was due to the fact that the mean reduction in hydrologically effective rainfall for the catchment area would be c. 0.5 % of the mean annual amount, which would be only a very minor portion, compared to the respective effect of cereals. Hall (2003) also suggested that even if SRC “consumes” more water than conventional agricultural crops, catchment scale effects of SRC on hydrology will be negligible, and that even when used as riparian buffers SRC will have little effect on river or streams due to low abstraction rates. However, the author suggested that relatively high-yielding SRC plantations (above 12 t DM/ha/yr) should be avoided – as a precaution – in areas where precipitation is below 550 mm, since the consequences of reduced hydrologically effective rainfall can be much more serious in such areas.

SRC is generally considered to improve the water quality relative to conventional agricultural crops in a given area due to the management practices of SRC (weed control only during the establishment phase, tillage only before the establishment phase, and lower inorganic fertilization than other crops). Most of the studies for SRC concerning water quality have dealt with N and P leaching to groundwater since these elements are considered responsible for eutrophication in water bodies. Bergström and Johansson (1992) measured very low N concentrations (less than 1 mg/l N) in the groundwater of an intensively fertilized willow SRC field in southern Sweden, and similar results were obtained by Aronsson et al. (2000) for a period of eight years with average annual application rates of 112 kg N/ha. Close to zero N concentrations in drainage water from Danish SRC fields were reported by Mortensen et al. (1998) as well. Goodlass et al. (2007) reported high N concentrations in drainage water during the establishment and the termination phase of SRC in the UK, with reduction to low levels after the crop was established despite application of 200 kg N/ha in 3 years. In this study, comparisons with the maximum N concentrations in the drainage water from agricultural crops in the area were made, which consistently exceeded 60 mg/l every year. Large differences in the amounts of N leached in the groundwater between SRC and a series of agricultural crops were reported in Denmark by Jørgensen and Mortensen (2000). In a sandy soil, c. 15 kg N/ha were leached on average from fertilized willow SRC, whereas the respective value for different cereals was between 70 and 120 kg N/ha. The above reported differences in N leaching between SRC and other crops are rather striking but they could be attributed to the lower input of fertilizer applied to SRC. Therefore, it is useful to examine if SRC is equally good in N-leaching performance under circumstances with higher N amounts applied and when irrigation occurs. For such comparisons, results for SRC fields intensively irrigated with wastewater can be used. This method for treating and utilising nutrient-rich wastewaters (usually in N but also P) for irrigation has gained interest in recent years in countries where SRC cultivation is rather common (Dimitriou and Aronsson, 2005; Berndes et al., 2004). According to Aronsson (2000) after testing different irrigation regimes with wastewater under different soil conditions, wastewater application at least 150 kg N/ha yr should not pose any threat to extensive NO3-N leaching in Sweden. Concentrations of N in the drainage water below 5 mg/l were recorded in an experimental willow SRC field in northern Ireland where c. 200 kg N/ha/yr was applied (Werner and McCracken, 2008). Moreover, Sugiura et al. (2008) applied much higher amounts (ca. 300 kg N/ha/yr) and N concentrations in the drainage water at different depths were between 5 to 10 mg/l. This figure is rather low considering the high application rate compared with findings for other agricultural crops. The above findings suggest that in general leaching of N from SRC in comparison with agricultural crops is significantly lower and a shift from conventional crops to SRC will probably imply an improvement of the groundwater quality and consequently of the surface water quality in a certain area, even when N fertilization exceeds the recommendations for good practice.

Concerning the impact of SRC on P leaching when large P amounts are applied, results from experiments involving applications of municipal sewage sludge to willow and poplars are relevant, since sludge application to SRC is a common practice in Sweden and in the UK that compensates P losses in newly harvested fields (Sagoo, 2004; Dimitriou and Aronsson, 2005). The application rates of P when sewage sludge is applied to SRC are rather high (e.g. in Sweden 22 or 35 kg P/ha/yr depending on the P content existing in the soil, for a 7-year period), and from the fact that willows can accumulate c. 8 kg P/ha/yr in the stems depending on the site, a surplus of P is usually applied with sewage sludge. Despite this, P is usually bound to soil particles and its leaching patterns differ from those of N which is in most cases related to water drainage. Results by Dimitriou and Aronsson (in press, 2010) on sewage sludge-amended lysimeters confirm the above speculation since P concentrations in the drainage water were close to zero throughout measurements during two growing seasons. Even when P was applied to willow and poplar plants with wastewater in rather high amounts (c. 29 kg/ha/yr, higher than the allowed amounts), P-retention was up 97% and 94% for the two species, respectively, during a 2-year experiment. However, future P leaching cannot be excluded as a possible scenario when sewage sludge is applied for a number of years at high rates.

