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Test of Tree Core Sampling for Screening of Toxic Elements in Soils from a Norwegian Site

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Test of Tree Core Sampling

Abstract.

Tree core samples have been used to delineate organic subsurface plumes. In 2009 and 2010, samples were taken from trees growing on a former dump site in Norway and analyzed for arsenic(As), cadmium(Cd), chromium(Cr), copper(Cu), nickel(Ni) and zinc(Zn). Concentrations in soil were in averages 30 mg/kg dw for Zn, 2 mg/kg dw for Cu, and < 1 mg/kg dw for Cd, Cr, As and Ni. The concentrations in wood samples from the polluted test site were compared to those derived from a reference site. For all except one case, mean concentrations from the test site were higher than those from the reference site, but the difference was small and not always significant. Differences between tree species were usually higher than differences between reference and test site. Furthermore, all these elements occur naturally, and Cu, Ni and Zn are essential minerals. Thus, all trees will have a natural background of these elements, and the occurrence alone does not indicate soil pollution. For the interpretation of the results, a comparison to wood samples from an unpolluted reference site with same species and similar soil conditions is required. This makes the tree core screening method less reliable for heavy metals than, e.g., for chlorinated solvents.

Keywords: Heavy metal; Soil; Wood; Polluted; Plant uptake; Monitoring

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1 Introduction

Biomonitoring for heavy metals is an established technique (Markert 1993, Markert et al. 1999). Mosses, lichens, but also trees and tree rings have been sampled to determine the concentration level of heavy metals in the environment (Gratani, Crescente, and Varone 2008, Markert and Wtorova 1992, Monticelli et al. 2009, Migeon et al. 2009). Phytoscreening is a new term and was given for the use of vegetation samples to screen subsurface pollution (Sorek et al. 2008). The technique to take tree cores to track pollution plumes below surface has been found to be a simple, fast, noninvasive and inexpensive screening method (Vroblesky, Nietch and Morris 1999, Ma and Burken 2002, Schumacher, Struckhoff and Burken 2004, Gopalakrishnan et al. 2007, Trapp et al. 2007, Sorek et al. 2008, Larsen et al. 2008). The principle is that roots take up pollutants from soil or shallow groundwater. With the transpiration stream, the contaminants are transported above the surface and into the stem, where they adsorb to the wood and other plant parts. Wood is sampled with a tree corer and analyzed for the pollutants. Elevated concentrations in wood indicate subsurface contamination (Vrobelsky et al. 1999). The method is rapid, simple, cheap, and allows a high sample number in short time without heavy equipment. Tree core sampling is thus seen as a reliable and inexpensive alternative method for investigating and monitoring the extent of shallow pollutants (Larsen et al. 2008). Subsequently, tree core sampling was recommended for initial screening of an area (Sorek et al. 2008) and for assessing the presence of pollutants (Larsen et al. 2008), and the method is used frequently in practice now (unpublished engineering work). However, so far all studies have dealt with

chlorinated solvents, such as trichloroethylene (trichloroethene, TCE), tetrachloroethene (PCE) and trichloroethane.

The purpose of this study was to test the tree core method for toxic elements, such as arsenic and heavy metals. Arsenic (As), cadmium (Cd), chromium (Cr), copper (Cu), nickel (Ni) and zinc (Zn) are frequent pollutants in soil, mainly from human activities but also from natural sources. At elevated levels, all these are toxic to humans and wildlife, and their occurrence in soil is regulated by legal standards in most countries. Their dissolution in soil solution and the subsequent uptake into vegetation depends on chemical speciation (and thus pH and redox potential), on organic matter, clay content, and on the concentration of other ions (Barber 1995, Hough et al. 2004, US EPA 2005, Swartjes et al. 2007, Legind and Trapp 2010). The bioavailable fraction in soils may decrease with time, leading to reduced uptake (Kirkham 2006). Fungi may facilitate transport to roots (Smith et al. 2010).

The individual elements may - depending on their xylem or phloem transport - move preferably into different plant parts, i.e. roots, stem, leaves and fruits (Thorne, Walke and Maul 2005). Wood was sampled because it is protected from aerial deposition, it is available throughout the whole year (samples were taken in winter) and it does not change much with time (as leaves do). A disadvantage is that little is known about the uptake of toxic elements into wood since most studies focus on edible plant parts such as fruits or leaves. Thus, data about accumulation of toxic elements in wood are needed, also for an assessment of the feasibility of phytoextraction.

