DOI: 10.1007/s00267-004-0245-0

Estimating the Unknown Components of Nutrient Mass Balances for Forestry Plantations in Mine Rehabilitation, Upper Hunter Valley, New South Wales, Australia

A. M. MERCURI
Ecosystem Management and
Agronomy and Soil Science University of New England
Armidale NSW, 2351, Australia

J. A. DUGGIN
Ecosystem Management
University of New England
Armidale NSW, 2351, Australia

H. DANIEL*
P. V. LOCKWOOD
Agronomy and Soil Science
University of New England
Armidale NSW, 2351, Australia

C. D. GRANT
Ecosystem Management
University of New England Armidale NSW, 2351, Australia
and Environment Department
Alcoa World Alumina Australia
Applecross WA, 6153, Australia

ABSTRACT / Commercial forestry plantations as a post-mining land use in the Upper Hunter Valley of New South Wales, Australia are restricted by both the poor nutrient availability of mining substrates and low regional rainfall. An experiment was conducted to investigate whether municipal waste products and saline groundwater from coal mining operations could improve early tree growth without impacting on the environment through salt accumulation and/or nutrient enrichment and changes in groundwater quality. Potential impacts were investigated by quantifying the nutrient cycling dynamics within the plantation using an input–output mass balance approach for exchangeable calcium (Ca^{2+}), exchangeable magnesium (Mg^{2+}), exchangeable potassium (K^{+}), exchangeable sodium (Na^{+}), nitrogen (N), and phosphorus (P). Measured inputs to and outputs from the available nutrient pool in the 0–30 cm of the overburden subsystem were used to estimate the net effect of unmeasured inputs and outputs (termed "residuals"). Residual values in the mass balance of the irrigated treatments demonstrated large leaching losses of exchangeable Ca, Mg, K, and Na. Between 96% and 103% of Na applied in saline mine-water irrigation was leached below the 0–30-cm soil profile zone. The fate of these salts beyond 30 cm is unknown, but results suggest that irrigation with saline mine water had minimal impact on the substrate to 30 cm over the first 2 years since plantation establishment. Accumulations of N and P were detected for the substrate amendments, suggesting that organic amendments (particularly compost) retained the applied nutrients with very little associated losses, particularly through leaching.

Open-cut coal mining leases in the Upper Hunter Valley of New South Wales (NSW), Australia cover over 61,000 ha, of which 16% requires rehabilitation (Andrews 1999). Commercial forestry plantations have been suggested as an alternative postmining land use to the current regional practice of returning mined land to improved pasture and grazing. However, previous tree plantings on coal mine overburden have shown low productivity because mining substrates have low fertility and a poor moisture-holding capacity in combination with low rainfall (Burns 1987; Ryan 1995; Hannan and Gordon 1996). Tree growth rates could be increased using irrigation and nutrients that are locally available in the form of saline mine water and municipal waste products such as green waste composts, sewage sludge (biosolids), and sewage effluent.

Plant-available substrate nutrients come mainly from rainfall deposition and chemical/biological transformations of the substrate into available forms

KEY WORDS: Coal mine rehabilitation; Salinity; Input–output analysis; Nitrogen; Phosphorus; Basic cations; Substrate nutrient storage

Published online February 6, 2006.

*Author to whom correspondence should be addressed; *email:* hdaniel@une.edu.au

(such as by weathering and mineralization). In forestry plantations, nutrient inputs can be supplied through silvicultural additions (such as fertilizers and irrigation) and lost from the substrate subsystem by leaching and plant uptake. Studies of mass balance (input–output) models reported in the literature [e.g., Likens and Bormann 1995 (and previous work from the Hubbard Brook Forest Ecosystem); Keolsch and Lesoing 1999; Ranger and Turpault 1999; Bindraban and others 2000; Pare and others 2002] were used to determine whether nutrients were accumulating (inputs > outputs) or being lost from the substrate (inputs < outputs). An understanding of such nutrient pathways enables an assessment of any potential hazards such as accumulations in, or losses from, the substrate subsystem that will ultimately affect the quality of the media and, hence, tree productivity. In addition, losses to the environment via leaching, particularly under saline irrigation, might affect groundwater quality.