Figure 3. Pictures with SRC related to water issues. Top left: Willow SRC planted between agricultural land and water course as a buffer; Top right: The wastewater treatment plant in Enköping, Sweden, where wastewater is applied to a 76-ha willow SRC field as a step in the treatment; Bottom left: Drip irrigation pipes in the above-described field; Bottom right: Irrigated poplar plantation in Chile.

All the above about water quality and SRC indicate that when SRC replaces conventional agriculture crops, groundwater quality improvement is anticipated. In fact, several authors suggest the use of SRC in intensively managed agricultural areas to improve the current water quality and meet EU obligations in terms of water quality expressed in the Water Framework

Directive (Jørgensen and Mortensen, 2000; Eppler et al., 2007; EEA, 2008) and simultaneously use the land for agricultural production for biomass for energy that also fulfils other obligations concerning renewable energy.

To promote future decision-making processes with respect to the envisaged expansion of SRC on productive and marginal soils, the potential local impact of SRC on water needs to be extrapolated to a larger scale. This can be assessed within the framework of existing ecological and spatial planning as a tool for rapid qualitative assessments. Such an effort is conducted by Busch (2009). Applying GIS analyses, the author calculated the ecological impact as a function of groundwater recharge for different SRC water use boundaries. Based on this regression, he qualitatively assessed the impact of SRC on groundwater recharge for two municipalities in the district of Uelzen, in northern Germany. As a result of this spatial-planning-oriented approach, it turned out that only SRC fields representing the lower water use boundary show a minor impact on groundwater recharge. “Minor”, in this context, means that the water use of this SRC type is equivalent to irrigated crops – a common practice in these municipalities. Further, the approach reveals the crucial importance of a sufficient available soil water capacity (AWC) if annual precipitation is lower than 600 mm. To reduce the ecological and economic risk of SRC, these four parameters are essential: (1) clone/species-specific water demand for a different rotation management (2) annual precipitation (3) precipitation during the vegetation period (4) AWC.

Better and more precise data concerning SRC water use are needed to reduce the uncertainty of an ecological impact assessment of SRC on water issues. Such impact assessments and ecological evaluations of landscape functions need to carefully consider site-specific conditions (e.g. soil type, species, climatic parameters etc.) as well as existing environmental and economic goals.

SRC IMPACT ON SOIL

The impact of SRC on soil affects C sequestration, nutrient cycling from litter and soil microorganisms. The phytoremediation ability of SRC species also depends on soil impact. Nair et al. (2009) have reported that C sequestration in arable soils depends on a number of site-specific biological, climatic, soil and management factors. Reported values for total C sequestration under SRC are significantly higher than under arable soils with annual crops, but still below the C sequestration in mature forests (Bowman and Turnbull, 1997). The C turnover under SRC grown on agricultural soils previously cultivated with conventional crops is more similar to that in forests than in arable soils (Svensson et al., 1994), but it is likely that C sequestration varies significantly between tree genotypes even within one genus as it was reported for different willow clones by Weih and van Bussel (2006). The C accumulation after conversion of fields with conventional arable crops to SRC is reported to be concentrated in the topsoil (Makeschin, 1994; Stetter and Makeschin, 1997; Neergaard et al., 2002; Dowell et al., 2009). Estimated rates of C accumulation in topsoil, e.g. 0-40 cm, of arable sites were 40-170 g C/m2/yr during the first decade following SRC establishment (Garten, 2002). The increased C concentrations in SRC soils can be explained by the lack of tillage in SRC and the high annual amounts of leaf litter deposited on the soil surface (average 1 to 5 t/ha) (Verwijst and Makeschin, 1996; Bowman and Turnbull, 1997) and by the increased transfer of assimilates into external mycelium of mycorrhizal fungi (Ek, 1997). The external mycelium of mycorrhizal fungi was the dominant pathway (62 %) through which C entered the soil organic matter (SOM) pool; this input exceeds the litter and fine root turnover under poplar SRC (Godbold et al., 2006). However, it is the annual leaf litter fall that is the main source of easily-available C for the soil microorganisms as derived from hot water extracts (Huang and Schoenau, 1996). Gordan and Matthews (2006) predicted that the potential for C sequestration under SRC with willow is largest in soils in which the C content has been depleted to relatively low levels due to aeration by annual deep ploughing. In cases where soil C is initially high, such as in improved pasture land, and SRC is introduced, soil C losses have been reported at least in the initial years of SRC cultivation (Cowie et al., 2006). To sum-up, the above-mentioned reports indicate that in general increased C sequestration when SRC is grown on agricultural soils previously grown with conventional crops is anticipated, however, the initial soil properties are responsible for the extent of C storage. Therefore, it would be useful to develop approaches for the selection of the most promising sites for C sequestration by SRC, in combination and taking into account other potential environmental problems in a specific area. Furthermore, it is almost completely unknown which chemical alterations the soil organic matter undergoes when SRC stands are grown for several years on fields previously grown with agricultural crops, and therefore such basic understanding is urgently needed for science-based predictions of C-sequestration opportunities in SRC.