Wood from trees (mainly birch, willow and poplar) growing on a former dump site was sampled and analyzed for As, Cd, Cr, Cu, Ni and Zn. The concentrations were compared to those from trees of the same species growing outside the contaminated area. The objectives of this study were to determine typical concentration levels in wood and to test the tree core sampling method for the screening of subsurface pollution with toxic elements (focus on heavy metals).

2 Methods

2.1 Test site

The *Møringa* (former) dump site near Horten, Norway, is an artificial half-island at the Oslo fjord created by the dumping of waste. From the 19th century until 1993, it has received waste oil, oil distillery waste, welding slags, blowing sand and building residues, originating from ship yards, oil recycling, ship and aircraft maintenance, and lead battery production. Investigations of the site between 1992 and 2005 (Amundsen et al. 2005) revealed that the site is contaminated with large amounts of heavy metals, petroleum products, polycyclic aromatic hydrocarbons and polychlorinated biphenyls. On the site, wild-type pioneer vegetation consisting of grassland and trees (such as willow, poplar, birch and cherry) has developed.

The depth of the waste deposit is approximately 3 m. The cover at the Møringa waste site consists of 0.2 to 0.5 m clean soil. The concentrations of the elements of interest in this cover are unknown but it can be assumed that they are close to natural soil (background levels).

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All soil samples from the site are composite samples (30-50 kg), where each was taken from deep (2-4 meters) pits of an area of about 16 m^2 (4 x 4 meter). The main aim of the sampling in 2004 (Amundsen et al. 2005) was to investigate the leaching potential of toxic elements in the waste to predict the future influence of the waste site on the local marine environment. Most waste samples were therefore collected from the lower part between groundwater table and 1 m above groundwater table, but some were also taken from the upper part of the deposited waste. Eight risk zones were mapped, each with relatively homogeneous waste filling (Fig. 1). Concentrations of toxic elements in deposited material from the eastern part of the landfill (Ø1, Ø2 and Ø3) are significantly higher than in most of the western areas (V4 to V8) (Tab. 1), but the concentration level of pollutants seems to be quite uniform with depth.

<Figure 1>

<Table 1>

2.2 Tree core sampling

Tree core sampling was performed at the Møringa site on the 8th and 9th of July 2009 and on the 30th of March 2010. Trees were sampled in the eastern part of the site which is densely covered by trees. Sampled tree species were predominantly birches (*Betula sp.*) and willows (*Salix caprea*), but included also cherry (*Prunus sp.*), aspen (*Populus tremula*), ash (*Fraxinus excelsior*) and mountain ash (*Sorbus aucuparia*) in the first campaign. Only willow (*Salix caprea*) and poplar (*Populus tremula* and other poplar species) were sampled in the second campaign. Reference samples were taken 20 to 50 m outside the area of the dump site, and at a location about 10 km away. All reference samples were closer to urban area (Møringa

peninsula is the remotest place in this area), and contamination from other sources than dumped waste can not be excluded.

All tree cores were taken at a stem height of 1 m using a 6 mm increment borer (Suunto, Finland). Tree cores had a length of 6 cm, where the outer centimeter (containing the bark) was discarded to avoid atmospheric influence. Only in 2009, the next centimeter (cm 1-2 towards stem center) was used for mixed samples, and cm 2-6 made up an individual sample. Mixed samples were collected in order to represent subareas, including between 3 and 9 individual tree cores. The aim here was to test whether the analysis of one mixed sample (i.e. several trees in the area of interest) instead of many individual samples (one per tree) is an appropriate method for subsurface characterization, as this would save laboratory efforts. During the second campaign, wood from cm 1 to 5 was used as sample, and two replicates from each tree were taken.

2.3 Extraction and chemical analysis

Soil samples were dried at 40 °C to constant weight, extracted with aqua regia (concentrated hydrochloric acid: concentrated sulfuric acid 3:1) and analyzed using ICP-AES (Amundsen et al. 2005).

