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Assessment of suitability of tree species for the production of biomass on trace element contaminated soils

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ABSTRACT

To alleviate the demand on fertile agricultural land for production of bioenergy, we investigated the possibility of producing biomass for bioenergy on trace element (TE) contaminated land. Soil samples and plant tissues (leaves, wood and bark) of adult willow (*Salix* sp.), poplar (*Populus* sp.), and birch (*Betula pendula*) trees were collected from five contaminated sites in France and Germany and analysed for Zn, Cd, Pb, Cu, Ca, and K. Cadmium concentration in tree leaves were correlated with tree species, whereas Zn concentration in leaves was site correlated. Birch revealed significantly lower leaf Cd concentrations (1.2–8.9 mg kg^{-1}) than willow and poplar (5–80 mg kg^{-1}), thus posing the lowest risk for TE contamination of surrounding areas. Birch displayed the lowest bark concentrations for Ca (2300–6200 mg kg^{-1}) and K (320–1250 mg kg^{-1}), indicating that it would be the most suitable tree species for fuel production, as high concentrations of K and Ca decrease the ash melting point which results in a reduced plant lifetime. Due to higher TE concentrations in bark compared to wood a small bark proportion in relation to the trunk is desirable. In general the bark proportion was reduced with the tree age. In summary, birch was amongst the investigated species the most suitable for biomass production on TE contaminated land.

1. Introduction

Cellulosic biomass can be used to produce heat and electricity through direct combustion, or it can be transformed through pyrolysis and gasification into bio-oil and fuel gas [1]. Compared to grain, the efficiency of energy production from cellulosic biomass is favourable because of lower energetic inputs to their feedstock production [2]. In the temperate climates of North America and Europe cellulosic biomass used for bioenergy production mainly derives from short-rotation willow (*Salix* spp.) and poplar (*Populus* spp.) coppice cultures. Some countries, such as the United States and the EU, heavily subsidise the production of crops for energy. As a result the use of vast swathes of land has changed from food crop to energy crop production, leading to increased food prices.

An alternative to the use of high value agricultural farmland for biomass production could be the use of contaminated land. In Europe approximately 10 million ha [3–7] have been contaminated by TE from either anthropogenic or natural sources. In the USA the volume of soil contaminated with radionuclides and/or TE within the U.S. Department of Energy complex is estimated to exceed 200 million m^3 [8]. Owing to the lack of funds, only a minority of these areas can be remediated [9]. Lal and Pimentel [10] proposed that bioenergy plantations could be established on land of marginal agricultural suitability, as well as degraded or drastically disturbed soils. Restoring degraded soils through establishing biofuel/biomass plantations would be a win–win strategy.

Consequently, in this study we investigate the possibility of "biomass production on contaminated land" (BCL), using trace element (TE) contaminated land on which food cannot be safely produced. From such an undertaking, risk can occur owing to the fate of TE. With regards to the fate of TE, many processes have to be considered in BCL. These can be grouped into two categories: (a) plant TE uptake from soil and TE deposition through leaves or dissolved organic carbon (DOC) from decaying leaves; (b) fate of TE during the energetically valorising process (biochar, biofuels, biogas) and after the process (use of char and ash). Category (a) describes the risk on the BCL site, which could result for example in contamination of surrounding areas and contamination of groundwater resulting in entrance of TE in the food chain. As plant litter might be dispersed via wind or water erosion, high TE concentrations in the leaves may lead to contamination of adjacent environments [11]. Besides the risks that could occur during the energetically valorising process, category (b) includes

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also the influence that metals could have on the process itself. Inorganic compounds within biomass promote the formation of gaseous species and char at the expense of bio-oil yield [12]. This would be a disadvantage for the production of biofuels, but an advantage when the production of biochar is the aim. Contaminated areas could be of great use for biochar production and its associated potential for carbon credits. In respect to the size of the possible market, biochar production could make a profitable alternative to bioenergy. For BCL to return an economic profit whilst minimising the environmental risks posed by the contaminants, the trees used for BCL should combine, a high biomass production for a high economic income with high evapotranspiration to reduce TE leaching and low TE uptake, to reduce TE transfer to the food chain [13]. Therefore, in a screening program we collected soil samples and plant tissues (leaves, wood and bark) of adult willow (*Salix* spp.), poplar (*Populus* spp.), and birch (*Betula pendula*) species from five contaminated sites and analysed them for Zn, Cd, Pb, Cu, Ca, and K. We investigated, *Salix* spp. and *Populus* spp. as these two tree species are used for biomass production and phytostabilisation projects and *B. pendula* as it is a pioneer plant. Pioneer plants are characterised by fast growth and low demands on the soil they are growing on (low nutrient demands, wide pH range, etc.) and are therefore possibly suitable for BCL.