Differences in the nutrient turnover when SRC is grown on fields previously cultivated with common agricultural crops are expected. The soil nutrient turnover is affected by SRC biomass, SRC rhizodeposits and management. Litter quality and decomposition rates are affected by SRC species and soil types (Ericsson, 1981; Püttsepp et al., 2007; Rytter, 2001). Meiresonne et al. (2006) investigated the hydrological fluxes, atmospheric deposition, litterfall, and soil percolation of the most important nutrients in an 18-year-old poplar plantation on a well-drained silt loam soil during 2 years. In this study around 80 % of total nitrogen input (6.6 kmol/ha in years 1 and 6.5 kmol/ha in year 2) originated from litterfall; after nitrification only a negligible amount of nitrate leached during the growing season. The yearly uptake of N by the poplar ecosystem in this study was equal to the input, of which more than 50 % was accounted for by the leaves. This indicated very efficient N cycling. Total deposition of base cations originated from two processes, dry deposition (Mg^{2+} and Ca^{2+}) and canopy leaching (K^+ and Ca^{2+}). Litter input of Ca^{2+} represented about 83 % of the total input (stand deposition + litterfall), Mg^{2+} about 61 %, and K^+ less than 50 %. Percolation of base cations at 1 m depth was very limited. Rather high Ca^{2+} and K^+ contents of the woody biomass can lead to high exports at harvest (relative to the amount in the soil). Meiresonne et al. (2006) concluded that the nutrient cycling in the poplar stand was very efficient, with no significant nutrient losses.

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

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

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

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

Mycorrhizal colonization of Populus and Salix varies greatly between species and genotypes (Khasa et al., 2002; Püttsepp et al., 2004), and also depends on soil properties and management effects (Loree et al., 1989; Baum and Makeschin, 2000; Baum et al., 2002). In intensively managed biomass plantations, the degree of mycorrhizal colonization and the diversity of ectomycorrhizal fungi appear to be lower compared to adjacent natural stands (Toljander et al., 2006). Furthermore, mycorrhizal colonization affects the leaf chemistry of willows (Baum et al., 2009), which, in turn, may influence willow leaf resistance to pests such as herbivorous insects. The effect of mycorrhiza on leaf chemistry varied between host plant genotypes (Baum et al., 2009). The interaction of plant genotype and herbivory can affect the leaf litter decomposition and alter the nutrient dynamics

(Schweitzer et al., 2005). This example shows that willow genotype directly affects soil ecology in terms of mycorrhizal colonization, which could indirectly influence crop safety through an effect on pest resistance. Understanding these complex multi-trophic interactions is crucial to our general understanding of soil ecosystem function and the regulation of fundamental ecosystem processes. Furthermore, understanding of multi-trophic interactions could also support yield increases and crop safety in willow and poplar biomass plantations in a most sustainable way, e.g. by the appropriate choice of willow varieties that favour pest resistance.