Wood samples from the first campaign were extracted using an autoclave. The wood samples were dried at 75-85 °C to constant weight. Between 0.5 and 0.8 g of the dried sample was weighed into 100 ml blue cap bottles, then 10 ml 65% HNO_3 and 10 ml miliQ water were

added. The sample was autoclaved for 30 min at 125 °C and cooled to room temperature afterwards. 5 ml 30 % H_2O_2 were added and the sample was placed on a sand bath for 20 min without cap. The sample was quantitatively transferred to a 50 ml volumetric flask. MiliQ water was added to the total volume of 50 ml. The flasks were shaken for 1 min and the sample was then filtered into a plastic (PE) bottle for storage at room temperature. Before analysis, 7 ml of sample was transferred to a test tube and then analyzed using ICP-OES.

Some samples of the first campaign had unusually high concentrations of Cu, Ni and Zn, and we found that the procedure erratically contaminated samples during extraction. Even though these samples could be identified, the results for Ni and Cu from the first campaign were discarded (the results for Zn could be used, though with a high DL, because the concentrations were sufficiently above the laboratory background), and the method was optimized and changed to sand bath extraction for the second campaign.

For the sand bath method, wood samples were dried as before. Between 0.5 to 0.8 g of the dried sample were weighed into a 50 ml volumetric flask. Then 10 ml 65 % HNO_3 was added, and the flask was placed on a sand bath for 2 hours at 70-80 °C. Samples were then removed and cooled at room temperature for 10 min. Afterwards, 2.5 ml 30 % H_2O_2 were added and the samples were placed back on the sand bath until the gas reaction was completed. The procedure was repeated with additional 2.5 ml 30 % H_2O_2. MiliQ water was added to get 50 ml volume. After shaking for 1 min, approximately 5 ml of sample were transferred to a centrifuge glass, shaken and emptied. The rest of the sample was transferred to the same centrifuge glass and centrifuged for 10 min with 2500 rpm. The supernatant was

transferred to plastic (PE) bottles for storage at room temperature. For analyses, 7 ml of sample were transferred to test tubes and then analyzed at ICP-OES.

The methods were validated by comparison to the referenced soil standard QC Loam Soil (Sigma Aldrich, DK). All concentrations for soil and wood are given for the dry weight (dw).

2.4 Statistics

The main question of the study was whether the concentration of toxic elements in wood from trees on contaminated sites is elevated, compared to reference sites. This was tested using a one-tailed t-test with an error probability of 0.05 (α = 5%). The distribution of the experimental data was tested using the Kolmogorov-Smirnov (KS) test for continuous distributions, implemented in the software Crystal Ball. Three distributions were tested, namely normal, log-normal and uniform (rectangular) distribution. The assumption of equality of sample distribution and tested distribution was rejected if the distance between both was above a critical distance D_{crit}. These critical distances were taken from Sachs (1991). For calculation of mean, standard deviation, minimum, maximum, F-test and t-test, values below detection limit were replaced by 1/2 detection limit. The data for Ni and Cu from the first campaign were not statistically analyzed, as well as the results for As from the second campaign, which were close or below to detection limit.

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The statistical difference between the measured concentration in the mixed sample and the concentrations in the corresponding individual samples was tested using the "one-value t-test" (Bahrenberg, Giese and Nipper 1990). The tested t-value is

$$t_{test} = \frac{|x - a|}{s/\sqrt{n}}$$

hereby, x is the mean of the individual samples ($n \geq 3$) and a is the concentration of the mix sample ($n = 1$, i.e. the fixed value). The null hypothesis H_0 is rejected if t_{test} is above the t-distributed t_{crit} with degree of freedom (df) = n-1 and $\alpha = 5\%$.

3 Results

Table 2 shows the overall characterization of the wood samples from Møringa. Highest concentrations were measured for zinc, followed by copper (2nd campaign only). The other elements (As, Cd, Cr, Ni) had similar concentrations, most of them below 1 mg/kg. The concentration results from the first campaign were typically more log-normal than normal distributed, which makes a statistical analysis with parametrical methods critical. For all results from the second campaign, normal distribution could be accepted. The measured concentration level was for all elements quite similar in campaign one and two. Only cadmium showed distinctly higher values in wood from the second campaign. The reason is that exclusively willows and poplars were sampled, and those species showed the highest cadmium uptake of all trees that were sampled at the site.