In this study we (a) evaluated TE uptake by adult trees on TE contaminated areas, (b) assessed the partitioning of TE between wood and bark, (c) evaluated the influence of bark proportion to metal concentration of tree trunks and (d) discuss possible implications for BCL.

2. Materials and methods

2.1. Plant and soil sampling

We selected four sites in North France (A1, M, P, Auby), near the Zn and Pb smelters of Auby (Umicore) and Noyelles–Godault (Metaleurop Nord) and one in South Germany (F), near the Zn smelter of Kappel in Freiburg. The smelter in Freiburg was closed in 1954, whilst that of Noyelles–Godault was shut down in 2003. The smelter of Auby is still operational; however its pyrometallurgic process was changed to an electrolytic one in 1975, thus reducing atmospheric emissions considerably [14]. The site A1 was situated 600 m north-east, downwind of the smelter at Noyelles–Godault. In November 1992, this site was planted with *Quercus robur*, *Alnus glutinosa*, *B. pendula*, *Robinia pseudoacacia* and *S. alba*. The site P (750 m north-west of the smelter), was planted with the same five tree species, in addition to *P. trichocarpa*, in 1997. Site M, located about 1 km east of the smelter at Noyelles–Godault was planted with *B. pendula* and *S. alba* in 1983. On the Auby site (350–500 m from the smelter), hybrid poplars (*P. euramericana*) were planted in 1974 (P1–P4) and *B. pendula* (B1–B7) started to grow in the same period. Due to the severe contamination of soil, some of them died. A second period of poplar plantation was therefore held in 1983 (P5–P10), along with the development of spontaneous species adapted to polluted sites: *B. pendula* (B8–B10) and *S. caprea* (S1–S5). Hyperaccumulators were detected only on the Auby site. The trees were planted for the purpose of soil stabilisation and to prevent cultivation of the contaminated soil. The site in South Germany was covered by birch trees, which had established themselves spontaneously from seeds of birches in the surroundings of this site.

Soil samples were taken from all sites at a distance of 30 cm from the investigated tree from the upper rooting zone (typically 0–20 cm). The trees were selected due to their accessibility to the tree canopy and their trunk diameter (20 ± 5 cm). For the poplars on the site of Auby the trunk diameter of the poplars was 24 ± 5 cm and for the birches in Freibug it was 23 ± 6 cm. Plant samples were collected in October 2009 in order to investigate TE accumulation. Leaves were collected randomly from the selected trees (approximately 50 g dry weight) and bulked to a homogeneous sample. Wood and bark samples were taken at a height of 1.5 m above ground using a wood-borer and a stainless knife. Plant samples were subsequently washed with tap and deionised water and dried at 60 °C until constant dry weight was achieved. The diameter was determined through the measurement of the trunk thickness at several positions around the trunk at the height of 1.5 m and also calculated from the trunks circumference. The values were averaged. The numbers of specimens of each tree species sampled were as follows: *B. pendula*: 34 (A1 7, Auby 10, P 5, M 5, F 7); *P. trichocarpa*: 5 M; *P. euramericana*: 15 (Auby 10, P 5); *Salix caprea*: 5 (Auby) and *S. alba*: 12 (A1 7, P 5).

2.2. Soil and plant analysis

The soil samples from the polluted sites were taken from the top 20 cm, dried at 40 °C, and sieved through a 2 mm plastic mesh. Soil properties are displayed in Table 1. The soil texture was determined using the hydrometer method after wet oxidation of the organic matter by hydrogen peroxide [15]. Organic matter content was determined using the dichromate method [16]. The carbonate content was measured by volumetric analysis of CO_2 evolution after addition of 4 M HCl to the soil. Total carbon (Ctot) and total nitrogen (Ntot) were analysed by means of a CN-Analyser (CNS-2000, Leco, USA). Conductivity was determined in an aqueous soil extract [17]. Soil pH was measured in 0.01 M $CaCl_2$ and H_2O [18]. Total soil TE concentrations were analysed by X-ray fluorescence spectroscopy (XRF) (Spectro X-lab 2000, Germany). Prior to XRF-measurements, the soil was dried at 60 °C and ground to a fine powder. Labile soil TE concentrations were determined by extraction with 0.1 M $NaNO_3$ [19]. Filtered extracts were analysed for TE by means of ICP-OES (Varian, Vista-MPX CCS simultaneous). Labile P soil was determined according to Olsen [20]. For quality assurance we analysed two WEPAL referenced soils (Wageningen, Netherlands, no. 900 and 915). Recoveries for Pb and Cd were 101–106%.

