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

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## Abstract

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

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

## Zusammenfassung

**Einfluss von Kurzumtrieb mit Weiden und Pappeln auf die Bodenökologie**

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

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

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## 1 Introduction

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

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

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

## 2 Carbon sequestration in the soil

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

Table 1:

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

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

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

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

## 3 Nutrient cycling from litter

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

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

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

## 4 Effects of SRC on soil organisms

### *4.1 Soil microorganisms*

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

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

Table 2:
Ectomycorrhizal fungi on *Salix* and *Populus* spp.

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

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

*4.2 Soil fauna*

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

## 5 Applications for phytoremediation of contaminated soils

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

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

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

### *5.1 Phytoextraction of heavy metals*

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

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

Table 3:

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

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

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

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

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

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

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

### *5.2 SRC and rhizodegradation of organic pollution*

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

Table 4:

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

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

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

## 6 Concluding remarks

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

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