Poplars and willows are used for phytoremediation to improve/clean soil from hazardous compounds such as heavy metals or organics, based on the function of the plants against hazardous compounds via different processes (Glass, 1999). Although willows and poplars are not metal hyperaccumulators of hazardous compounds, they are preferred in commercial phytoremediation projects due to their fast and high growth, and because existing agronomic management practices for SRC ensure good and fast growth. Willows and poplars evapotranspire high amounts of water (Persson and Lindroth, 1994) and tolerate high heavy metal concentrations in soil (Hammer et al., 2003; Laureysens et al., 2004). Furthermore, willows are tolerant to anoxic conditions (Jackson and Attwood, 1996). All the above traits enable growth in harsh environments, but as SRC is seen as a biomass production system, conditions promoting high productivity are preferred. In many cases however, soils moderately contaminated with heavy metals from previous uses of inorganic fertilizers or sewage sludge are available for SRC. In this report we consider the use of large-scale SRC cultivation systems for phytoremediation related to phytoextraction since the hazardous compounds are removed with the harvest.

Since the early stages of their commercial bioenergy use, willows have been reported to take up large amounts of Cd (Perttu, 1992; Riddell-Black, 1994). Initially most research was done on Cd uptake by willows; later the uptake of other metals together with Cd such as Cu, Pb, Zn, Cr, Ni, As was also studied (Granel et al., 2002; Kuzovkina et al., 2004; Meers et al., 2007). Metal uptake by poplars was studied at later stages once poplar gained interest as an alternative to willow for biomass production for energy (Robinson et al., 2000; Sebastiani et al., 2004; Licht and Isebrands, 2005). The phytoremediation potential of willows and poplars has been reported to be high based on the combination of high accumulation of metals in the plant tissues together with the high biomass produced (see Table 2). Related research has been conducted in controlled laboratory conditions where individual willow and poplar plants were grown in contaminated soils (Landberg and Greger, 1996; Vandecasteele et al, 2002; Fischerova et al, 2006; Wieshammer et al., 2007) or in hydroponic systems (Kuzovkina et al., 2004; Dos Santos Utmazian and Wenzel, 2007). Very promising results for uptake of certain metals by willow and poplar plant parts have been reported from those experiments, leading to speculation that willow and poplar offer great potential for cleaning contaminated soils. Concerns due to differences between controlled small-scale experiments and large-scale field situations have been raised (Dickinson and Pulford, 2005), however results from pot trials have been validated in some cases in the field (Robinson et al., 2000; Sebastiani et al., 2004). Many studies have also proposed the use of a range of chelating agents such as ethylendiamin-tetraacetat (EDTA), ethylenediamine-N,N'-disuccinic acid (EDDS), oxalic and citric acids, and others, to increase the positive metal uptake rates by willow and poplar plants (Hooda et al., 1997; Robinson et al., 2000; Schmidt, 2003; Hammer and Keller, 2002; Komarek et al., 2008). Despite the positive results for induced phytoextraction indicated in the previous publications, chelating agents have been reported to cause toxicity symptoms in the plants, leaching of metals and negative impact on soil biota, casting doubt on the potential future use of chelate-assisted phytoextraction (Evangelou et al., 2007; Dickinson et al., 2009). Another opportunity for the improvement of phytoextraction by willows is the inoculation with mycorrhizal fungi and bacteria (Baum et al., 2006; Kuffner et al., 2008;

Zimmer et al., 2009). The potential of these biologically based improvements of phytoextraction efficiency is just beginning to be explored.

