ASSESSMENT OF PHYTOREMEDIATION AS BEST MANAGEMENT PRACTICE FOR DEGRADED ENVIRONMENTS

Brett Robinson, Steve Green, Tessa Mills, Brent Clothier, Lindsay Fung, Sarah Hurst, Val Snow, Ian McIvor, Siva Sivakumaran, Carlo van den Dijssel and Nola Babbage.

Environment and Risk Management Group, HortResearch, Palmerston North, New Zealand.

ABSTRACT

Phytoremediation, the use of plants to improve degraded environments, is touted as a low-cost, 'clean green' solution to ameliorate degraded lands, and a possible method of mining high-value metals from low-grade ores. The effectiveness of this new technology is, however, variable and highly site-dependent. The decision to implement phytoremediation is usually made on cost. Phytoremediation should compare favourably to the cost of inaction and the best alternative technology. One of major barriers to the use of this technology is uncertainty of its performance on a given site, as compared to 'tried and true' alternatives. A robust assessment of the likely success and cost of phytoremediation for each site may help circumvent this barrier. This assessment necessarily has to consider biogeochemical, climatic and economic variables.

A phytoremediation decision support system (phyto-DSS) was constructed to integrate environmental and economic data to provide a rapid assessment of phytoremediation as an appropriate management practice across various scenarios. The phyto-DSS calculates daily plant water-use, plant metal uptake, leaching and the cost-benefit of the operation. The system requires daily climate data, as well as data on the substrate and the plants that are to be used. An economic assessment is made by comparing the costs of phytoremediation with those of inaction, and the best alternative technology.

The phyto-DSS was used to assess the viability of a commercial phytoremediation operation. Mesocosm and field trials were used to parameterise the inputs. Based on phyto-DSS outputs, phytoremediation was implemented on a 3.6 ha sawdust pile in New Zealand, that is leaching unacceptable amounts of boron into a local stream. In 2000, the site was planted in poplars to control leaching and remove boron from the pile. When the trees are mature, selective coppicing will be used to remove boron from the site. Plant material could even be used as an organic mulch on nearby boron deficient orchards. Leaching has been reduced to three months of the year. The leaching that occurs is collected in a small pond at the foot of the pile and used for irrigation during the summer months. The cost of phytoremediation was US$ 50,000 with annual costs of US$ 5000 for fertiliser and running costs of the irrigation system. The cost of capping the site, the best alternative technology, was estimated at US$ 600,000.

KEYWORDS contaminated sites, phyto-DSS, modelling

INTRODUCTION

Phytoremediation is the use of plants to improve degraded environments. This burgeoning new technology exploits plants as ‘bio-pumps’ that use the sun’s energy to remove water and contaminants from the substrate to the above-ground portions, and return some of the products of photosynthesis back into the root-zone. Transpiration is the cornerstone of phytoremediation. By removing water from the substrate, plants help to reduce erosion, runoff and leaching, thereby limiting the movement of contaminants off-site. Some substrate contaminants are taken-up in the transpiration stream, where they may be metabolised, volatilised or stored. By drying out the substrate profile, the plant roots may also create an aerobic environment where metal mobility is reduced and biological activity is enhanced. Plants stimulate microbiological activity in the root-zone by providing a carbon source from root exudates and decaying root material. It is well documented (Gudin and Syratt 1975; Reilley *et al.* 1996) that substrate micobiota enhance the degradation of some organic contaminants as part of their normal metabolism.

Phytoremediation has several advantages over other remediation and metal extraction technologies. First, and foremost, is the low cost of phytoremediation, which is, in essence, not dissimilar to normal agricultural cropping practices. Competing technologies such as substrate removal, capping and *ex situ* cleansing can cost around $US 1 M/ha, as compared to an estimated US$ 60,000 - 100,000/ha for phytoremediation (Salt *et al.* 1995). Other benefits of phytoremediation include the ultimate fertility of the cleansed site, the high public appeal of ‘green’ technology, and the possibility of producing secondary products that offset the cost of the operation or even produce a small profit.

