ARSENIC:
Accumualtion, Adaptation and Toxicity in Plants

For:

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Contaminants and Waste Management
Department of Indian and Northern Affairs
Whitehorse, YT.

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March 12, 1999

ORIGINAL

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Department of Geography
University of Toronto
Toronto, Ontario, Canada M5S 3G3
345 - 300 Main St.
Whitehorse, Yukon Y1A 2B5

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INTRODUCTION

This literature review was conducted in response to concerns about arsenic (As) concentrations in native vegetation growing in an area that has received mining spoils. The review presents a brief overview of As chemistry and then discussions of As in soils (bioavailability), uptake by plants, levels in separate plant components, phytotoxicity levels, tolerance and adaptation by plants, plant As uptake modelling and the role of As in biogeochemical prospecting. The objective of the review is to assess whether the measured concentration may pose a toxicological risk to the vegetation. The risk to consumers of the vegetation must be assessed in the context of the rate at which this vegetation is consumed, as well as other dietary and non-dietary routes of exposure.

BACKGROUND

Arsenic is a ubiquitous element that ranks twentieth in crustal abundance. Since it's discovery by Albertus Magnus in 1250, this element has been associated with medicine, cosmetics, and of course, poison. Bhumbla and Keefer (1994) estimated that more than 99% of the total natural arsenic in the environment is present in rocks due to the element's ease in which it can substitute for Si, Al or Fe in the crystal lattices of silicates. Table 1 summarizes the naturally occurring reservoirs of As, normalized to the soil compartment.

The majority of As released into the environment is by anthropological means. In the last few decades, As has been used as a doping agent in solid-state devices (ie. transistors), for laser material, bronzing, pyrotechnics and for improving the sphericity of gunshots (Bhumbla and Keefer, 1994). Arsenic is also commonly released from lead, zinc, gold and copper ores during

the smelting process. The toxicity of As to biological systems has made it an active constituent of insecticides, herbicides, fungicides, rodenticides, desiccants, soil sterilents and wood preservatives. The use of these organic arsenicals in the 1800s to mid-1900s has caused the majority of the contamination in soils.

A popular pesticide was acid lead arsenate ($PbHAsO_4$) that was used for insect control of deciduous fruit trees before the introduction of DDT in 1947. The frequent and high usage rates of this compound have caused substantial Pb and As accumulation in agricultural soils. Studies have found that soils with these pollutants have reduced plant growth and productivity because of As rather than Pb toxicity (Creger and Peryea, 1994).

The toxicity of As to non-target plants has become a major problem from an agricultural perspective. Phytotoxicity usually minimizes human dietary intake of affected plants and thus, can indirectly protect humans (Sheppard, 1992). The human toxicity level of As is 200-300 mg (Gorby, 1994). Although it is known that As is a carcinogen, we do not have a risk specific dose.

ARSENIC CHEMISTRY

Arsenic is a metalloid that exists as an anion in aqueous conditions. It can form strong covalent bonds. Under circumneutral conditions, arsenic acid (As (V)) will be mono and divalent anions. However, arsenous acid (As (III)) will exists as a protonated species. There are many studies of the effect of pH and redox potential on As mobility through the environment. Figure 1 illustrates the effect of Eh and pH changes on arsenic speciation.

In aqueous solutions, As usually exists in its trivalent (arsenate) and pentavalent (arsenite)

states. The stable form of As in soils under oxygenated conditions in arsenate. Arsenite predominates in reducing conditions. Arsenite will readily convert to arsenate under oxidizing conditions greater than +100 mV (Onken and Hossner, 1995). Arsenate is known to preferentially form surface complexes (precipitates) with Fe, Al, Mn, and Ca (Cox *et al.*, 1996).

ARSENIC IN SOILS AND BIOAVAILABILITY

In the past, soil levels of As have been studied on a total sum basis in regards to contamination and plant availability. However, these two approaches may not be adequate for characterizing the dynamic nature of As in soil as it relates to uptake by plant roots. The bioavailability of As to organisms is governed more by As speciation than by the total amounts of As present in the soil (Bhumbla and Keefer, 1994). Arsenic in soil can remain in soil solution, be adsorbed on the solid phase, be specifically adsorbed, or precipitate (Cox *et al.*, 1996). Arsenic in soil-water environments can also be present as monomethyl arsenic acid and dimethyl arsenic acid due to chemical and microbial methylation (Marin *et al.*, 1992). Soil pore water can have 0.07 mg of free-ions per L of pore water as the expected no effect value (ENEV). ENEV values are for the most sensitive terrestrial and aquatic receptors (EC, 1998). Arsenic concentrations in contaminated soils can average between 5 to 10 mg/kg and higher when As-containing agrochemicals have been used. For example, in Louisiana, As averages 23 mg/kg in cotton-producing soils. Arsenic availability to plants is usually highest in coarse-textured soils, having little colloidal material and little ion exchange capacity. Availability is lowest in soils high in clay, organic material, iron, calcium and phosphates (NRCC, 1978). The researchers also found that poorly drained soils increased the salinity stress and can enhance the As available to plants.