Some critical points must be considered for successful phytoextraction with SRC. Great variations in metal uptake ability of willows and poplar have been reported in different SRC fields. This might depend on different contamination levels within the fields, and/or differences in the clone material used. Vandecasteele et al. (2002) suggested that Cd uptake in above-ground plant parts tends to increase with increasing Cd in soil. This was also reported in other studies with elevated metal concentrations where willows and poplars took up larger amounts of heavy metals in aboveground tissues (Hammer et al., 2003; Rosselli et al., 2003; Unterbrunner et al., 2007) than in less contaminated soils (Pulford et al., 2002; Klang-Westin and Eriksson, 2003; Dimitriou et al., 2006). Moreover, even spatial variability of contamination within one field might be responsible for great variations in metal uptake (Dickinson and Pulford, 2005). Differences in metal uptake by willow species and clones have been reported by several authors (Landberg and Greger, 1996; Vyslouzilova et al., 2003; Kuzovkina et al., 2004), and Laureysens et al. (2004) reported great differences in the ability of poplar clones to take up metals. Therefore, much attention should be paid to clonal selections in relation to the contamination source and level at the site. However, Dickinson et al. (2009) suggested that predictable uptake patterns for all metals will be unlikely to be found for accumulation in aboveground biomass, and only genotypes that take up more mobile elements such as Cd and Zn can be selected for a specific site. This was based on reported differences in the metal uptake among different families, species, clones, and within individual plants. The mobility and plant availability of metals in soil might be also responsible for the great differences in uptake patterns. For example, Eriksson and Ledin (1999) indicated that plant available Cd concentrations in soil were reduced in a willow SRC field, but higher uptake of different metals in willow shoots were not found when plant available fractions differed due to pH changes in a field willow experiment (Dimitriou et al., 2006). It seems that for cleaning soils a “site-specific” approach with pre-testing of several clones to identify the best performing ones for further use at large-scales should be performed in advance, although difficulties due to the heterogeneity of localization of the pollution are to be expected (Keller et al., 2003). The above raises the question which soils can be satisfactorily remediated by phytoextraction of heavy metals with willow and poplar SRC, and what strategies should be followed to achieve the best remediation combined with the best economic value in a certain time frame. Although willow and poplar have been shown to have equally good or better phytoextraction efficiency than other species (Rosselli et al., 2003; Fischerova et al., 2006), recent studies suggested that short-term remediation is not to be expected in heavily contaminated soils such as mine spoils or heavily contaminated industrial sites due to long time needed (Dickinson and Pulford, 2005). Furthermore, such sites might be polluted in deep layers which cannot be cleaned with poplars and willows that are more appropriate for rather shallow contamination (Keller et al., 2003) since most of their active roots are concentrated near the soil surface (Rytter, 2001). However, large-scale SRC cultivation offers great potential for removing metals such as Cd, Zn, Cu, Ni, and Se from moderately contaminated soil (Dickinson et al., 2009). An example would be agricultural soils with a legacy of Cd from P fertilization and metal contamination from sludge applications. Berndes et al. (2004) calculated that 100 times more Cd would be removed by willow SRC than harvested by straw in Sweden if SRC will be grown on agricultural soils with elevated Cd concentrations from phosphate fertilizer. These amounts would compensate for the atmospheric deposition each year and would drastically reduce the amount of Cd in agricultural soils in Sweden, and would give economic incentives for the farmer for reducing Cd in the soil (ca. 10 % of total revenue). Similar calculations were made by Lewandowski et al. (2006), suggesting that phytoextraction with willows cultivation for a certain period can allow the future use of moderately contaminated fields for more profitable food production, thus increasing farmers´ income.

Figure 4. Top left: Poplar plantation (2-year old shoots on 16-year old roots) and accumulation of C

in top soil; Top right: Sewage sludge application to willow SRC fields in Sweden; Bottom left: Willow SRC fields located close to other agricultural crops; Bottom right: Landscape restoration and soil/wind erosion with willow SRC in peat fields.

Table 2. Concentrations of heavy metals in the biomass of Salix and Populus at contaminated soils (Baum et al., 2009).

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

SRC IMPACT ON BIODIVERSITY (PHYTODIVERSITY, ANIMAL DIVERSITY)

1) Influences on phytodiversity in SRC

Fast-growing tree species like poplar and willow are grown in a short rotation coppice (SRC) system to fulfill the increasing demand for wood as a renewable energy source. The ground vegetation of SRC plantations is influenced by many factors, which are discussed below.