Huang *et al.* (1991) and Pulford (1995) suggested that phytoremediation could be combined with conventional silviculture, as long as the growth of the trees was unimpeded by the substrate contaminnt. An elevated concentration of contaminants in the wood of the trees is unimportant for human health. Vegetation could also be combusted to produce electricity in a bio-energy operation (Nicks and Chambers, 1994). If a metal hyperaccumulator is used, and the metal is of sufficient value, then the metal could be smelted from the plant-ash and resold. Plants that accumulate essential trace elements such as Zn, Co, and B may be used as an organic mineral supplement to crops, livestock or even humans.

Basic plant physiology, nonetheless, limits the scope of phytoremediation. Only surface contamination can be removed or degraded, and the clean-up is restricted to areas that are amenable to plant growth. Most importantly, it may take a long time for site remediation to be effective. Phytoremediation can only be used if it meets environmental regulation during the operation as well as at its end point.

Plants may provide an exposure pathway for the substrate contaminants to enter the food chain if the plants are consumed (Tibazarwa *et al.* 2001). This will be particularly relevant if plants that are genetically modified to accumulate heavy metals cross-pollinate with crop species. Care has to be taken to avoid such scenarios that could stifle innovation by adding fuel to the anti genetic engineering lobby (Watanabe, 2001).

Phytoremediation suffers from one major drawback in the commercial environment: it can be difficult to predict its cost and performance. Technologies such as capping, ‘dig and dump’ and thermal desorption are not greatly affected by climate, or substrate type. Therefore, per-hectare costings are easily derived and success can essentially be guaranteed. Phytoremediation on the other hand, is very sensitive to climatic conditions, substrate properties and local ecological conditions, such as the presence of pests. This lack of certainty inhibits market acceptance, despite the potential cost savings.

The aim of this study therefore was to develop a phytoremediation Decision Support System (phyto-DSS) that could be used to overcome some of the aforementioned uncertainty. The phyto-DSS should use easily obtainable parameters and provide information on the feasibility of phytoremediation and its economic performance compared to alternative technologies or inaction.

DEVELOPMENT OF THE PHYTO-DSS

To predict the performance of phytoremediation, the affect that planting has on the movement of water and contaminants needs to be calculated. As discussed above, transpiration is responsible for many of the benefits of phytoremediation and therefore forms the foundation of the phyto-DSS. Figure 1 shows a diagrammatic representation of the physical processes considered in the phyto-DSS