Cox *et al.* (1996) found that when more As was added to the soil solution, the As remained in soil solution (as opposed to adsorbing until saturation was reached) thereby increasing its availability to plant roots. It was found that when As was adsorbed, other anions were displaced from the limited number of free adsorption sites and transferred to the soil solution. Arsenic is able to do this because it is specifically adsorbed, so it has the ability to displace other specifically adsorbed anions such as phosphorus (P); (Meharg and Macnair, 1994).

Studies by Creger and Peryea (1994) found that the use of fertilizers containing P (such as monoammonium phosphate, $NH_4H_2PO_4$) can increase As solubility and phytoavailability in soils. In contrast, ammonium sulfate does not promote As release. Thus, depending on soil type and the number of adsorption sites, soil chemistry will greatly influence the potential As bioavailability to plants.

ARSENIC UPTAKE BY PLANTS

The uptake of As by plants is influenced by many factors: plant species, concentration of As in soil, soil properties such as pH, redox potential, clay content and the presence of other ions (Marin *et al.*, 1992). As mentioned earlier, plant uptake is dependent on the plant-available fraction of As in the soil. The As in soil solution can be taken up by plant roots from the soil by mass flow and diffusion. Mass flow occurs when the solution of As moves with the convective flow of soil water and diffusion occurs due to random kinetic movement of the ions (Brownian movement) due to the presence of an As concentration gradient produced by root absorption. The amount of diffusable As is the fraction that is considered to be plant available, including both solid-phase As and solution-phase As (Cox *et al.*, 1996).

Many studies have concentrated on specific examples of variable plant As uptake. One study found that plant uptake is affected by mycorrhizal fungi (Benson *et al.*, 1981). Carbonell-Barrachina *et al.* (1997) found that high levels of salinity in soil solution can decrease the uptake of As and subsequently the concentration of As in roots, stem and leaf of the bean plant. Marin *et al.* (1993) confirmed that the redox potential and pH affected the speciation and solubility of As, and thus affecting the phytoavailability and phytotoxicity to rice: the lower the soil redox potential and pH, the greater the amount of water soluble As. Thus, plant tissue concentrations were highest under reduced soil conditions. The low redox conditions also increase the solubility and phytoavailability of monomethyl arsenic acid which affects the absorption of Zn and Cu by rice. The amount of As taken up by rice plants followed this trend regardless of the rate of addition to the soil: dimethyl arsenic acid < As (V) < monomethyl arsenic acid < As (III) (Marin *et al.*, 1992).

Phosphorus is an essential nutrient to plants, and As is known to behave similarly to P in the plant-soil system (Bhumbla and Keefer, 1994). Arsenate can substitute for phosphate (PO_4^{2-}) in metabolic processes and thereby become a toxicant. However, empirical evidence has not all been in agreement since added P can either be deleterious, negligible or beneficial to As uptake in plants depending on specific environmental conditions (Creger and Peryea, 1994). Although As is not to be an essential nutrient for plants, low concentrations of As have been reported to increase growth of maize and potatoes. Marin *et al.* (1992) have suggested that the displacement of soil phosphate by arsenate results in an increased plant P availability, thereby enhancing growth.

It has been found that for plants to reach 1 mg/kg of As on a fresh weight basis, the soil level must exceed 200-300 mg/kg of As (Aten *et al.*, 1980). However, depending on the species, some plants can accumulate higher levels of As at lower soil concentrations. Alfalfa and pasture

grass can accumulate 6-12 mg/kg of As on a fresh weight basis from soils containing only 50-60 mg/kg of As. Thus the bioaccumulation or enrichment factors range from $^{1}/_{200-300}$, $^{6}/_{60}$ and $^{12}/_{60}$. Table 2 summarizes typical As concentrations in some plant species with no evidence of toxicity. The plant As concentrations measured from the south Yukon are well within this range.

ARSENIC MODELLING OF PLANT UPTAKE

Mechanistic models have been developed to predict the nutrient uptake by plants growing in soils (Barber, 1962). These models mathematically describe the soil supply characteristics of the nutrient, root growth, changes in morphological characteristics of the roots, and uptake kinetics of the plant for the nutrient (Barber, 1962). These models rely largely on the soil supply characteristics of the nutrient, and so an essential part of the modelling is to characterize the relationships of the different As species in the soil (Cox *et al.*, 1996). Since As reaction in the soil is very complex to model, researchers have mathematically described the reactions by using P as a substitute. The models generally investigate the relationship of soil solution As and site-exchangeable P.

ARSENIC LEVELS IN PLANT COMPONENTS

Plants grown in uncontaminated areas can accumulate As and distribute it throughout the plant body in nontoxic amounts (NAS, 1977). However, As will accumulate in toxic amounts in plants grown in contaminated soils. Many studies have found that As preferentially concentrates in different plant components. However, As is not readily translocated to shoots and most is found only in the roots. Figure 1 depicts the variation of metal content in different tissues of a single

lodgepole pine, near Sullivan Mine (Pb/Mn), Kimberley, British Columbia. Table 3 shows the As concentration of soils and in the tips of Douglas fir from mineralized regions in British Columbia.