All plant samples were dried at 60 °C to constant weight and the dry weight was recorded before they were ground with a Retsch ZM-200 centrifugal titanium mill. Subsamples (0.2 g) of all ground plant samples (bark, wood, and leaves) were digested in 15 mL of HNO_3 (65%), for 1 h at 120 °C, and diluted to 25 mL with H_2O. The extracts were analysed for TE by means of ICP-OES (Varian, Vista-MPX CCS simultaneous). For quality assurance we analysed certified plant reference material from Tobacco (Virginia tobacco leaves CTA-VTL-2, Polish Reference Material). Recoveries for Pb were >90% and for Cd 80–90%.

2.3. Calculations and statistical analysis

The percentage of bark relative to the total volume of the trunk was calculated as follows:

$$\text{Bark\%} = \frac{(r_{\text{total}}^2 - r_{\text{wood}}^2) \times 100}{r_{\text{total}}^2}$$

r_{total} is the radius of wood and bark at 1.5 m in height. r_{wood} is the radius of wood (without bark).

Bark to wood ratio of elemental concentration was calculated as follows:

$$\text{Bark to wood ratio} = \frac{\text{concentration of element in bark}}{\text{concentration of element in wood}}$$

All statistical analyses were carried out with SPSS 16. The statistical significance of soil effects was determined by analysis

Table 1
Characteristics of the contaminated sites selected for the investigation.

	A1	M	P	Auby	F
Coordinates	50°25′/3°01′	50°25′/3°02′	50°26′/3°00′	50°24′/3°05′	47°58′/7°55′
Mean annual precipitation (mm)	723	723	723	723	930
Av. temperature (°C)	11	11	11	11	11
Soil texture	Silty clay loam	Clay	Silty loam	Silty clay loam	Loamy sand
pH	7.0–7.3	4.5–7.2	4.1–7.0	5.2 - 6.1	5.7–6.5
Org. C (%)	1.9–3.7	0.9–1.6	0.6–1.1	n.d.–2.8	0.4–1.8
$CaCO_3$ (%)	2.5–7.8	n.d.–2.4	0.2–2.3	n.d.–2.7	0.5–4-6
C/N	14.8–32.6	21.7–42.5	27.2–33.4	25.7–42.2	11.2–16.6
El. conductivity ($\mu S\,cm^{-1}$)	111–240	116–201	42–146	37–132	30–64
Total TE conc. ($mg\,kg^{-1}$)					
P	950–1380	450–620	790–2300	450–2000	940–1400
N	1420–4460	400–1100	2700–16,000	432–1500	79–2020
Mg	7770–8900	8200–12,800	3400–8410	5970–9300	5800–7600
Ca	22,590–44,100	4050–16,930	2910–6170	5070–7360	4760–8060
K	14,320–15,090	16,900–18,880	15,450–25,750	15,500–17,440	17,180–20,250
Cd	26–53	11.4–69.3	0.8–17.4	15.4–49.1	17.1–44
Cu	54–94	26–61.5	16.4–74.9	19.1–56.6	22.8–36
Pb	1750–3520	540–1940	109–950	175–950	960–1500
Zn	1590–2670	534–2060	170–1005	2360–7200	4990–11,400
Labile TE conc. ($mg\,kg^{-1}$)					
P	40–101	12–36	16–54	9–26	b.d.
N	n.d.	n.d.	n.d.	n.d.	n.d.
Mg	40–77	61–79	220–290	58–81	21–44
Ca	629–701	374–690	210–260	424–550	80–235
K	75–116	311–447	110–220	109–180	15–39
Cd	0.12–0 31	0.5–4.44	0.03–0.15	1.75–9.63	0.06–1.94
Cu	0 18–0.35	0.08–0.15	0.1–0.18	0.03–0.15	0.03–0.1
Pb	0.15–0.6	0.18–0.83	0.3–0.47	0.08–5.5	0.03–2.8
Zn	0.63–2.23	2.1–49.4	0.89–5.74	103–515	0.83–8.7

n.d.: not determined; b.d.: below detection.

of variance (ANOVA). Differences were considered significant if $p < 0.05$.