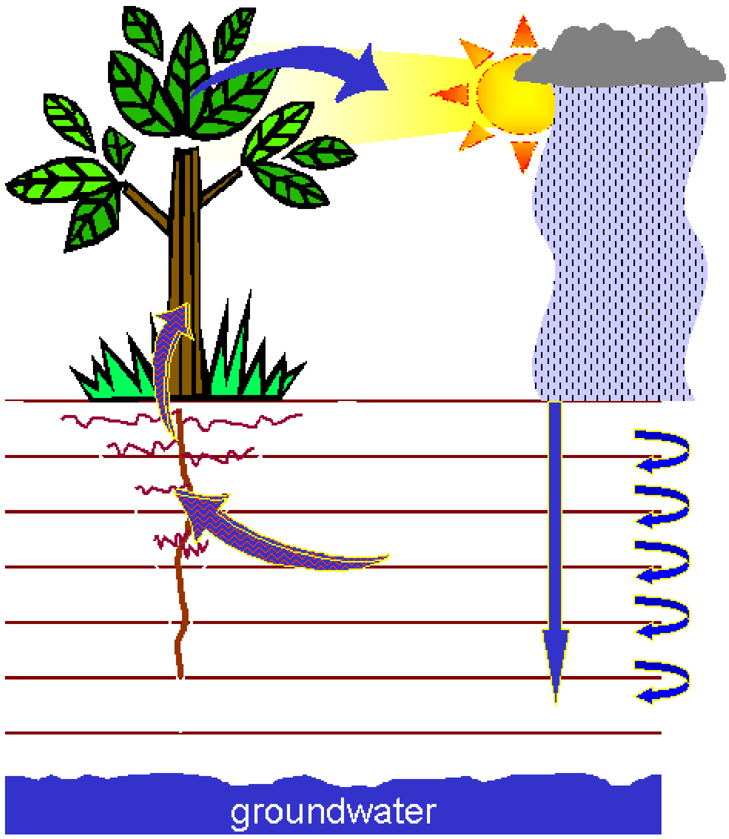

Figure 1 A diagram of the physical process that are considered in the phyto-DSS

The phyto-DSS uses potential evapotranspiration (ETo) that is calculated from solar radiation, temperature, relative humidity and wind-speed using the FAO Penman-Monteith equation (Allen *et al.*, 1998). When there is no water stress, crop evapotranspiration (ETc) is the product of ETo and a crop factor (Kc). Kc is dependent on the species and its stage of development. At canopy closure, the maximum Kc values (Kc_{max}) for most species is between 0.8 and 1.2, with pasture having a Kc_{max} of 1.0 (Allen *et al.*, 1998). The phyto-DSS calculates Kc as a fraction of Kc_{max} depending on the biomass of the crop as a fraction of the biomass at canopy closure, i.e. maximum interception of sunlight. In deciduous trees, Kc is also dependent on the developmental stage of the leaves.

Water availability in the substrate is calculated from the substrate's water retention properties. Water extractable at -1 bar or greater, is considered readily available for plant-uptake. Transpiration is progressively reduced as the substrate dries out and the extractable water falls from -1 bar to -15 bars. Water that is extractable at <-15 bars is considered unavailable for plant-uptake. The phyto-DSS calculates plant-water uptake by dividing the substrate into 1cm slices, each having a specific root density, water retention curve, and water content. The phyto-DSS assumes that water uptake activity is greater in substrate layers that have more readily available water. In times of mild drought where the substrate surface is dry, deeper roots become preferentially more active and the transpiration is not reduced.

The movement of water and solutes in the substrate is calculated using a tipping-bucket approach similar to that described in Hutson and Wagenet (1993). In this method, the substrate is divided into slices, with each slice having a specific water retention properties, water content, solute content and a partition coefficient for the solute in the substrate – substrate solution equilibrium. Rainfall or irrigation that infiltrates the soil, increases the water content of successive soil layers to <0.1 bar, then the 'excess' substrate solution migrates to the next layer down (Figure 1). For any substrate, a percentage of the substrate solution will migrate down macropores of 2 or more soils 'slices' in length.

Plant uptake of substrate contaminants is calculated as a function of transpiration using a model described in Robinson *et al.* (2003a). Here, the roots may actively or passively 'filter' the substrate solution, so that the total contaminant taken up by the plant is the product of the soil solution concentration, total transpiration and a 'root adsorption factor':

$$M(t) \quad = \quad \int_0^{z_R} \int_0^t \quad R(t`, z)\, T(t`)\, C(t`z)\, \phi\left(C\left(t`z\right)\right) \;\; dt \quad dz$$

The average metal concentration in the plant is therefore:

$$[M] = M / B$$

where M = the amount of metal taken up by the plant at time t, [M] is the average plant metal concentration (mg kg^{-1}), B equals above-ground dry biomass (kg), z_R = the depth of the root-zone (m), z = depth (m), t = time (days), $R(t`,z)$ = root density fraction [(root mass at depth z) / (total root mass) at time t], T = water use (L day^{-1} at time t), C_z =

concentration of metal in soil solution (mg L^{-1} at depth z and time t), ϕ= root absorption factor for the metal at depth z and time t.