Carbonell-Barrachina *et al.* (1997) found that the highest quantities of As residues were found in the roots, then the leaves and stems, and the smallest concentrations in the fruit and seeds. However, monomethyl arsenic acid can be readily translocated to the shoot and thus reduce rice yields (Marin *et al.*, 1992). As (III) has a high toxicity for radicular membranes, since it readily reacts with sulfhydryl groups in proteins. This usually leads to malfunctioning of the root and cellular death. However, some species like couch grass (*Cnodon dactylon*) are able to bioaccumulate 10850 mg/kg of As in the roots and 1660 mg/kg in the stem before toxicity occurs. Porter and Peterson (1975) also found that *Agrostis tenuis* preferentially accumulated 2080-3470 mg/kg of As in its foliage. They also found that they were tolerant of only the arsenate species and not arsenite.

ARSENIC TOLERANCE AND ADAPTATION

Studies have focussed on the physiology and mechanisms of ion transport across the plasmalemma in microbes which has led to the isolation of mutants with altered physiology. A similar approach to angiosperms would also be useful, but due to difficulties such as screening for mutants within a generation time, does not allow for artificial selection. A better way to screen for mutants in higher plants is to screen plants with an evolved tolerance to As contaminated soils and the evolution of plant adaptation to metal contaminated soils. The plant adaptations should change their normal physiology. Since As behaves as a phosphate analogue, Meharg and Macnair (1994) believe the physiological adaptations to contaminated metals may be due to an adaptation

of their ion uptake systems.

Tolerance to As can be due to two factors. The first is the partial avoidance of As due to ion competition between arsenate and phosphate. The other is a detoxification mechanism that occurs in the root cells of both tolerant and As-sensitive plants, but effective at low As levels. The detoxification mechanism should be able to treat As in such a way that it is unable to interact with phosphate and affect phosphorylation (de Koe and Jaques, 1993). The tolerance of plants to As is believed to be a single major gene for tolerance and one or more modifying genes that interact with the major gene and allow for a heritable variation of tolerant populations. This was shown on *Agrostis castellana* and *Agrostis capillaris* (Watkins and Macnair, 1991) and (de Koe and Jaques, 1993).

Some plants can take up As and not have any toxicity effects. In a study done by Creger and Peryea (1994), apricots did not show any symptoms of As toxicity, even though maximum spikes 1300 mg/kg of As were added to the soil. However, the researchers were unsure of the plants' parentage (may have included nonexpressive cultivars) or whether the plants had not been grown long enough to show symptoms.

Plant uptake systems can be described as either high affinity or low affinity systems. Low affinity systems are defined to be active at substrate concentrations above 100 uM As or P, and high affinity uptake systems with substrate concentrations of 0 -100 uM As or P. For example, if the soil had very low concentrations of As and P, the plant would utilize its high affinity uptake system so that it can take up enough P nutrients. For soils having high levels of P or As, the plant will use its low affinity uptake system to avoid taking up excess P. In the case of soils with As displacing P, the plant will utilize the same, corresponding high or low affinity uptake system. For

example, under a condition of 500 uM P, arsenate can enter the root cells via the low affinity uptake system such that the low flux of arsenate is slow enough for the detoxification mechanism to be effective. De Koe and Jaques (1993) found that this was probably the reason why both tolerant and sensitive populations of plants were both not affected by low external arsenate concentrations. However, the arsenate tolerant clones of the grass *Holcus lanatus* are believed to be able to reduce the amount of influx of As through the suppression of the phosphate uptake system in plant roots. This means that there is a suppressed uptake of both P and As. Since the high affinity uptake system is induced under low plant phosphorus status, to reduce a high arsenate influx, the grasses will inhibit the synthesis of the high affinity phosphate carrier. However, As sensitive plants do not have this mechanism to reduce As uptake under low P conditions (Meharg and Macnair, 1994).

Other grasses, such as *Deschampsia cespitosa* and *Agrostis capillaris,* can become tolerant to high arsenate levels and this tolerance is also due to the reduction of arsenate influx. However, one observation made was that the As tolerance of the uptake system of the grasses can be suppressed by increasing phosphate nutrients to both tolerant and nontolerant plants (Meharg and Macnair, 1994). *Holcus lanatus* (velvetgrass) grows on highly contaminated mine spoil soils in England, accumulated P preferentially to As even when As (extractable) was greater than P (Bensen et al., 1981).

The survival of plants on contaminated mine soils have led some researchers to believe that some plant species have evolved mechanisms of tolerance. Hill (1983) studied the *Cynodon dactylon* that were able to grown on mine waste with up to 1980 mg/kg of As. Benson *et al.* (1981) found that bent grass (*Agrostis tenuis)* and *Agrostis stolonifera* had the ability to grow on

smelter wastes and accumulate As up to 1% of its dry weight.