Site preparation has to be carried out for successful establishment of willows and poplars. It is common to plough and to harrow the soil as in conventional agriculture. A broad-spectrum herbicide is applied after ploughing in autumn (Boelcke, 2006; Burger et al., 2005; Fry and Slater, 2009; Schildbach et al., 2009). Before planting in spring the field is grubbed (Schildbach et al., 2009), ploughed (Burger et al., 2005) or harrowed (Boelcke, 2006; Burger et al., 2005), whereas the application of a pre-emergence herbicide is recommended (Boelcke, 2006; Burger et al., 2005; Fry and Slater, 2009). These treatments control competing vegetation to ensure rapid establishment of the SRC plantation. Fertilization is not normally necessary because there are enough nutrients available from the former usage. Due to harvesting in winter, the majority of the nutrients remain in the fields (Boelcke, 2006, Schildbach et al., 2009).

Light-intensity has a great influence on species composition. It depends on the stage of canopy closure and therefore on the age of the planted crop as well as on the planted tree species. Light and nutrient-demanding species, especially annual ones, are typical for young plantations (Delarze and Ciardo, 2002), whereas with increasing plantation age there is a shift toward more shade-tolerant, perennial species typical of forests (Britt et al., 2007; Delarze and Ciardo, 2002; Kroiher et al., 2008, DTI, 2004; DTI, 2006). During the first two years after establishing an SRC plantation, the species number increases and thereafter decreases with increasing age of the plantation (Delarze and Ciardo, 2002; DTI, 2004; Gustafsson, 1987; Wolf and Böhnisch, 2004).

Normally, only few species with regional conservation status are found in the plantations (Britt et al., 2007; DTI, 2006; Gustafsson, 1987; Vonk, 2008; Weih et al., 2003), which are dominated by ruderal species like nettle, thistle and grasses (Gustafsson, 1987; Britt et al., 2007). The recorded endangered species are mainly light-demanding pioneer species which occur in the first year of the plantation and disappear with increasing age (Delarze and Ciardo, 2002). Due to a positive edge effect, which is expressed by higher ground vegetation cover and species numbers at the edge than within the plantation, the size and shape of the plantations seem to be important for plant diversity (Augustson et al., 2006; DTI, 2004; DTI, 2006; Gustafsson, 1987; Weih et al., 2003). Small-sized plantations are suggested to favor species diversity (Gustafsson, 1987; Weih, 2008a). The plant colonization of a plantation takes place from the surrounding area, the soil seed bank and through living vegetative tissues like rhizomes, tillers or living roots in the soil (Gustafsson, 1987; Stjernquist, 1994; Weih, 2008). The soil seed bank is greatly influenced by the former vegetation and therefore by the former use. This influence decreases with increasing age of the SRC, whereas the magnitude and temporal development of the changes differ between land uses (Gustafsson, 1987). Studies on edge effect suggest that plantation colonization occurs predominantly from the surrounding landscape. The more diverse the surrounding, the more opportunity the species have to reach the SRC and become established there (Weih, 2008). More species were recorded in willow and poplar plantations than in conventional agricultural fields (Augustson et al., 2006; Britt et al., 2007; Burger et al., 2005; DTI, 2004; DTI, 2006; Fry and Slater, 2009; Heilmann et al., 1995; Perttu, 1998; Wolf and Böhnisch, 2004). Species richness of young poplar plantations was similar or lower in comparison to old-growth mixed deciduous forests (Weih et al., 2003). To conclude, SRC plantations can have positive as well as negative effects on phytodiversity. Therefore, economical and environmental aspects should be considered carefully. The location for an SRC should be chosen very carefully, and areas dominated by agriculture or coniferous forests might be preferred depending on environmental and economic objectives, and areas in need of protection like wetlands or peat bogs should be avoided. The establishment of small-structured plantations with different species and different rotation times is advisable to enhance structural, and therefore biological, diversity. Lastly, in case that chemical and fertilization treatments do not compensate for drastic profit increases, they should be reduced or avoided.

2) SRC impact on animal diversity

In the overall diversity of animals in an SRC, as in other ecosystems, vertebrates make up only a small fraction. Up to now, very little research has been conducted on mammals in SRC; most research has been conducted on the diversity of birds in SRCs. Tangible data and meaningful overviews from Great Britain (Sage & Robertson, 1996; Sage et al., 2006; Anderson et al., 2004), Sweden (Berg, 2002), Germany (Jedicke, 1995; Liesebach and Mulsow, 1995, 2003; Gruß and Schulz, 2008) and the USA (Christian et al., 1997; Christian et al., 1998; Dhondt and Sydenstricker, 2000; Dhondt et al., 2004; Dhondt et al., 2007) already exist on this subject. The cited number of bird species in SRC varied from 8 to 60 species. Different bird species are associated with different age classes of SRC (Gruß & Schulz, 2008). The abundance of birds in SRC has been shown to be linked with age, with coppice stem or planting density and with increased weediness (Sage et al., 2006; Gruß and Schulz, 2008). But the different numbers of species are due to many other factors, such as variety of areal sizes, management intensities, landscape context and regional species pool (Schulz et al., 2009).