3. Results and discussion

Many processes affecting the fate of TE, such as TE plant uptake and deposition through leaves as well as change of TE speciation during energy gaining process, can occur during BCL leading to risks for environment and humans. This imminent risk is evident from the accumulation of high amounts of foliage on the forest floor near smelters (as it was visible on our sites), primarily as an effect of suppressed litter decomposition [21]. Additionally, plants used for bioenergy production should reduce contaminant fluxes by minimising wind and water erosion as well as leaching. Economically the plants used for bioenergy production should yield a high biomass.

3.1. TE in soil and accumulation in leaves of B. pendula, Populus spp. and Salix spp.

Total soil TE concentrations ranged for Cd from 0.8 to 69 $mg\,kg^{-1}$, for Pb from 110 to 3500 $mg\,kg^{-1}$, for Cu from 17 to 94 $mg\,kg^{-1}$ and for Zn from 540 to 11,400 $mg\,kg^{-1}$ (Table 1). Cadmium concentration in the leaves was species dependent, for *B. pendula* it ranged between 1.2 and 8.9 $mg\,kg^{-1}$, independent of total soil Cd concentration (Fig. 1), in contrast to *Salix* sp. and *Populus* sp. where it increased with increasing Cd soil concentration, reaching concentrations of up to 60 $mg\,kg^{-1}$. Zinc concentration in leaves was more site dependent than species dependent (Fig. 1). No significant difference could be detected amongst the sampled tree species at one site, with the exception of site P where *B. pendula* depicted significantly higher Zn leaf concentration (1500 $mg\,kg^{-1}$) than S. *alba* (220 $mg\,kg^{-1}$). No correlation could be found for Pb leaves concentration with either the site or the tree species. This could be owing to the fact that there was a large heterogeneity in the total soil Pb concentrations. On site P for example, total soil Pb concentrations were 100–500 $mg\,kg^{-1}$ for *B. pendula* stands and 800–900 $mg\,kg^{-1}$ for *S. alba* stands. In general Pb uptake was very low because of the low plant availability of Pb. Lead is preferentially adsorbed by soil organic matter (Table 1) as well as the apoplast of plant roots [22]. Copper accumulation is in good agreement with those of other authors. There is in general little variation in Cu concentrations in above-ground parts of the investigated tree species, which usually ranged between 2 and 20 $mg\,kg^{-1}$, regardless of soil Cu concentrations, type of soil and tree species [23–25].

Besides wind and water dispersion of foliage, risk could derive from the consumption of plant parts by animals, thus reaching the food chain. This added stress originating from the contaminants may interact with other adverse effects to lower overall reproductive success and survival [26]. According to Swiss regulations [27], the maximum allowed TE concentration in fodder dry weight are: 150 $mg\,kg^{-1}$ Zn, 40 $mg\,kg^{-1}$ Pb, 15–35 $mg\,kg^{-1}$ Cu and 1 $mg\,kg^{-1}$ Cd. On the examined sites Pb and Cu concentrations in leaves did not exceed these guideline values. Zinc and Cd leaf concentrations however, surpassed these values by at most 26-fold and 60-fold, respectively. Thus, we propose *B. pendula* as the most suitable tree for biomass production on Cd contaminated sites. Even though *B. pendula* surpassed the values published by BUWAL, they were still 4.1–7.7 fold lower than the average values reached by *Salix* spp. and *Populs* spp. For the reduction of risk from Zn it is not possible to make a tree species recommendation as the uptake was correlated to the site rather than the tree species. All species, with the exception of *S. alba* at site P, reached concentrations which were approximately 4–24-fold higher than the allowed values (Fig. 1).

3.2. Metal distribution in wood and bark of Salix spp., Populus spp. and B. pendula

Metals in BCL can affect the energy/biochar production process, as they influence the decomposition temperature and the