A study was established in June 2000 to measure the growth of four relatively salt-tolerant tree species on rehabilitated coal mine overburden over 2 years, when irrigated with saline mine water and amended with the municipal waste products of green compost or biosolids or using current industry amendments (topsoil and fertilizer) and to assess the environmental implications of using such products (fate of applied salts and nutrients). The aim of this article is to examine plantation nutrient cycling dynamics by determining nutrient input–output mass balances (Ca^{2+}, Mg^{2+}, K^+, Na^+, N, and P) in the substrate subsystem for each treatment and to use the results to assess the environmental impacts of the silvicultural techniques for plantation establishment. Tree growth results and changes in substrate salinity ($EC_{1:5}$) have been reported by Mercuri and others (2005).

Methods

Site Description and Experimental Design

The study was conducted between June 2000 and June 2002 on a rehabilitated mine overburden site at the Drayton Colliery, 10 km southeast of Muswellbrook (latitude 32°15′ S and longitude 150°53′ E), New South Wales, Australia. The climate of the area is temperate, with summer-dominated rainfall averaging 645 mm per annum.

The study site was representative of overburden waste piles from the mine and was located within 2 km of the saline mine-water storage facility for ease of irrigation. The site had been partially rehabilitated about 10 years earlier by backfilling with 3–10 m of overburden, topsoil respreading, and pasture establishment (Simpson, personal communication, 2000). The experiment was set out as a split–split plot design arranged as a randomized complete block with two replications. Main treatment plots were either irrigated with saline mine water or rainfed. Size and cost constraints precluded any option to increase replication of the main treatment. The amendments (subplots) were overburden (no treatment), topsoil, fertilizer, green waste compost, and biosolids. Within each amendment application plot, subsubplots of 4 species (*Corymbia maculata*, *Eucalyptus botryoides*, *E. tereticornis*, and *E. occidentalis*) were planted in 4 rows of 10 trees at a 3 × 3 m spacing.

Irrigation with saline mine water (EC = 2.54–4.36 dS/m) was applied through drippers placed 10 cm on either side of each tree. The dripper irrigation tube had a nominal discharge rate equivalent to rainfall of 0.71 mm/h. Irrigation scheduling was determined by WATCHECK, a model customized for this study by CSIRO Forestry and Forest Products. It is similar to the WATLOAD model of Myers (1992) but includes site-specific climatic information and a leaching fraction. The model determined the soil water availability based on water cycle variables (rainfall, irrigation, and evaporation), forest stand development, and leaching requirements. Irrigation was undertaken if the soil water deficit in the upper 10 cm exceeded –20 mm. Sufficient water was applied to bring the water availability to field capacity plus an additional amount (leaching fraction) for adequate leaching of excess salts from the root zone. The leaching fraction was determined to be an additional 20% of that required to reach the desired water balance (Polglase and Myers 1999). Irrigation water samples were taken weekly to determine the solute loads added through irrigation. Each sample was analyzed for cations (Ca, Mg, K, and Na) and nutrients (total N and total P) using standard analytical methods (American Public Health Association 1981).

During mining, scraping of topsoil is recommended to the average depth of the A horizon (~10 cm) and often results in the incorporation of some B-horizon material (Burns 1987). In preparation for this study, the returned topsoil was stripped and stockpiled to expose underlying overburden to allow for the application of all experimental treatments. A pre-emergent herbicide (Simazine) was sprayed to control weeds.

The stockpiled topsoil was reapplied for the topsoil treatment using the current rehabilitation strategy (10 cm depth or 1100 tons/ha). Biosolids were sourced from the Hunter Water Corporation sewage waste treatment facility and comprised the solid component remaining from tertiary treatment of sewage wastewater. Municipal waste compost was sourced from the