USE OF THE PHYTO-DSS: A CASE STUDY

The following case study, taken from Robinson *et al.* (2003b), describes how the phyto-DSS was used to predict the success of phytoremediation on a contaminated sawdust pile.

The Kopu sawdust pile is located at the base of the Coromandel peninsula, North Island, New Zealand (37.2^{o} S, 175.6 ^{o} E). The pile has a surface area of 3.6 ha and an average depth of 15 m. Over a 30-year period, from 1966, sawdust and yard-scrapings from timber milling in the region were dumped on the pile. Land around the pile has been engineered so that no surface or ground water enters the pile, and all leachate resulting from rainfall is collected in a small holding pond at the foot of the pile. In the past, vegetation has failed to establish and evaporation from the surface of the pile has been negligible, even in the summer months. This is demonstrated by the presence of saturated material at depths as shallow as 20 mm.

Leachate resulting from the annual rainfall of 1135 mm, as measured at a nearby meteorological station at Thames, regularly caused the holding pond to overflow and enter a local stream. This overflow elevated B concentrations in the stream to levels that were in excess of 1.4 mg/L, the New Zealand Drinking Water Standard (NZDWS), especially in the summer months when stream flow was low. In response to these breaches, the local environmental authority placed an order on the forestry company responsible for the site that the problem be remedied.

In July, 2000 a one ha trial was established on the Kopu site using 10 poplar and willow clones as well as two species of *Eucalyptus.* Two *Populus deltoides* hybrid clones were then chosen as the best candidates for phytoremediation based on survival, biomass production and B uptake. The following year, the remainder of the pile was planted in these two clones at a density of 7000 trees/ha. Fertilisers were periodically added to the trees and a pump was installed near the holding pond at the foot of the pile for irrigation during the summer months.

A concurrent lysimeter experiment was set up at HortResearch, Palmerston North, to derive a model that could be used to predict the uptake and leaching of water and contaminants at the Kopu site. Details of this experiment can be found in Robinson *et al.* (2002).

Figure 2 Photographs of the Kopu sawdust pile taken at planting in July 2000 and after 2 years growth in April 2002.

Figure 2 shows tree growth on the Kopu pile over the first two years. Approximately 30% of the trees are two years old, and the remainder are only one year old. Figure 2 demonstrates clearly how phytoremediation helps the contaminated site become part of the landscape by transforming the bare pile into an actively growing “green” plant cover. The estimated above-ground biomass production in the first and second years of the trial was 1.2 and 13.3 t/ha dry matter, respectively. The average leaf area index of the two-year-old trees was 2.6, which is well below the value of 6 reported by Heilman *et al.* (1996) for maximum photosynthesis and transpiration of poplar under high light conditions. We would not expect the water use of the trees to be maximised until the biomass exceeds 30 t/ha on a dry matter basis. This level of productivity should occur in the 3rd or 4th years.