Table 3a shows the comparison of results from reference and test site from the first campaign. The mean values of arsenic from reference and test site were significantly different, but it should be noted that all values from the reference site were below DL. The concentrations of cadmium were far higher in willow wood than in birch wood. The difference between reference and test site was significant for both birch and willow. For chromium, concentrations in willow wood were also higher than in birch wood, and elevated at the test site, though not significant. For zinc, a significant difference was found only for willow wood, even though concentrations in birch were higher.

From campaign one, it became obvious that willow and poplar trees took up most elements in higher concentration than birch, cherry and ash. Also, the difference of concentrations in

wood from the test site, compared to those from the reference site, was more pronounced. This was the reason to choose willow and poplar, both from the family *Salicaceae,* as preferred species for the second campaign. In the second campaign (Table 3b), Cu was found to be significantly increased in the wood from the test site. The concentration for Cd were elevated in samples from test site for willows and reduced for poplar. Ni and Zn were elevated in samples from the test site, and only significantly for poplar. The mean concentrations of Cr were similar in all samples.

The test for differences between mixed samples taken during the first campaign (cm 1-2, first campaign only) and the individual samples, by which the mixed sample was composed, yielded that in about half of the cases there was a significant difference (one-value t-test, α = 5%), and in the others not. Mixed samples can reduce the sample number, but due to the relatively small differences between trees from reference sites and those from the test site, a high sample number is preferable, to get better statistics.

<Table 2>

<Table 3 ab>

4 Discussion

4.1 Differences in uptake between test and reference site

The main objective of the study was to test the feasibility of phytoscreening for toxic elements. This was done by comparing concentrations of As, Cd, Cr, Cu, Ni and Zn in wood samples from the Møringa dump site (test site) with concentrations in samples from nearby reference sites. The results (Table 3) show that in all except one case (Cd in poplar wood), the average concentrations of the investigated toxic elements were higher in wood from the test site. This is promising. However, the differences were sometimes very small, and individual trees from the reference site may show much higher content than trees from the test site. Figure 2 shows some typical results. Figure 2 a (Cr in willow wood) displays a situation where the mean concentrations in wood from the test site (0.41 mg/kg) is much higher than those in wood from the reference site (0.24 mg/kg). Still, the second highest concentration of all samples was measured in wood from a reference tree, and the difference of the means is statistically not significant (Tab. 3a). Contrary, Figure 2 b (Cu in willow) shows an example where this difference is statistically significant. Indeed, the concentration level in wood from the test site is clearly elevated. Nonetheless, individual trees from the reference site may have concentrations above individual trees from the test site. This demonstrates that the method - if applied - requires sampling of a many trees to avoid false conclusions.

Elevated concentrations of toxic elements in trees from contaminated sites were also reported by other authors.

277

278 Arsenic in tree rings was measured and related to pollution by Markovic et al. (2009). The
279 concentration in individual tree rings varied largely over the years. The average concentration
280 of arsenic in poplar wood was 12.9 mg/kg in wood from the less polluted and 20.2 mg/kg in
281 wood from the more polluted site. In a study with birch growing on a chromite processing
282 waste site and willows growing on a sewage disposal site, Cr was poorly taken up into the
283 aerial part of the plant (i.e. all values, including wood, were below DL = 5 mg/kg). Cr was
284 measurable only in the roots. Zinc levels in wood from contaminated sites were above 200
285 mg/kg (Pulford, Watson and McGregor 2001).

286

287 < Figure 2 ab >

288

289 4.2 Differences between tree species

290 The difference between tree species (birch and willow, willow and poplar) was for two heavy
291 metals (Cd and Cr) larger than the difference between test and reference site. For two heavy
292 metals (Ni and Zn), the variation due to species was approximately as large as the difference
293 between the sites, and only for two elements (As and Cu) the site was mainly determining the
294 concentrations in wood. This means that for a comparison between reference and test site, the
295 same tree species must be chosen. This will not always be possible.