The tolerance by *Andropogon scoparius* is related to the total arsenic concentration in the soil (de Koe and Jaques, 1993). Porter and Peterson (1977) found that *Agrostis capillaris* on contaminated land had a higher degree of tolerance than plants from non-contaminated areas. The grasses *Holcus lanatus*, *Agrostis capillaris*, *Deschampsia cespitosa* and *Silene vulgaris* have been found to have As adaptive mechanisms. Mine dump studies have been performed in Zimbabwe by Jonnalagadda (1997). Thatch grass (*Pinicum sativum*), couch grass (*Cnodon dactylon*) and mhowa (annual dicotyledonous shrub, *Amaranthus hybridus*) were studied, and only couch grass was able to survive on the mine dumps. However, the remaining two species were tolerant at some distances from the mine dumps.

Meharg and Macnair (1992) observed that As tolerant angiosperms grown in contaminated soils physiologically differed towards other variables also. Tolerant plants have the adaptation to drought due to poor soil structure, soil compactness, low pH and low mineral nutrient status. Thus, a more accurate way of identifying tolerant mutants, is to compare their genotypes. By this method, the researchers found that 65% of the population of *Holcus lanatus* L. had As tolerant individuals, and that the most tolerant individuals having the lowest rates of arsenate influx.

ARSENIC TOXICITY TO PLANTS

Among plants that are non-tolerant, As is very toxic element. There is no universal plant availability index for As which could predict As toxicity or uptake by all plants (Sadiq, 1986). Often, reported phytotoxicity levels of As are very close to the background levels (Sheppard,

1992). The majority of phytotoxicity studies have concentrated on agricultural plants. Table 4 shows the ranges of toxic As concentration in the soil and their corresponding accumulated As in plants. Arsenic toxicity in plants has been described as root plasmolysis and leaf wilting, followed by root discolouration and necrosis of leaf tips and margins. These symptoms indicate that water movement in the plant was limited, resulting in death (Marin *et al.*, 1992).

Arsenite (AsO_2^-) is more toxic to plants than arsenate (AsO_4^{-3}), but both are more phytoxic than organic arsenicals in soil (Carbonell-Barrachina *et al.*, 1997). Arsenite acts by inhibiting photosynthetic carbon dioxide fixation. It has been suggested that the inhibition of light modulation affects the buildup of photosynthetic intermediates during the induction phase of photosynthesis (Marques and Anderson, 1986). Arsenite in plants will inhibit light activation by interfering with the pentose phosphate pathway (Sheppard, 1992).

Many studies have reported different toxicity levels for the same plant or crop. These variations are probably due to differences in soil type, arsenic source and speciation. The source of As in order of increasing toxicity: waste < inorganic < residue < organic. The waste As is usually solid minerals which would be unavailable to plants, and residue As includes the mineralized As from more toxic organic sources (Sheppard, 1992). However, the type of soil for inorganic sources greatly influences arsenic phytotoxicity. The order of increasing toxicity is: clay > loam = sand. The phytotoxicity of organic and inorganic arsenicals are different in plants.

Sheppard (1992) found that monocotyledonus and dicotyledonous crops did not differ in toxicity response to As. In tomato plants, As uptake has been found to reduce the concentration and uptake of some macronutrients (ie. calcium, potassium, magnesium, nitrogen and phosphorus) however, this does not appear to be the mechanism responsible for arsenite toxicity

(Carbonell-Barrachina *et al.*, 1998). The As causes structural damage to the tomato plant, causing a reduction in the translocation of these micronutrients to the higher parts of the tomato and bean plants. Arsenate is known to be a decoupler of phosphorylation in mitochondria and can inhibit leaf uptake of other chemicals. Organic arsenicals is thought to block protein synthesis or some other biosynthetic pathway (Carbonell-Barrachina, 1998).

ARSENIC AND BIOGEOCHEMICAL PROSPECTING

The field of geobotany and biogeochemistry has a rich history of folklore. In areas of exotic overburden (ie. till, glaciofluvial or lacustrine material) that have no geochemical relationship between mineralized bedrock and the overlying material, plants can be used as a prospecting tool. Some vegetation have the capacity to accumulate metals from groundwater with their extensive root system. However, there are only a few species that are true indicators of precious metals. It is common for biogeochemical prospectors to analyze the needles, bark, twigs and leaves of plants for the accumulated metal. Arsenic is an important element in this field of prospecting because it is often associated with gold deposits. It is considered to be a 'pathfinder' element. These elements have the properties that provide anomalies more useable than the element that is sought, and are geochemically associated with that element (Brooks, 1983).

A study by Cohen *et al.* (1987) found that they could look for gold by analyzing balsam fir in the Canadian Shield for anomalous levels of Mo, Sb, Ba and As. Balsam fir has also been used in prospecting for gold in the Sulphide Lake area. Girling *et al.* (1979) found that *Phacelia sericea* (Hook), *Oxytropis campestris* (Hult) and *Sodum lanceolatum* (stone crop) were indicators of As-Au mineralization in British Columbia. They found that these species had

elevated levels 10 - 100 times more than background levels with their efficient uptake systems. However, sometimes the correlation of As and Au was not always coincident. Douglas fir (*Pseudotsuga menziesti*) was found to be a useful species because it can accumulate up to 500 ug/g of As in dry weight in the twigs, where as backgound levels are < 1 ug/g.