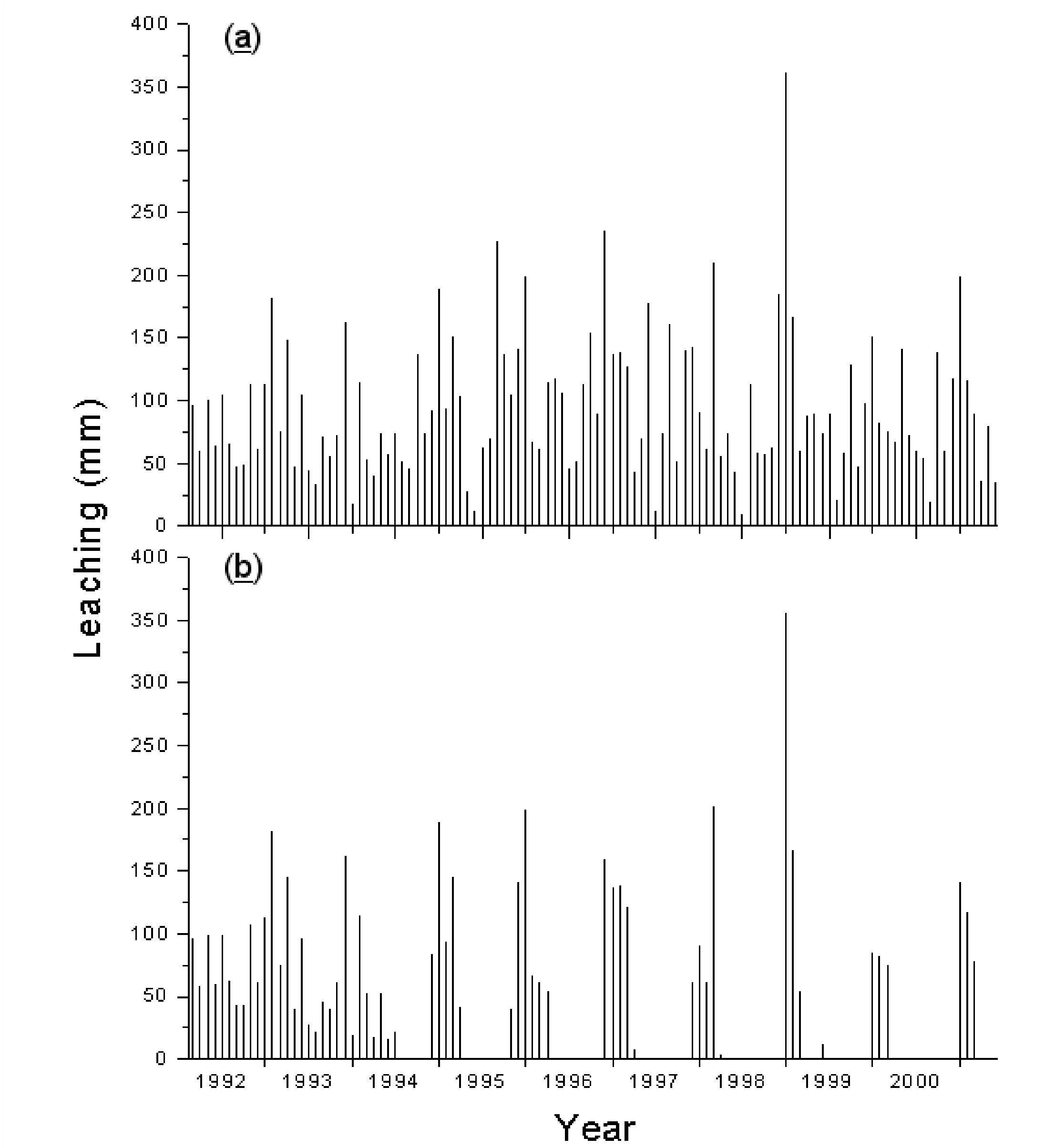

Figure 3 Estimated monthly leaching (mm) from the Kopu sawdust pile (a) without trees and (b) with trees. Meteorological data was used from the nearby town of Thames from 1991 to 2000. It was calculated that maximum transpiration would be achieved after three years

The monthly water balance of the pile was calculated using the phyto-DSS. Daily weather data was taken from a meteorological station at nearby Thames. Some parameters for the phyto-DSS (water use efficiency, crop coefficient) were derived from the lysimeter experiment. Field measurements at Kopu were used to estimate plant root distribution, and the leaf area index. The water retention curve for the sawdust was measured using a combination of Haines' apparatus and pressure plates. Disk permeameters were used *in situ* to derive the hydraulic conductivity close to saturation. Phyto-DSS calculations of leaching are shown in Figs. 2a and 2b respectively. As expected for such a high rainfall site, the bare pile leaches a considerable amount of drainage water through all months of the year (Figure 3a) The impact of trees, is to substantially reduce the drainage of water during the summer months when the trees are fully leafed and transpiring at their maximum. The summer months are of greatest concern for contamination of the local waterways because stream flows are lower and there is less dilution of the contaminants. The reduced leaching that occurs during the winter months can be irrigated onto the trees in times of drought during the summer, or alternatively, released into a nearby stream at times of high flow when the risk of exceeding the New Zealand Drinking Water Standard (NZDWS) is minimal.