296

297 Some tree species (e.g., willow) were better suited than others (e.g., birch). In the study of
298 Migeon et al. (2009), who measured the uptake of heavy metals into 25 tree species growing
299 on polluted soils, cadmium was highest in Salicaceae family members, identical to our

finding. Under unpolluted conditions, the normal Cd concentration level in plants is 0.1 mg/kg and the maximum is 0.2 mg/kg (Kirkham 2006). In our study, concentrations in wood were below this range, except for *Salicaceae* (willow and poplar). Large variations between species were also found at a French site (Migeon et al. 2009) for Cd, Cr and Zn, and less for Cu. Concentrations varied with age of the tree ring (Monticelli at el. 2009), and Hagemeyer and Schäfer (1995) found a variation of the concentrations of cadmium, lead and zinc with season. Riddell-Black, Pulford and Stewart (1997) found a certain natural variability of the accumulation even within the same species. Arsenic uptake into needle trees was measured by Haug, Reimer and Cullen (2004). Spruce tree samples from an arsenic-rich site had total As concentrations between 0.04 and 0.13 mg/kg, i.e., even below the values obtained here, while concentrations in Douglas pine were much higher, up to 176 mg/kg in stem. The concentration in new-grown stem was higher than in old stem.

Copper, nickel and zinc are essential micronutrients. According to Marschner (1995), the average concentration of copper in plant shoots that is sufficient for adequate growth is 6 mg/kg dw. Concentrations of copper found in wood from Møringa ranged from 0.5 to 5 mg/kg. Average concentration of nickel in plant shoots that are sufficient for adequate growth are about 0.1 mg/kg (dw). Nickel concentrations in wood from the Møringa site ranged from 0.12 to 0.75 mg/kg. Zinc concentrations in dry shoot of 20 mg/kg are required for growth (Marschner 1995). Measured concentrations in wood ranged from < 10 to 97 mg/kg. Plants can not grow without a certain minimum level of these elements (Marschner 1995). The presence of these metals alone can therefore never be a proof for soil or groundwater pollution. Furthermore, it is likely that the uptake of the essential elements is

enzyme-regulated and thus follows Michaelis-Menten kinetics (Barber 1995, Chen et al. 2008, Trapp et al. 2008). This means that the uptake decreases with higher environmental concentrations (Markert et al. 1999). It also follows that the concentration differences in wood will be smaller than those in soil, making the detection of subsurface contamination from differences in wood concentrations more difficult.

4.3 Limitations

Concentrations of heavy metals in soil at the Møringa site were determined in a separate study, and only at few sample points. It is therefore not possible to compare concentrations in trees to those in soil, i.e., a correlation of concentrations is not possible. Furthermore, the waste with high pollutant concentrations was covered with a less-polluted layer of soil, which was thick enough so that the trees probably were not in contact with the more toxic underground. Only few soil samples were taken from the cover (Tab. 1).

In order to allow a conclusion on the subsurface pollution level from vegetation samples, the bioavailability of the toxic elements should not be different. Kirkham (2006) reports that the pH of the soil is usually the most important factor that controls uptake, with low pH favoring Cd accumulation. High phosphate and zinc concentrations decrease Cd uptake. The reference site should thus have very similar conditions to the test site (e.g., soil type, pH, nutrient supply, tree species, weather), except, of course, the concentrations of toxic elements. This turned out to be difficult for the Møringa site. A difference in pH is likely, because the waste at the site was partly mixed with bricks, cement debris etc. which leads to alkaline pH (pH 7

to 9), while normal forest has typically pH 5 to 6. Furthermore, the area along the coast is densely populated, and urban waste (such as defect TVs) was found all around, also on the reference area. A fence was close by (eventually releasing zinc, cadmium, nickel and chromium), and a road. Generally, it will be difficult to find totally unpolluted soils in urban areas, and thus well-suited reference sites. Also, no soil samples were taken and analyzed from the reference site, so neither concentrations nor soil conditions are known.

Concentrations in wood were generally low, typically factor 100 or more lower than concentrations in soil. At the same time, sample volumes were necessarily small (< 1 g). Subsequently, the measured concentrations for some elements (As, Ni) were often close to or even below the detection limit. The use of another analytical instrument (ICP-MS, AAS with graphite oven) might improve the limit of determination. Also, from this aspect, the measurement of leaves might be superior, because concentrations are generally higher than in wood (Vandecasteele et al. 2008, Harada et al. 2010). On the other hand, atmospheric deposition is often an important source for heavy metals in leaves (Gratani et al. 2008) and could disturb the phytoscreening. Indeed, atmospheric deposition (Gratani et al. 2008, Legind and Trapp 2010) could be one reason for the often small difference of concentrations in samples from test- and reference site.