A genus that is common in biogeochemical prospecting is horsetails (*Equisetum*). These species have displayed moderately anomalous levels of As (Cohen *et al.*, 1987). Brooks *et al.* (1991) found that horsetail at gold mining sites in Nova Scotia had high tolerance to As. The researcher found that up to 738 ug/g of As in dry weight existed in the plant material. The background As levels for other vegetation was 1 ug/g. Horsetails were the only colonizers of the hostile environment. Table 5 shows that As is a local and indirect indicator of gold in *Equisetum* species.

Sagebrush has also been used as an indicator of gold. Due to its widespread distribution in Nevada and its root extension to groundwater, it is the best As indicator of concealed gold deposits. Plants grown in gold-containing soil contained 4.4 - 6.4 ug/g of As in comparison to 1.6 ug/g dry weight in non-gold, control sagebrush plants. Stewart and McKown (1995) also found that Douglas fir and pine in the Cordilleras had a more consistent pattern of As in vegetation than till or soil over deposits. Studies by Brooks *et al.* (1995) of biogeochemical prospecting involving As have been summarized in the following tables.

Table 6 shows that ashed samples of lodgepole pine (*Pinus contorta*) from Nickel Plate Mine, Hedley, B.C. have unusual enrichment on both the inside and outside of the trees, indicating absorption occurs through the roots and not from airborne contamination. The background levels of As and Au are <5 ug/g and <10 ug/g, respectively. Table 7 shows ashed samples of Labrador

tea (*Ledum groenlandicum*) in a bog near gold mineralization at Jasper Pond, Saskatchewan. Table 8 shows the distribution of As and other elements at different spots of a western hemlock (*Tsuga heterophylla*) branch. Table 9 gives a summary of elemental concentrations in the ash of bark from birch (Betula papyrifera) from an abandoned Cu/Au mine in La Ronge, Saskatchewan. Table 10 show the elemental concentrations in the inner and outer parts of red spruce (Picea rubens) bark from Nova Scotia. Table 11 provides correlation values between Au and As in the ash of bark near Nickel Plate mine, British Columbia

CONCLUSIONS

As the numerous studies indicate, arsenic is a toxic element to plants in all regions of the world. Their uptake from soils is governed mainly by the amount of As in the soil solution and the species it exists in. Other factors can greatly influence As uptake in plants, but are specific to the conditions of the study site and the type of plant. Also due to the variability of influencing factors, a range of levels for phytotoxicity exists for many agricultural plants. Adaptations to high As levels in soils, shown in studies of mine waste sites, show that there are a few species of plants that can tolerate As. These traits are probably genetic in nature and can be passed on to offspring. This tolerance of high As can also be used in the emerging field of As in biogeochemical prospecting for gold. Anomalous levels of As in relation to background levels can allow prospectors to identify areas of gold deposits. Although As occurs naturally around the world, it is yet another toxicant released by the activities of humans and causing pollution in the environment.