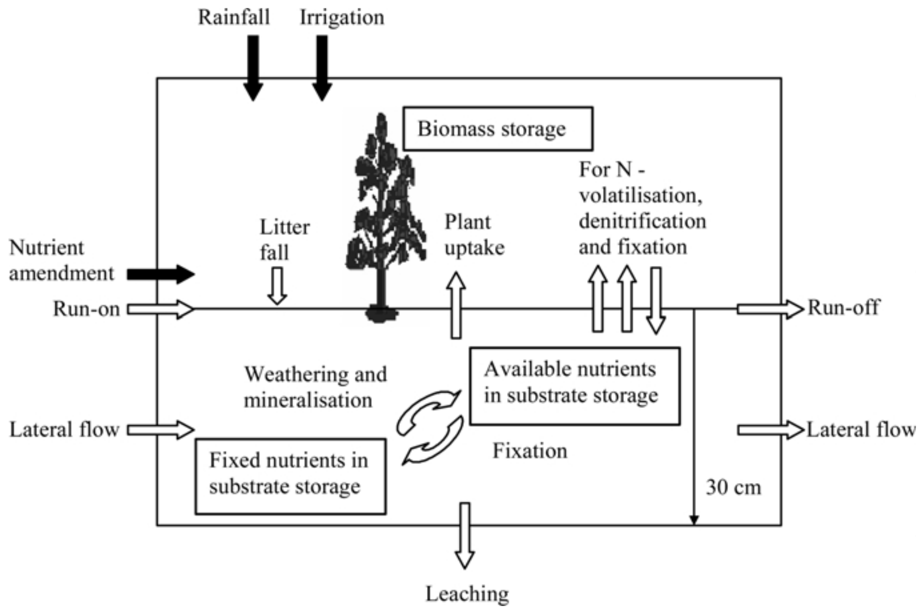

Figure 1. A schematic model for the soil substrate subsystem showing inputs and outputs to the available nutrient pool in the top 30 cm of overburden. Solid arrows and boxes show the measured variables. For small plot studies, run-on, runoff, and lateral groundwater flows are assumed to balance. Litterfall in this study made a negligible input to the soil substrate subsystem, and a recognizable litter layer did not form. Unmeasured (residual) inputs include weathering and mineralization, and for N, fixation. Unmeasured (residual) outputs include leaching, and for N, denitrification and volatilization.

Muswellbrook Shire Council and was a combination of garden green waste cocomposted with sewage sludge.

Fertilizer [167 kg/ha diammonium phosphate (DAP)], compost (3 cm depth or 183 tons/ha), and biosolids (0.98 cm depth or 60 dry tons/ha) were applied at rates recommended by Polglase and Myers (1999), based on industry standards. Biosolids, compost, and topsoil were spread over each area using a Bobcat to even out the material. After all amendments were spread, the substrate was deep ripped to a standard depth (600 mm) in preparation for planting. Fertilizer was applied as 150 g DAP, by placing it directly into each tree planting hole.

Substrate and Plant Measurements

Substrate samples were taken for analysis prior to plantation establishment in June 2000 and 2 years after plantation establishment (June 2002). Three composite surface samples at two depths (0–10 and 10–30 cm) were collected from each plot, air-dried, and passed through a 2 mm sieve. Due to the depth of the applied topsoil (10 cm), three substrate depths were collected from this amendment treatment. These were 0–10 cm (the respread topsoil treatment) and 10–20 cm and 20–40 cm (considered to reflect the 0–10 cm and 10–30 cm samples from the other treatments). Samples were analyzed for exchangeable Ca, Mg, K, Na, total N, total P, and extractable P, using standard laboratory techniques (Rayment and Higginson 1992). Nutrient content (kg/ha) was calculated using a measured overburden bulk density of 1600 kg/m^3. This articles focuses only on the above-listed nutrients, although the accumulation and changes in soil electrical conductivity were also highly important due to salt additions from irrigation. Changes in substrate salinity ($EC_{1:5}$) are reported by Mercuri and others (2005).

In June 2002, all above-ground components from two trees were removed from each sub-subplot. Samples were dried by components (stems, branches, and leaves) to constant weight at 70°C. Regression equations were developed for each species using height and diameter at 30 cm (D_{30}) to estimate individual tree aboveground biomass (kg/ha) for the 24-month measurement. Nutrient concentrations were determined on samples collected for biomass estimation. Subsamples were finely ground, digested in a sealed container, and analyzed for Ca, Mg, K, Na, and P using inductively coupled plasma (ICP) spectroscopy (Anderson and Henderson 1986). Total N was determined by Kjeldhal digestion (Page and others 1982). Results were expressed as aboveground stored nutrients (kg/ha) for each treatment.