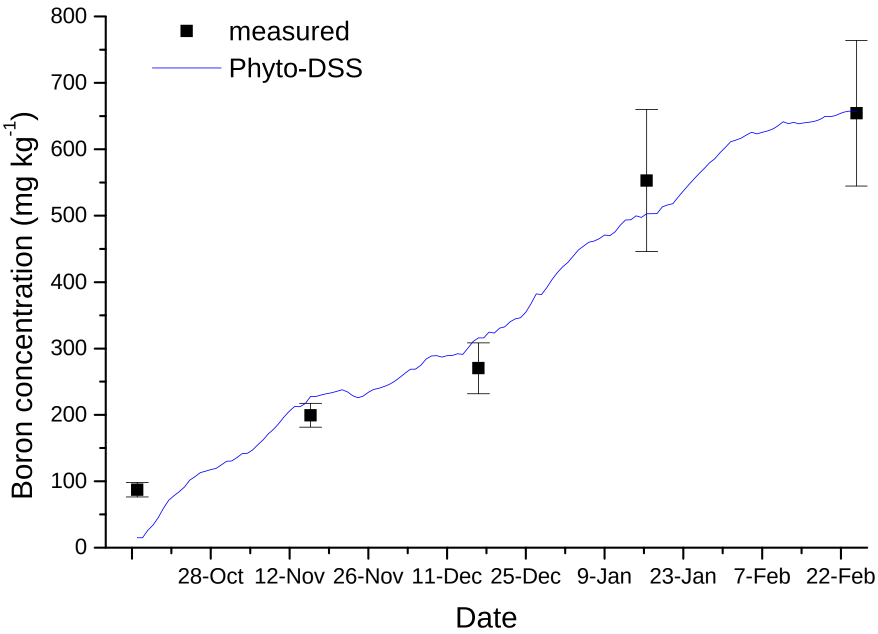

Figure 4 The concentration (mg/kg dry mass) of B in poplar leaves from the lysimeter experiment during the 2001-02 season. Values are averages of four trees. Bars represent the standard error of the mean. The line represents concentrations calculated by the phyto-DSS.

Table 1 shows some lysimeter results of the total concentrations of Cu, Cr, As and B in the sawdust, as well as the average concentration in the leachates over the duration of the experiment. Boron was the only detectable contaminant in the leachate that exceeded the New Zealand Drinking Water Standard (NZDWS). The level of As in the leachate was below detection limits (0.1 mg/L) but could still have exceeded the NZDWS for As (0.01 mg/L).

Table 1. From Robinson *et al.* (2003b). Average metal concentrations in leachate (mg/L) sawdust (mg/kg dry mass) and leaves (mg/kg dry mass) from four lysimeters. Leachate concentrations represent the average of monthly values collected throughout the experiment. Values in parentheses are the standard error of the mean.

Element	NZDWS[A]	Leachate	Sawdust (initial)	Leaves (final)
Cu	2	0.30 (0.07)	140 (25)	6.6 (1.7)
Cr	0.05	0.03 (0.01)	15 (0.8)	4.9 (1.2)
As	0.01	<0.1	6.3 (0.8)	<1
B	1.4	1.87 (0.10)	39.9 (2.7)	654 (109)

[A]New Zealand Drinking Water Standard (Drinking-Water Standards for New Zealand 2000).