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

Poplars and willows act as host to a large number of insects. For example beetle species belonging to the Phratora and Chrysomela genera can cause major damage (Helbig and Müller, 2008 and 2009). Due to the overwhelming diversity of invertebrates, investigations have been limited to individual indicator groups. Up to now, earthworms (Lumbricidae) (Makeschin et al., 1989; Makeschin, 1994), web-spinning spiders (Blick and Burger, 2002; Blick et al., 2003) and butterflies (Britt et al., 2007; Haughton et al., 2009) have been investigated in SRCs. The invertebrate group studied in greatest detail in SRC is ground beetles (Carabidae) (Allegro and Sciaky, 2003; Liesebach and Mecke, 2003; Britt et al., 2007; Schulz et al., 2008a; Schulz et al., 2008b; Lamersdorf et al., 2008). Species numbers ranging from 10 to 43 were discovered on SRC. These numbers are of little significance, however, if they are not related to adjacent habitats and to the various influencing factors. Haughton et al. (2009) summarize that compared with cultivated areas of energy crops such as oilseed rape, SRCs have particular advantages as bioenergy sources: there is no annual cultivation cycle, they achieve rapid growth with the potential to produce large yields with low fertilizer and pesticide requirements, there are only a few disturbances in the growing period, harvesting is carried out in winter and therefore causes less disturbance, and there is a greater richness of spatial structures. This has an overall positive effect on the animal diversity. Animals that depend heavily on the vertical structure, such as many breeding birds, can benefit from the growth characteristics of SRC. This is particularly true when SRCs are planted in an agricultural landscape with little structural diversity. Many insect groups benefit from the decreased use of pesticides in SRCs and earthworms, e.g., are favored by the longer soil rest period (Makeschin, 1989; Makeschin, 1994). Liesebach et al. (2000) demonstrate that a higher diversity of epigeal invertebrates is present in an SRC than in a barley field. Britt et al. (2007) found a greater abundance and diversity of butterflies (Lepidoptera) and a higher number of springtail species (Collembola) in hybrid poplar fields than in agricultural fields. Regarding arachnids, Blick & Burger (2002) and Blick et al. (2003) found more individuals and species of arachnids in German SRCs than on nearby agricultural crop land.

Figure 5. Top left: Vegetation in a willow SRC field in between a double row; Top right: Vegetation in between two sections of a willow SRC field; Bottom left: Nest of Turdus philomelos in willow SRC in Georgenhof, Germany; Bottom right: Pieris rapae on willow SRC at Jamikow, Germany, feeding on Echium vulgare.

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

Ground beetles (Carabidae) are species-poorer in some SRCs than on agricultural crop land. Britt et al. (2007) found significantly more ground beetle species in agricultural fields than in poplars on English sites. Also, fewer species of ground beetle were found in various northern German SRCs than on neighboring intensively farmed agricultural crop land (Liesebach and Mecke, 2003; Lamersdorf et al., 2008; Brauner and Schulz, 2010).

The animal diversity on SRCs depends on various environmental factors as will be shown with the following examples. Concerning the impact of age, it should be mentioned that poplar and willow plantations change very quickly due to their rapid increase in height. Thus, the habitat conditions relevant to animals such as spatial structure, structural density, complexity of vegetation, shade and humidity also change. Berg (2002) measured increasing bird species numbers with an increasing height of Salix plantations. Different bird species are associated with different age classes of SRCs (Sage et al., 2006; Jedicke, 1995; Gruß and Schulz, 2008). According to Christian et al. (1998) and Londo et al. (2005), three phases of an SRC can be identified: open area phase, shrub-like stands and tree-like forms. In the breeding bird studies carried out by Dhondt and Sydenstricker (2000), Berg (2002), Sage et al. (2006) and Gruß and Schulz (2008, 2009), the highest numbers of species and the greatest breeding density were found in the second, shrub-like development phase.