Toxic elements are also toxic to trees (Marschner 1995). Perhaps, trees avoid growth in polluted soil and extend their roots preferably into cleaner soil areas. Also, maybe trees cannot grow at all in highly polluted soils, which mean in turn that trees cannot be used as indicator for such high pollution. The method is therefore restricted to a certain concentration

range, limited by detection limit at the lower end and by severe toxic effects at the higher end.

Uptake from soil into roots is most likely from the bioavailable pool (McLaughlin 2002). This means, elevated levels in wood do not necessarily indicate elevated total concentrations in soil. This can, of course, also be seen as an advantage of the method, because it directly tracks the fraction of the chemical that is freely available for uptake, toxicity and leaching. Legal standards, however, are typically based on total concentration in soil, e.g., in Denmark (Miljøstyrelsen 2009).

Toxic metals reside often in soil layers close to the surface and are therefore available for hand-driven borers. It is probably easier and more certain to determine the heavy metal concentrations of soil samples, instead of using the indirect analysis of wood samples. On the other hand, trees do integrate over a large volume (up to > 100 m^3 root zone per tree) and smooth out inhomogenities of soil contamination. Also, they yield directly the bioavailable (and thus toxic and mobile) fraction.

5 Conclusions

We tested phytoscreening of toxic elements and heavy metals for an abandoned waste site, by comparing concentrations in wood samples from the test site with concentrations in samples from reference sites. In all except one case, the concentrations of the investigated toxic elements were higher in wood from the test site. However, toxic elements occur in

higher or lower amounts in any soil. Subsequently, the elements were also present in reference samples. The uptake underlies natural variations and depends on tree species and soil properties. Consequently, the differences between contaminated test site and (nominally) unpolluted reference site were not always statistically significant.

Although it is too early to judge the feasibility of the tree core method for toxic metals, it became already apparent that the method is more difficult to use than for chlorinated solvents, which are purely anthropogenic compounds. In particular, the occurrence of a toxic element in wood alone can not be used as criterion for subsurface pollution, a statistically sound comparison to samples from a well-suited reference site (same tree species, same age, similar soil properties, non-polluted) is necessary. This increases the efforts and the uncertainty of the method.

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546

547 Tables and Figures

548

549 **Table 1.** Total concentration in soil (mg/kg) measured at Møringa (Amundsen et al. 2005).

Sample	As	Cd	Cr	Cu	Ni	Zn
Ø1-2	61	14	200	1500	130	7300
Ø1-1+2	69	20	170	1700	120	5100
Ø2-1+2+3	75	16	170	3700	190	9800
Ø2-4+5	15	2.2	49	860	42	3900
Ø3-1+3	44	8	130	2500	120	6000
Ø3-2+4	28	9.5	76	1400	120	3000
V4 bottom	5	1.3	81	3700	160	540
V4 sand	5	0.2	71	76	560	450
V5 bottom	21	3.9	100	4000	63	3900
V5 top	15	3	99	1100	88	11000
V6-1+2+3	5	1.4	58	280	410	1000
V6-4+5	5	0.2	110	18	2300	320
V 7 1+2	5	0.9	92	140	460	790
V 7 4+5	5	1.8	150	940	260	1400

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553 **Table 2.** Description of the the wood samples from Møringa, first campaign (n = 71) and second
554 campaign (n = 68). Concentrations in mg/kg dry weight; std = standard deviation; min = minimum;
555 max = maximum; DL = detection limit (mg/kg dw); <DL =number of samples below DL.

Element	Campaign	mean	std	min	max	DL	<DL
As	1	0.32	0.24	0.23	1.64	0.45	57
Cd	1	0.18	0.26	0.015	1.01	0.03	30
Cd	2	0.66	0.31	0.15	1.49	0.03	0
Cr	1	0.23	0.26	0.06	1.94	0.04	0
Cr	2	0.40	0.26	0.06	1.17	0.11	3
Cu	2	2.17	1.06	0.49	5.28	0.97	1
Ni	2	0.29	0.17	0.12	0.75	0.24	23
Zn	1	28.6	18.0	4.1	92.6	8.1	6
Zn	2	33.1	13.5	14.2	96.9	0.48	0