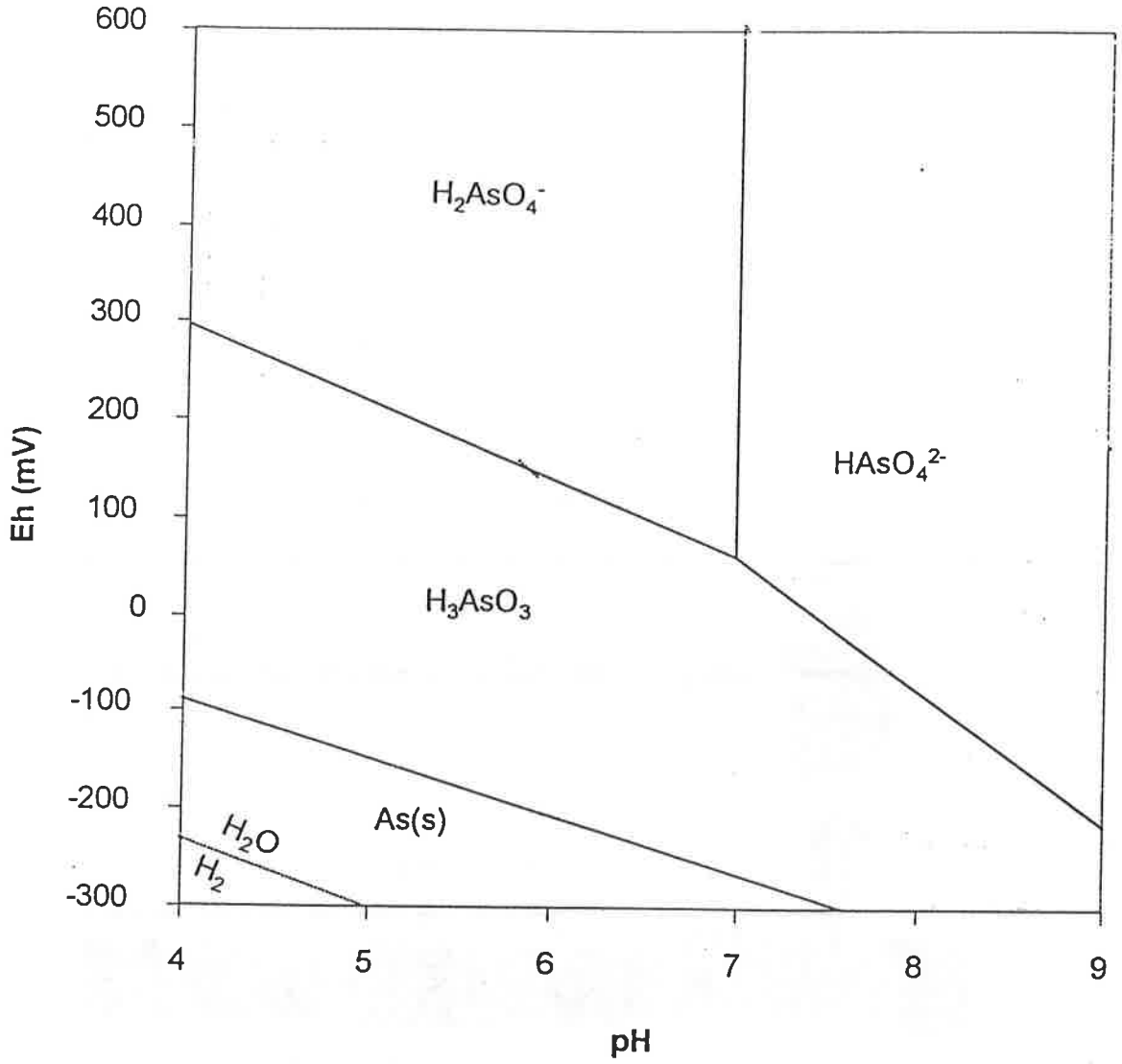
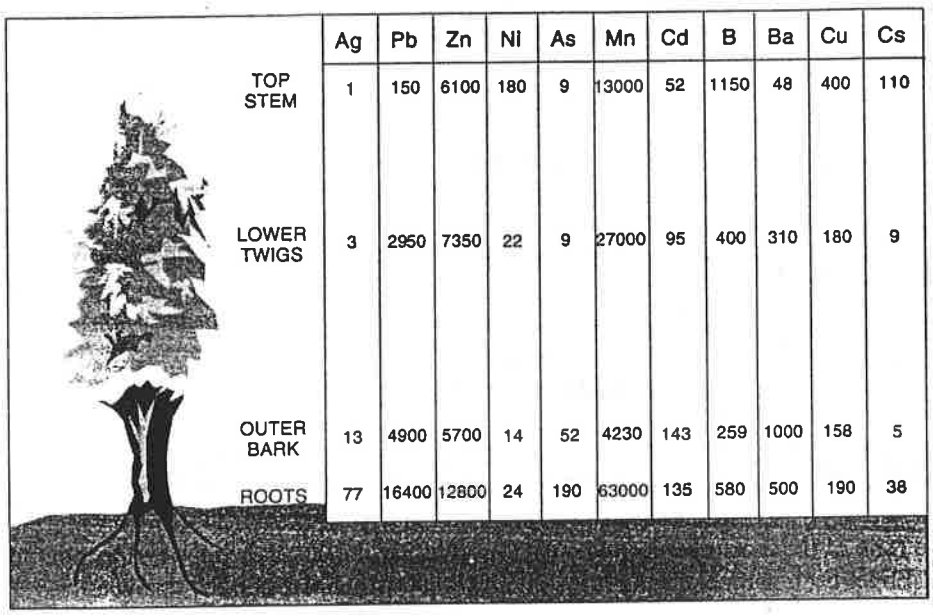


Figure 1: Eh-pH diagram for the system As-H_2O. Source: Masscheleyn et al. (1991)



	Ag	Pb	Zn	Ni	As	Mn	Cd	B	Ba	Cu	Cs
TOP STEM	1	150	6100	180	9	13000	52	1150	48	400	110
LOWER TWIGS	3	2950	7350	22	9	27000	95	400	310	180	9
OUTER BARK	13	4900	5700	14	52	4230	143	259	1000	158	5
ROOTS	77	16400	12800	24	190	63000	135	580	500	190	38

Figure 2: Variations in metal content (ug/g in ash) within different tissues of a single lodgepole pine (Pinus contoria) on sulphie-rich tourmalinite near the Sullivan Mine (Pb/Zn), Kimberley, southern British Columbia. Source: Brooks *et al.* (1995).

Table 1: Calculated Ratios of Arsenic Concentrations in Natural Reservoirs with Respect to soils

Reservoir	Approx. Ratio with Respect to Soil
Rocks	25000
Oceans	4
Soil	1
Biota (plants, humans, microbes)	0.0005
Atmosphere	0.000001

From Bhumbla and Keefer (1994).