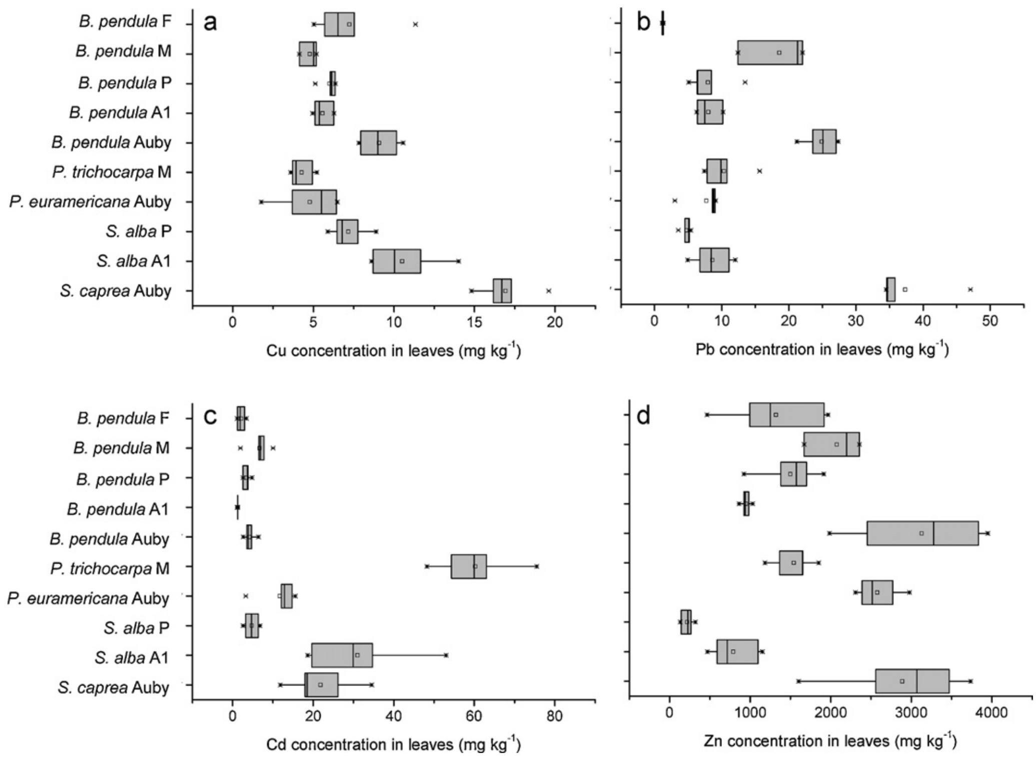

Fig. 1. Range of Zn, Pb, Cu and Cd concentration in leaves of the investigated trees. Boxes represent the median (vertical line) and 25–75% percentile. Whiskers represent the 90th and 10th percentile. Outliers are represented by x.

bio-oil yield. A number of inorganic ions are known to exert a great influence on the thermal degradation of polysaccharides and lignin, because they act as catalysts, lowering the decomposition temperature and increasing char yield. High concentrations of K in wood fuel decrease the ash melting point reducing the plant lifetime by sintering or slag formation [28]. According to Obernberger et al. [29] typical wood fuels from deciduous wood and bark (from uncontaminated areas) contain on average 1200–15,000 mg kg^{-1} Ca, 800–2000 mg kg^{-1} K, 2–5 mg kg^{-1} Pb and 10–50 mg kg^{-1} Zn, respectively. Cu concentrations in bark and