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Table 3a. Mean of measured concentrations (mg/kg dw) of elements in wood samples from Møringa, first campaign; R is reference site (nominally low polluted) and T is test site (high polluted). Significant differences in bold (α = 5%).

	all trees n = 71		birch n = 34		willow n = 16	
	R	T	R	T	R	T
As	0.23	**0.39**	0.23	**0.32**	0.23	**0.47**
Cd	0.14	0.18	0.015	**0.035**	0.33	**0.71**
Cr	0.19	0.26	0.13	0.15	0.24	0.41
Zn	25.4	30.0	33.1	39.8	15.0	**24.7**

Table 3b. Mean of measured concentrations (mg/kg dw) of elements in wood samples from Møringa, second campaign; R is reference site (nominally low polluted) and T is test site (high polluted). Significant differences in bold (α = 5%) or italic (α = 10%).

	willow n = 44		poplar n = 24	
	R	T	R	T
Cd	0.69	0.76	*0.62*	0.51
Cr	0.34	0.35	0.48	0.49
Cu	1.95	**3.05**	1.33	**1.66**
Ni	0.29	0.34	0.14	**0.29**
Zn	32.0	36.4	27.0	*32.5*

569 Figure legends

570

571 **Figure 1.** Map of the Møringa peninsula with risk zones and soil sampling points (Amundsen et al.
572 2005) and areas of tree core sampling, July 2009 and March 2010.

573

574 **Figure 2.** Example results from the tree core analysis (mg/kg dw); top: Cr in willow wood (1st
575 campaign); below: Cu in willow wood (2nd campaign). x-axis indicates location of trees (Fig. 1): Ref
576 refers to reference site; Ø refers to eastern part of the site, V to western part. Results from individual
577 replicates are shown.

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Figures

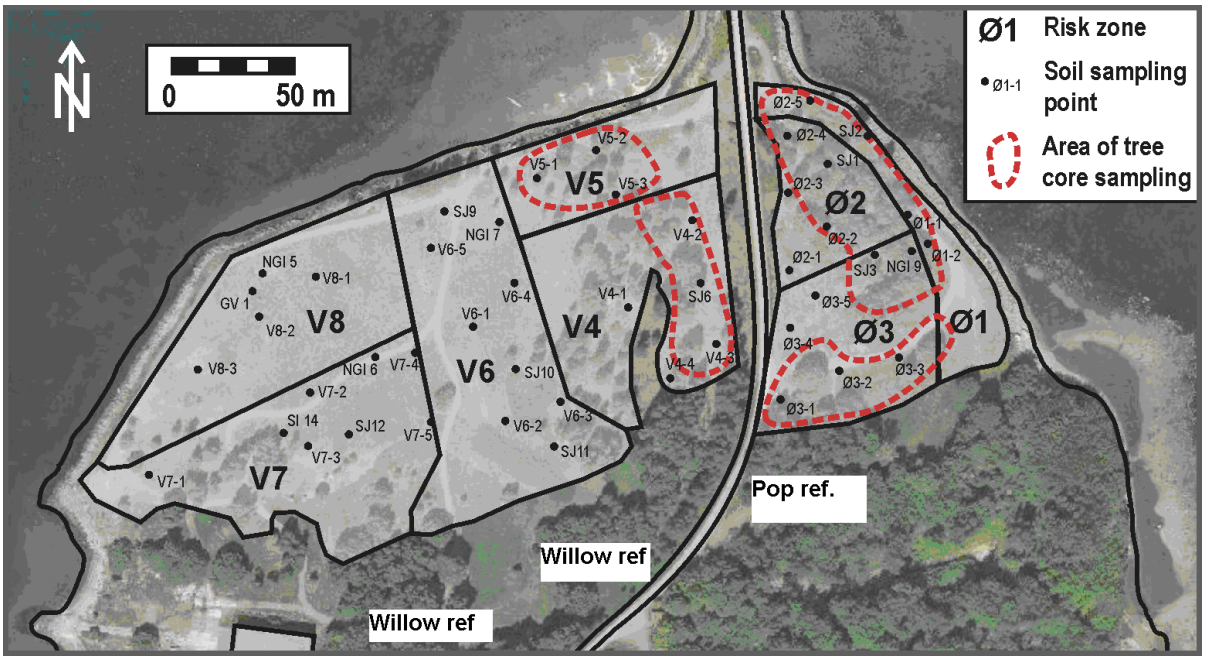
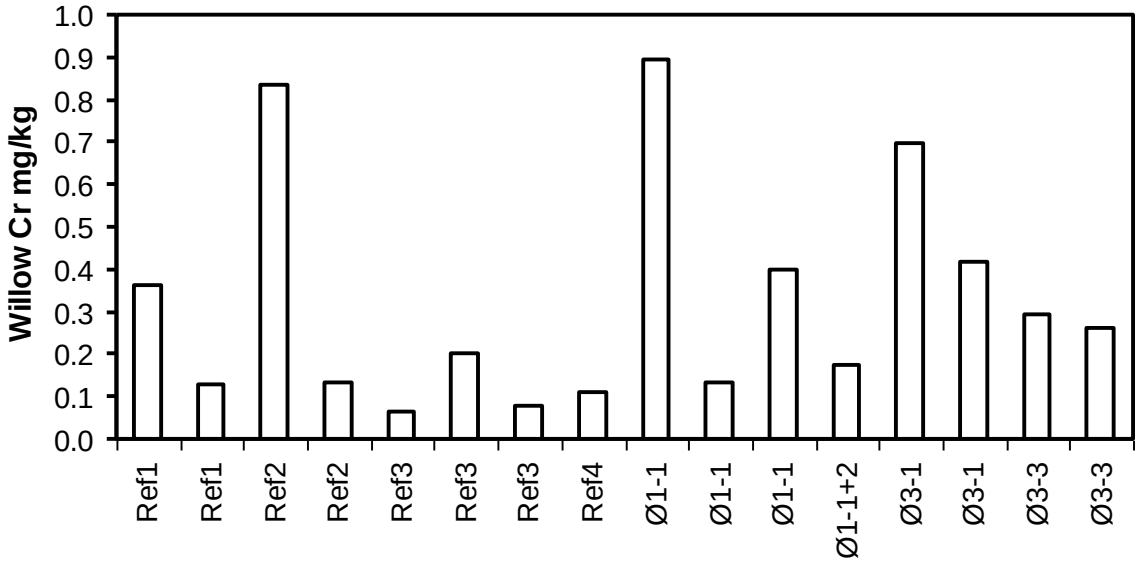