TABLE 2: Arsenic Concentrations in Field Collections of Selected Species of Flora

Plant	Concentration (mg As/ kg)	
Colonial bentgrass, Agrostis tenuis		Jenkins (1980)
on mine waste site	1480-3470 (dry weight)	
on low arsenic soil	0.3-3 (dry weight)	
Scotch heaterh, Calluna vulgaris		Jenkins (1980)
on mine waste site	1260 (dry weight)	
on low arsenic soil	0.3 (dry weight)	
Coontail, Ceratophyllum demersum		Jenkins (1980)
from geothermal area, New Zealand	20-1060 (dry weight)	
Ceral grains		NAS (1970)
from arsenic-treated areas	<3 - 252 (dry weight)	
non treated areas	<0.5 - 5 (dry weight)	
Grasses		NAS (1970)
from arsenic-treated areas	0.5 - 60 000 (dry weight)	
non treated areas	0.1 - 0.9 (dry weight)	
Apple, Malus sylvestris	< 0.1 (fresh weight)	
	< 1.8 (dry weight)	
Alfalfa, Medicago sativa		Jenkins (1980)
USA	1.6 (fresh weight)	
smelter area, Montana	0.4 - 5.7 (fresh weight)	
White spruce, Picea alba		Jenkins (1980)
arsenic contaminated soil		
branch	2.8 - 14.3 (dry weight)	
leaf	2.1 - 9.5 (dry weight)	
trunk	0.3 - 55 (dry weight)	
root	45 - 130 (dry weight)	
Pine, Pinus silvestrus, needles		Jenkins (1980)
smelter area, USSR	22 (fresh weight)	
Lowbush blueberry, Vaccinum		NAS (1970)
angustifolium, leaves		
arsenic contaminated soil	6.8 - 15 (fresh weight)	
uncontaminated soil	0.8 (dry weight)	

Table 3: The arsenic content of soils and growing tips of Douglas fir trees from mineralized regions of British Columbia.

District	Arsenic in vegetation (ppm)	Arsenic in soils (ppm)	Plant/soil ratio
Bridge River	10,000	4,633	2.2
Similkameem	8,800	2,980	3.0
Bridge River	2,500	1,072	2.3
Bridge River	50	218	0.2
Bridge River	1,550	39	39.7
Salmo	560	38	14.7
Kimberley	17	14	1.2
Bridge River	33	9	3.7

Source: Warren, Delavault, and barakso (1964) cited in Brooks *et al.* (1995).

TABLE 4: Toxic Concentrations of As in Soils Causing Yield Reduction

Plant/Crop	Toxic Concentration in Soil (mg/kg)	Corresponding Concentration in Plant (mg/kg)
apple	50 - 100	n.a.
apricot	< 50	n.a.
barley	283	n.a.
bean	0 - 414	< 0.02 - 50
blueberry	44 - 70	8 (leaf)
cabbage	50 - 100	2 - 3
cane	2	n.a.
carrot	140	n.a.
cherry	50 - 100	n.a.
corn	0 - 2600	1 - 35
cotton	25 - 196	4
grass	3.2 - 320	2 - 20
hemlock	21 - 42	n.a.
millet	1.4 - 13	n.a.
oat	0 - 850	n.a.
pea	11 - 140	50
peach	30 - 145	1 - 2
pear	50 - 100	n.a.
pine	200 - 1500	13 - 62
potato	45 - 180	33 - 85
radish	2.5 - 500	23 - 93
radish (shoot)	15 - 390	82 - 89
rice	0.5 - 150	n.a.
sedge	1.8	n.a.
soybean	12.5 - 84	1 - 10
spinach	0 - 10	10
spruce	800	5
strawberry	50 - 125	n.a.
tomato	0 - 510	0.7 - 6
vetch	94	n.a.
wheat	1 - 250	270

modified from Sheppard (1991)

Table 5: Antimony, arsenic and gold concentrations (ug/g dry mass) in *Equisetum* species.

Species	Location	Mineralization	As	Au	Sb
E. arvense L.	*Montague Gold Mines, N.S.	Gold	191	<0.04	30
		Gold	179	n.d.	40
	Thetford, Que.	Asbestos	4	<0.3	23
	Bathurst, N.B.	Not known	2	<0.3	22
	Newcastle, N.B.	Coal tailings	<5	<0.8	13
	Ile Haute, N.S.	Not known	<3	<0.4	11
	*Moose R. Mines, N.S.	Gold	72	n.d.	<4
		Gold	185	n.d.	<4
	*Larder Lake, Ont.	Gold	<1	<0.3	4
	*Caribou Gold Mines, N.S.	Gold	222	n.d.	<1
	*Goldenville, N.S.	Gold	138	n.d.	7
	*Salmon R., N.S.	Gold	41	n.d.	<1
	**Amisk L., Flin Flon, Man.	Gold	400	0.008	n.d
E. fluviatile L.	Newcastle, N.B.	Not known	<5	<0.8	21
	Ponhook L., N.S.	Gold	738	<0.5	77
E. hyemale L.	Cape North, N.S.	Gypsum	3	<0.2	147
	Tavastia australis, Finland	Now known	11	<0.2	19
	Mabou Harbour, N.S.	Gypsum	4	<0.1	11
E. palustre L.	Ironside, Que.	Not known	38	<0.4	74
E. scirpoides Michx.	Gaspé Quest, Que.	Not known	10	<0.2	23
	Mabou Harbour, N.S.	Not known	8	<0.2	35
E. sylvaticum L.	Cape North, N.S.	Gypsum	185	<0.3	32
	Thetford, Que.	Not known	6	<0.3	17