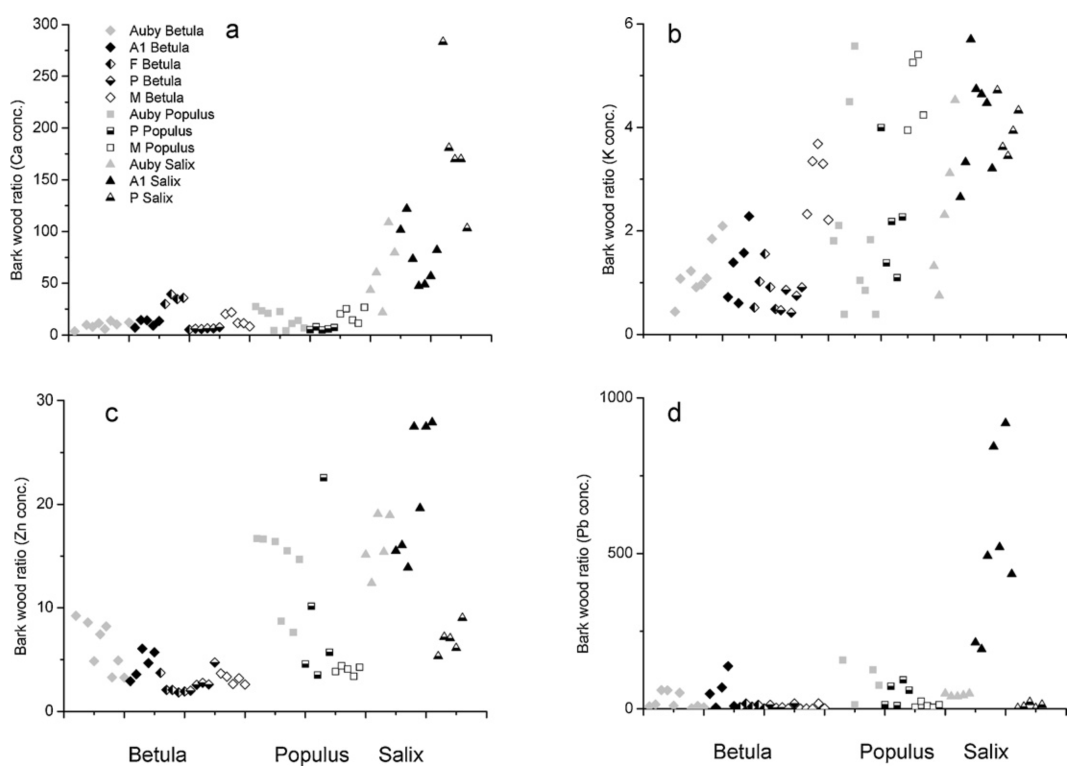

Fig. 2. Bark wood ratios of Zn, Pb, Ca and K in *Salix* sp., *Populus* sp. and *B. pendula* collected at the sites Auby, A1, M, P and F.

wood, which range between 0.1 and 5 mg kg^{-1} [29] are so low, that they will not be discussed further. Cadmium concentrations were similarly low, however due to its volatile nature Cd plays an important role when the biomass is burned or gasified (Table 2). Only *B. pendula* had lower Ca and K concentrations in wood and bark than the respective mean value given by Obernberger et al. [29]. In contrast, *Salix* sp. and *Populus* sp. surpassed these values by 1.2–2.2-fold, reaching concentrations that would affect the energy/biochar production process. Studies by Patwardhan et al. [12] and Kawamoto et al. [30] showed that Ca and K in similar concentrations to *Salix* spp. and *Populus* spp. bark have a significant influence on the char yield and the bio-oil production. Lead and Zn concentrations in wood and bark were in average 2.7 and 49-fold and 8.4 and 13.3-fold, higher than the values given by Obernberger et al. [29]. However, on all sites but Auby, their concentrations, compared to those of Ca and K, were so low that their influence on the process of energy production would be negligible. In general we propose that *B. pendula* is the most suitable tree species for BCL as the metal concentrations in wood and bark were lower than those of *Salix* spp. and *Populus* spp.

3.3. Proportion of bark in tree species and bark to wood ratio of K, Ca, Pb and Zn concentration

The proportion of tree bark is low in comparison to the whole trunk. However, the metal concentrations on bark can be so high that they have significant influence in the process of energy production. Fig. 2 shows the bark to wood ratio of TE and K, Ca concentrations, emphasising the higher metal concentrations of bark compared to wood. In general *B. pendula* displays a lower bark to wood ratio for Zn (1–9) and Ca (3–30) concentration than *Salix* sp. In the case of K all species on all sites had similar bark wood ratio concentrations (0.5–5.5). For Pb, with the exception of the *S. caprea* on site A1, the bark to wood ratios for all trees ranged between 4 and 130. Thus, besides selecting trees which have a low TE uptake, the proportion of TE rich bark has to be reduced.

To reduce the bark proportion (BP), Adler et al. [28] proposed an increase in the age of coppiced trees from 1–2 y to 3 y. Although the trees we sampled were approximately 15–40 y old we too came to the same conclusion, that the larger the trunk diameter the smaller the bark proportion (Fig. 3). The proportion of bark ranged from 9% to 17% on the sites A1, P, Auby and M. This is in the same range as found by Klasnja et al. [31] for bark proportion. Contrary to the other sites, at the site F, *B. pendula* (Fig. 3d) showed the converse correlation, the BP increased with increasing trunk radius, and showed also the highest BP ranging from 15 to 39%. It is possible, there is variability in the proportion of bark within a species and subspecies, thus making it impossible to generalise about the relationship between tree age and bark proportion and stressing the fact that the species and subspecies have to be selected carefully to achieve as low a BP as possible.

In order to keep bark proportion and plantation establishment costs as low as possible a longer rotation time would be more favourable than shorter times for coppicing. Thus in BCL, similar to forestry intended for wood and paper production, rotation periods of 20–50 y, depending on tree species and site would be preferable. This would generate income similar to forestry intended for wood and paper production, using the trees for the profitable production of bioenergy/biochar and simultaneously reducing the risk deriving from contaminated sites.