Figure 1

Test of Tree Core Sampling



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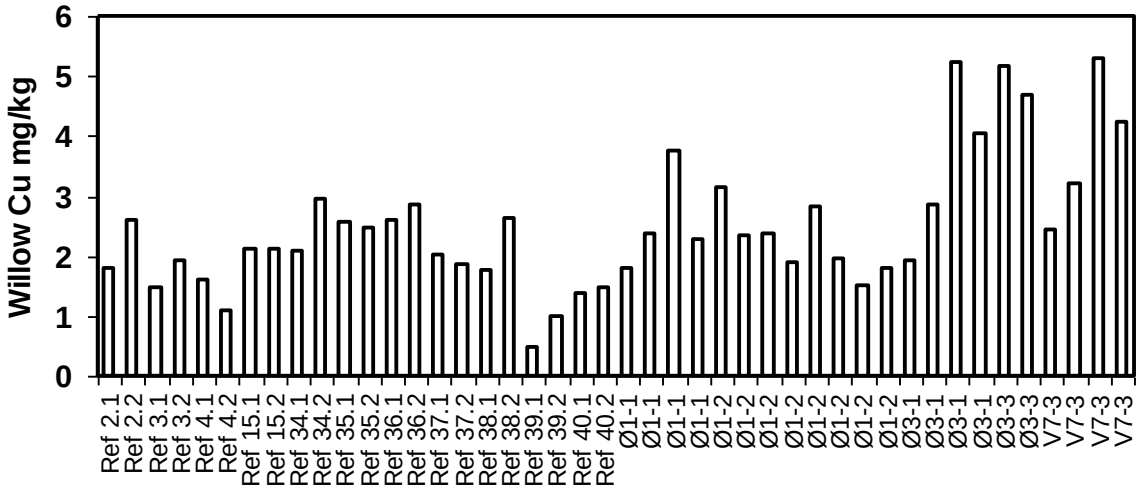

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587 **Figure 2 ab**

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