Poplar leaves contained Cu and Cr concentrations that were on average 6.6 and 4.9 mg/kg dry mass respectively (Table 1). Arsenic concentrations were below detection limits (1 mg/kg). Results from the lysimeter experiment revealed that poplar have the capacity to accumulate significant amounts of B in their leaves (Figure 4). This trait has previously been reported for poplar by Bañuelos *et al.* (1999). At the end of the growing season, the average leaf-B concentration was nearly 700 mg/kg on a dry matter basis, over 28 times higher than the B concentration in the sawdust (40 mg/kg dry matter). The amount of B extracted by the poplar was predicted using the phyto-DSS. There was very good agreement between the calculated and measured B concentration in the leaves of the lysimeter trees (Figure 4). The B concentration in the leaves of the trees at the Kopu site has also been estimated using parameters from the lysimeter experiment. Figure 5 shows estimated B concentration against the actual measured B concentration in the leaves. Once again, there was a reasonable correlation ($r=0.74$ $p<0.01$) between the estimated and measured values.

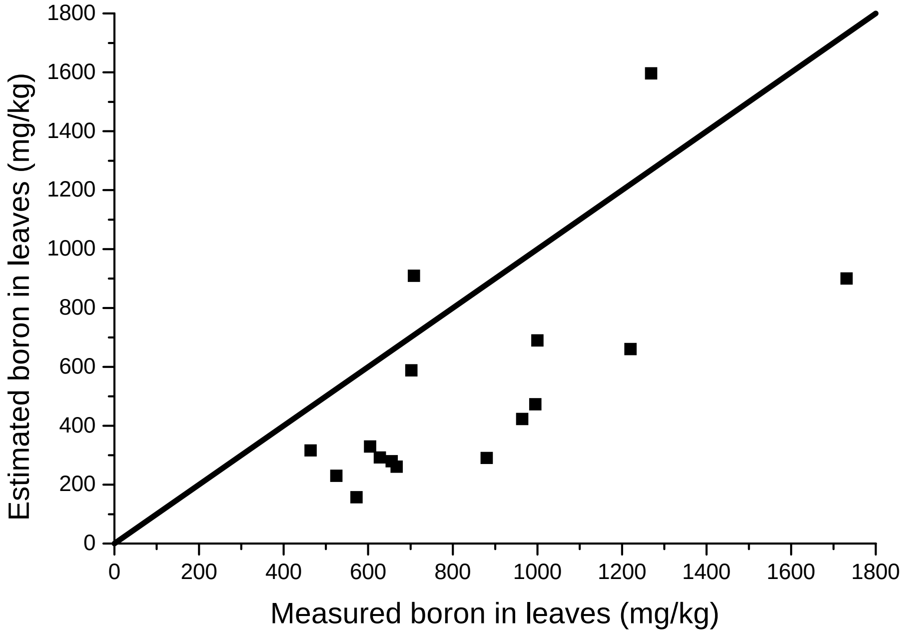

Figure 5 Measured vs phyto-DSS calculated B concentrations (mg/kg dry mass) in leaves from the trees growing on the Kopu sawdust pile at the end of the 2001-02 season. The solid line represents y=x.

The results indicate that in addition to controlling leaching at the site, poplars may also be able to reduce the B loading by phytoextraction. Unless the trees are harvested, most of the B is returned to the sawdust via leaf-fall. Harvested material could, however, be used as an organic B supplement to trees in orchards that are B-deficient in other parts of the country. The concentrations of other heavy metals in the leaves (Table 1) are unlikely to cause further environmental problems.