3.4. Future issues (fate of products and by-products)

Unlike phytoremediation/phytoextraction where plants are considered waste and thus their fate often neglected; in BCL the

Table 2
Ca, K, Pb, Cd and Zn concentrations in wood and bark tissue of *Betula pendula*, *Populus* spp. and *Salix* spp. from the sites Auby, A1, F, P and M.

	Site	Metal conc. in wood samples (mg kg^{-1})					Metal conc. in bark samples (mg kg^{-1})				
		Ca	K	Pb	Zn	Cd	Ca	K	Pb	Zn	Cd
B. pendula	Auby	488 ± 100	408 ± 46.7	2.9 ± 2.1	189 ± 70.1	0.7 ± 0.5	4644 ± 1446	503 ± 197	55.1 ± 51.1	1206 ± 488	4.9 ± 2.9
B. pendula	A1	440 ± 41.6	464 ± 89.2	7.3 ± 1.7	44.6 ± 14.3	b.d.	4745 ± 1420	570 ± 185	380 ± 420	180 ± 47.6	2.6 ± 1.3
B. pendula	F	192 ± 77.9	386 ± 62.3	1.3 ± 0.1	136 ± 18.3	b.d.	6178 ± 1530	387 ± 137	12.7 ± 6.1	308 ± 82.5	2.5 ± 1.5
B. pendula	P	355 ± 20.4	467 ± 39.9	8.1 ± 2.2	49.9 ± 11.3	b.d.	2270 ± 235	320 ± 111	52.4 ± 23.9	144 ± 53.7	2.3 ± 1.0
B. pendula	M	472 ± 49.6	430 ± 89.9	12.8 ± 4.9	65.6 ± 14.1	1.0 ± 0.5	7153 ± 3380	1250 ± 252	9.9 ± 4.9	203 ± 62.7	4.2 ± 1.0
P. euramericana	Auby	1800 ± 1260	1508 ± 603	1.1 ± 0.6	168 ± 37.6	2.9 ± 1.1	19190 ± 3131	2007 ± 1000	125 ± 61.4	2284 ± 809	13.7 ± 5.3
P. trichocarpa	P	2930 ± 836	1780 ± 802	7.4 ± 1.0	49.7 ± 21.9	2.4 ± 1.1	17350 ± 3465	3590 ± 1437	350 ± 256	340 ± 95.1	13.6 ± 1.5
P. trichocarpa	M	1085 ± 587	1300 ± 463	12.1 ± 2.8	68.5 ± 14.2	8.5 ± 2.3	18460 ± 3390	6900 ± 1335	120 ± 65.4	270 ± 42.0	35.1 ± 10
S. caprea	Auby	527 ± 215	2006 ± 1470	1.2 ± 0.2	93.4 ± 10.9	3.7 ± 0.9	27550 ± 5837	2940 ± 540	50.2 ± 9.3	1507 ± 275	21.8 ± 5.6
S. alba	A1	454 ± 177	1010 ± 80.5	3.1 ± 1.1	33.4 ± 4.6	3.6 ± 0.8	30910 ± 4620	4185 ± 1325	1520 ± 884	721 ± 277	42 ± 12.7
S. alba	P	231 ± 103	1160 ± 46.0	3.8 ± 2.0	25.1 ± 5.3	1.3 ± 0.1	32580 ± 4170	4640 ± 520	20.8 ± 9.1	170 ± 31.9	4.1 ± 0.4