n.d. = not determined.
*Field samples.
**Data supplied by Dr. C.E. Dunn.

Source: Brooks *et al.* (1995).

Table 6: Concentrations of gold (ng/g) and arsenic (ug/g) in ashed samples from lodgepole pine (Pinus contorta) from the Nickel Plate Mine, Hedley, B.C. Data show unusual enrichment of metals on the outside and inside of the trees, thus indicating absorption through the roots rather than from airborne contamination. Typical background levels are < 10 ug/g Au and < 5 ug/g arsenic.

Element	Sample	Outer bark	Inner bark	Trunk wood
Gold	Pine #1	420	114	128
	Pine #2	308	28	56
	Pine #3	238	32	36
Arsenic	Pine #1	220	25	59
	Pine #2	160	22	41
	Pine #3	150	20	33

Source: Brooks *et al.* (1995).

Table 7: Elemental concentrations in ashed twigs of nine samples of Labrador tea (Ledum groenlandicum) from nine sites in a bog near gold mineralization at Jasper Pond, Saskatchewan, Canada.

Element	Range	Mean	Background	Conc. factor*
Gold (ng/g)	38-289	129	20	6
Arsenic (μg/g)	2-4	3	2	1.5
Cobalt (μg/g)	5-65	25	5	5
Chromium (μg/g)	12-37	25	10	2.5
Thorium (μg/g)	1.0-3.6	1.8	0.2	9
Uranium (μg/g)	<0.1-3.3	1.4	<0.1	>15

* Mean concentration divided by background.

Source: Brooks *et al.* (1995).

Table 8: Element distribution along branches of western hemlock (Tsuga heterophylla) from the disused Carolin gold mine, southern British Columbia, Canada. Concentrations in ash, determined by INAA (instrumental neutron activation analysis).

	Thick (>10 mm diam.)	Medium (5-10 mm diam.)	Thin (<5 mm diam.)
Au (ng/g)	530	650	1590
As (μg/g)	22	31	82
Cr (μg/g)	32	26	84
Co (μg/g)	11	12	21
Ca (%)	29	24	14
Fe (%)	0.8	1.1	2.3
Na (%)	0.4	0.4	1.1
La (μg/g)	2	3	6
Br (μg/g)	19	18	18
Cs (μg/g)	2	2	2
Sr (μg/g)	430	480	450
Zn (μg/g)	1500	1400	1900

Source: Brooks *et al.* (1995).

Table 9: Concentrations of elements in the ash of bark from paper birch (Betula papyrifera) from near an abandoned Cu/Au mine (Anglo-Rouyn) bear La Ronge, Saskatchewan, Canada.

	Inner	Middle	Outer
Au (ng/g)	10	153	108
As (μg/g)	0.9	18	22
Ba (μg/g)	1700	870	450
Ca (%)	28.7	12.2	5.2
Cr (μg/g)	3	14	38
Fe (%)	0.05	0.71	2.76
Na (μg/g)	506	3410	12000
Rb (μg/g)	190	160	120
Zn (μg/g)	8800	16000	3000
La (μg/g)	2	7	20

Source: Brooks *et al.* (1995).

Table 10: Concentrations of elements in ash of inner and outer bark of red spruce (Picea rubens) from Nova Scotia.

	Tree A		Tree B	
	Inner	Outer	Inner	Outer
Au (ng/g)	<5	51	9	126
As (μg/g)	2	56	93	300
Sb (μg/g)	0.1	10	0.7	3.5
Cr (μg/g)	1	41	7	18
Fe (%)	0.05	1.6	0.22	1.6
La (μg/g)	0.5	16	3	18
Ba (μg/g)	3600	1500	5100	2500
Zn (μg/g)	3300	1600	9200	3900
Ca (%)	30	18	32	28

Source: Brooks *et al.* (1995).

Table 11: Correlations (values of r) between gold and arsenic in bark ash and underlying soils near the Nickel Plate mine, Hedley, southern British Columbia.

Soil horizon	Douglas fir (n=12)		Engelmann spruce (n=13)	
	Au (ng/g)	As (μg/g)	Au (ng/g)	As (μg/g)
Forest litter	0.13	0.10	0.48	0.58
A horizon	0.63	0.63	0.65	0.65
B horizon	0.60	0.55	0.79	0.80
C horizon	0.76	0.64	0.90	0.88

Source: Brooks *et al.* (1995).

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