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

Concerning the clone choice, Dhondt and Sydenstricker (2000) found 41 % of the nests in the poplar clone S365, but only 24 % in the poplar clone NM6. The choice of nesting site appears to be influenced by the branching pattern of the respective clone. To increase the attractiveness for several breeding birds, Dhondt et al. (2004) therefore recommend a mix of different clones when establishing large-scale SRC plantations and not planting clones such as S301, as these are less preferred by breeding birds.

About the role of plantation size, Christian et al. (1998), Cunningham et al. (2004), Sage et al. (2006) and Gruß and Schulz (2008) concluded that significantly more bird species with a higher concentration of individuals populate the periphery of the SRC and that the most obvious effect of plantation size on biodiversity is the higher proportion of edge habitat in small plantations. On large plantations, lower overall bird densities were observed in plantation interiors than on edges (Christian et al., 1998).

The biodiversity of an SRC is influenced to a large degree by the surrounding landscape. Berg (2002) emphasized the strong influence of adjacent habitats on bird community composition in the SRCs. He found major differences between bird communities depending on whether the SRC bordered on woodland or open land, for example. On the other hand, SRCs affect the biodiversity of the surrounding landscape. Planting SRCs has a positive effect on the biodiversity in cleared landscapes, but a negative effect in valuable open countryside (Sage et al., 2006; Schulz et al., 2009).

Concerning the impact of accompanying structures, structurally rich blocks of trees and heterogeneously composed SRCs increase the diversity and the density of breeding birds (Berg, 2002; Sage et al., 2006; Gruß and Schulz, 2008). In particular, the diversity of vertebrates and invertebrates in SRCs can be greatly increased by accompanying structures in boundary and internal border areas.

To conclude, it cannot be stated generally that SRCs have a positive effect on animal diversity. Instead, one has to differentiate among animal groups, spatial structures and ecological conditions of SRCs. In addition, the respective landscapes concerned need to be considered and certain influencing factors such as age and form of the area taken into account.

CONCLUSIONS

In order to achieve maximum positive effects and minimize potential negative effects from large-scale SRC cultivation on agricultural soils to produce biomass for energy, proper site selection and management adjustments should be implemented taking into account the research results related to each of the aspects affected by SRC cultivation. However, such management “modifications” and the sustainable production of biomass from SRC, keeping in mind that SRC is a commercial crop for production of biomass for energy, competing with high value agricultural crops. Balancing maximum environmental benefits and maximum attained biomass production from SRC is a large challenge that all stakeholders involved in SRC cultivation (farmers, decision-makers, researchers, and others) must consider. Despite all the expected positive environmental impacts of SRC, farmers need to be convinced to grow the crop; typically, this is achieved when the economic profit from the cultivation of a new crop such as SRC is equal to or higher than that of other “established” or “conventional” crops. Decision-makers may consider various direct or indirect incentives for farmers, to encourage shifts in land use from conventional crops to SRC in areas where this would result in environmental benefits. For instance, a potential economic compensation could be a form of “reward” to farmers helping to fulfill set environmental goals, while keeping agricultural land in production. A prerequisite for such incentives is, however, science based methods for quantification of the environmental benefits of shifting to SRC cultivation and evaluation of the value of these benefits for society. Identification of opportunities for SRC cultivation to contribute to environmental objectives, while providing biomass for the production of biomaterials and solid/liquid/gaseous biofuels, can help promoting SRC as a valuable component in future sustainable land use systems.

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IEA Bioenergy

IEA Bioenergy is an international collaboration set up in 1978 by the IEA to improve international co-operation and information exchange between national RD&D bioenergy programmes. IEA Bioenergy's vision is to achieve a substantial bioenergy contribution to future global energy demands by accelerating the production and use of environmentally sound, socially accepted and cost-competitive bioenergy on a sustainable basis, thus providing increased security of supply whilst reducing greenhouse gas emissions from energy use. Currently IEA Bioenergy has 22 Members and is operating on the basis of 13 Tasks covering all aspects of the bioenergy chain, from resource to the supply of energy services to the consumer.

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