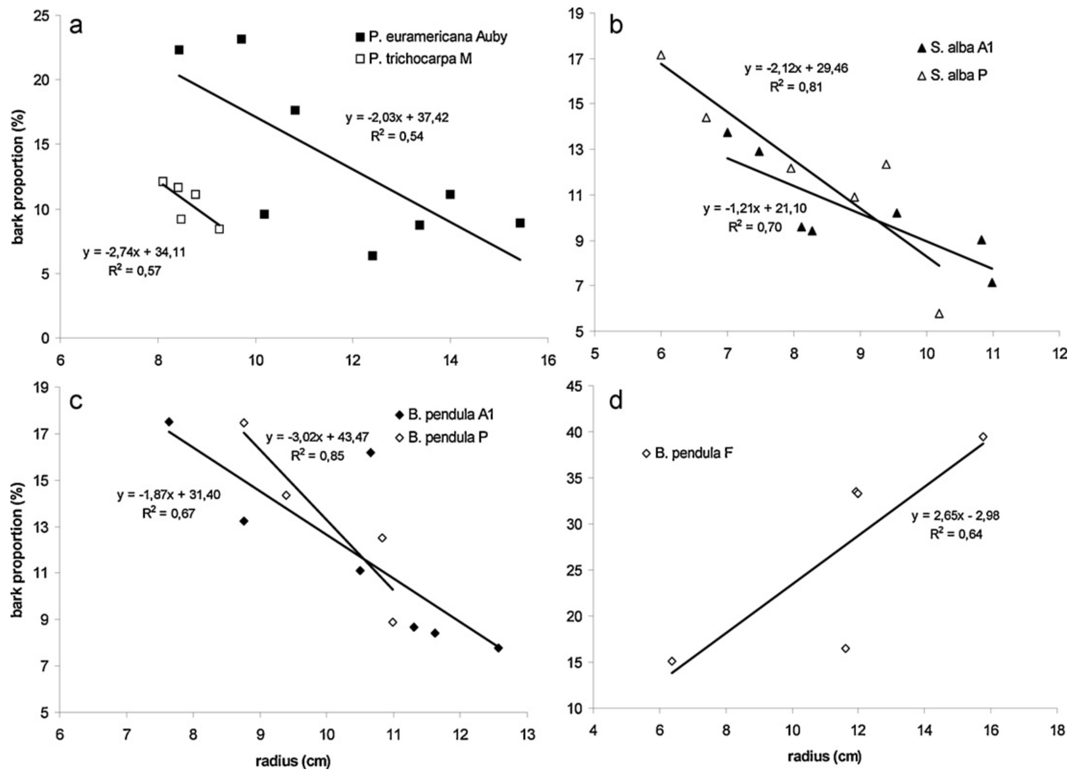

Fig. 3. Proportion of bark (volume %) in relation to trunk radius of (a) *P. euramericana* on Auby and *P. trichocarpa* on M, (b) *B. pendula* on A1 and P, (c) *S. alba* on A1 and P and (d) *B. pendula* on F.

plants employed produce revenue and thus their fate and that of the by-products is very important. In this article we have only focused on the biomass production, however due to the importance of the subject we discuss shortly their fate and possible implications. As well as influencing the bioenergy production process, metals also influence the fate of the by-products, such as fly ash, digestate, biochar and ash. Depending on the TE concentrations there are various options; biochar from gasification could be applied in the area it was derived from in order to sequest C and reduce greenhouse gas production. However, an assessment needs to be made on the solubility and mobility of the TE in the biochar. After anaerobic digestion, the digestate could be applied as a fertiliser on the land where the trees were grown. This would return extracted TE to the soil, possibly in a different chemical form. Thus, the toxicity and environmental fate of these reapplied TE need to be assessed. Additionally, a green coverage should be applied on the ash and/or digestate so that the dust production is minimal. In modern biomass combustion plants it is possible to retain 30–60% of the total Cd input and 25–50% of the total Zn input in the filter fly ash [29], thus unlike decentralised heating power stations using wood pellets, they should not face the problem of TE contamination of surrounding areas.

4. Conclusion

The increasing demand for energy will make it necessary to explore the use of biomass for energy production. This, however, should not come at the cost of food production. Production of Biomass on Contaminated Land (BCL) seems to be a worthwhile option though. From the investigated tree species in this study *B. pendula* seems to be the most suitable as it accumulated the lowest TE concentrations in leaves, wood and bark. Nevertheless, it remains necessary to verify the effectiveness of its energy production. It is also necessary to verify its effects on the pollutant migration in soil (before accumulation) as *B. pendula* may promote acid soil conditions that may increase the depth migration of metals. Similarly, a thorough study of the root system would evaluate if *B. pendula* creates preferential pathways for migration. Additionally, in terms of agricultural practices a long rotation period of approximately 25 y would be appropriate for BCL as the proportion of TE rich bark decreases with time. By-products such as digestate (biogas production) and ash (combustion) could be applied on the areas of origin as fertiliser. The production of biochar, which could be also applied on the area of biomass origin, could open new possibilities for the utilisation of these contaminated areas. However, before an application of ash/biochar/digestate is possible, the speciation of the TE in them has to be assessed in order to determine the possible toxicity and the bioavailability so as to avoid any unnecessary risks.

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