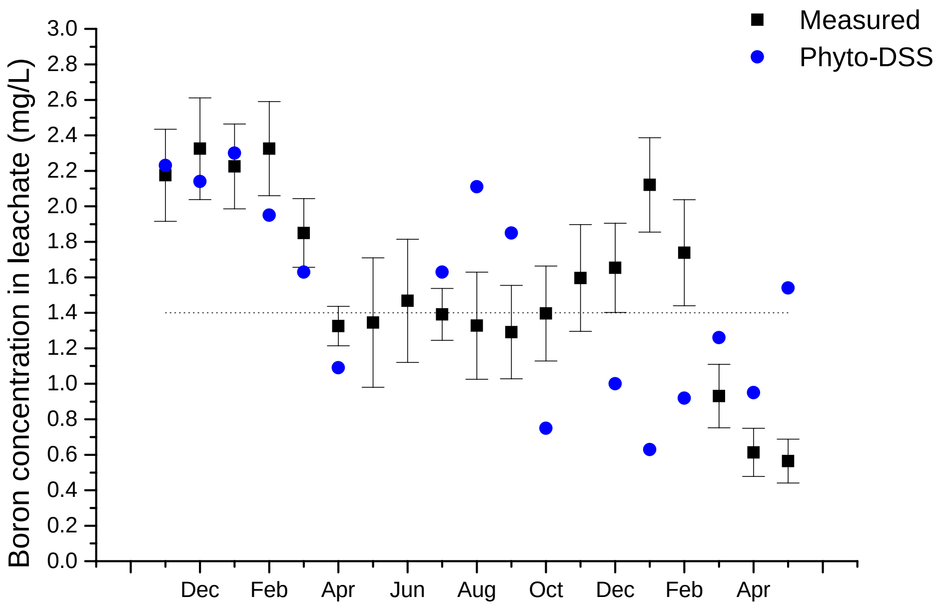

Figure 6 Concentration (mg/L) of B in the drainage of the lysimeters from November 2000 until May 2002. Values are averages from four lysimeters. Bars represent the standard error of the mean. The circular dots represent values calculated by the phyto-DSS.

The average B concentration in the drainage water from the lysimeters decreased progressively during the course of the experiment. After two years the B concentration dropped below the NZDWS (1.4 mg/L Figure 6). This decrease occurred late in the growing season when tree water uptake was at a maximum. The phyto-DSS also calculated a decrease in the B concentration of the drainage. There have been insufficient measurements on the leachate from the Kopu sawdust pile to determine how our predictions compare to the field situation. Further measurements are planned over the next few years to determine the success of phytoremediation at Kopu.

The phyto-DSS calculated that phytoextraction satisfies environmental legislation on this site. The total cost of phytoextraction over a ten-year period will be US$ 170,000 including a site maintenance plan over 5 years. Half of this total cost was taken up as site assessment, involving scientist time to conduct the plant trial and chemical analysis. Selective coppicing will allow B to be removed from the site. Harvested material could be used as an organic B-rich mulch on nearby avocado orchards that are deficient in B. The net-present cost of capping the site, the best alternative technology is US$950,000.

CONCLUSIONS

Phytoremediation is the best management practice for degraded lands only if it the most cost-effective long-term option that satisfies environmental legislation. Phytoremediation offers the possibility of a low-cost cleanup for a wide range of contaminated sites. The technology offers long-term solutions, but is constrained by basic plant physiology in the types of environmental degradation that can be ameliorated. Combining phytoremediation with the production of saleable products may circumvent phytoremediation's Achilles heel, which is the time needed for remediation.

The phyto-DSS can be used to provide a rapid assessment of the feasibility of phytoremediation, given climatic and edaphic data. Further experiments will be needed to determine the accuracy of the phyto-DSS on different sites. Models will doubtlessly be improved as we improve our understanding of plant-substrate interactions in degraded environments.

Modelling cannot, however, consider all variables that affect phytoremediation. This technology is site-specific and some outcomes, such as the growth and performance of various species or varieties, cannot be calculated using models. These have to be determined experimentally, and this adds cost to phytoremediation. Nevertheless, where it can be applied successfully, phytoremediation is still a small fraction of the cost of most competing technologies. The challenge for scientists and environmental engineers is to demonstrate to regulators that for many degraded environments, phytoremediation is the best management practice.

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