Substrate (0–30 cm) Nutrient Mass Balances

The differences in nutrient cycling of exchangeable Ca, exchangeable Mg, exchangeable K, exchangeable Na, total N, total P, and extractable P were assessed for each treatment of the substrate (0–30 cm) mineral fraction (<2 mm) using a mass balance approach (Figure 1). The change in substrate nutrient storage (ΔSS) is a balance between all measured and unmeasured (residual) nutrient inputs and all measured and unmeasured (residual) outputs (Equation 1). The term "residual" is used to refer to the sum of unmeasured inputs and outputs.

$$\begin{aligned}\Delta \mathrm{SS(kg/ha)} = {} & [\text{measured inputs (kg/ha)} \\ & + \text{residual input}] \\ & -[\text{measured outputs (kg/ha)} \\ & + \text{residual output}]\end{aligned} \tag{1}$$

The schematic model for this experiment (Figure 1) included the inputs of rainfall, irrigation, nutrient addition from amendment application, net mineralization (e.g., mineralization – immobilization), surface and subsurface lateral flow, weathering of substrate minerals, and decomposition of plant biomass (litterfall). The outputs were plant uptake into the aboveground and belowground biomass, surface and subsurface lateral flow, formation of secondary minerals or fixation into unavailable forms, and leaching losses to below 30 cm. Not all of these processes were measured or could be reliably estimated (e.g., rainfall input and belowground biomass), whereas others were considered to balance each other (e.g., run-on and runoff) or were excluded, as they were deemed to be negligible (Table 1).

The total pool of available nutrients in the substrate consists of all ions adsorbed to colloids and dissolved in the substrate solution (Bormann and Likens 1967). However, data are only available for the exchangeable forms of the cations and only total N was measured. No conclusions/predictions can be made regarding the total cation pools. The total and extractable P was used for the P balances.

Amendment samples were collected prior to application and were analyzed for particle size distribution following the pipette method of Gee and Bauder (1986). Overburden bulk density was based on results of baseline sample particle size analysis being subjected to a packing model (Gupta and Larson 1979). Bulk density of compost, biosolids, and freshly spread topsoil was estimated by measuring the weight of the <2 mm sample required to fill a 10 mL volumetric flask. The bulk densities were 610, 610, and 1100 kg/m^3, respectively.

The mineral nutrient pool (kg/ha) prior to the experiment commencing (initial baseline overburden) and after 2 years was determined for each element, to calculate the change in nutrient storage (ΔSS) over the 2 years of the experiment (initial to final). Values were calculated using the soil sample depth in meters (d), estimated soil bulk density in kilograms per cubic meter (BD), nutrient concentration in milligrams per kilogram (N), and the percentage of fines <2 mm (F), according to the following equation:

$$\begin{aligned}\text{Mineral nutrient pool (kg/ha)} &= (d \times \mathrm{BD}) \\ &\times (N \times 0.000001) \times (F/100) \times 10{,}000\end{aligned} \tag{2}$$

Equation 2 was also used to calculate the nutrient input provided by each substrate amendment (Table 1). Mineral substrate nutrient storage was reported as profile storage (0–30 cm in kg/ha) by adding nutrient pools from 0 to10 cm and from 10 to 30 cm using appropriate proportions.

The topsoil treatment was estimated differently than all other amendment treatments. This treatment had three separate substrate sampling compartments, as outlined earlier. Although the biosolids and compost applied at the start of the experiment had disintegrated and mixed in with the underlying 0–30 cm overburden profile by year 2, the topsoil was still obvious, resulting in the need to maintain a separate and distinct compartment (0–10 cm). A separate mass balance of inputs and outputs for the two compartments (0–10 cm, topsoil applied; 10–40 cm, underlying overburden) was required. As plant uptake could have come from either compartment (referred to as uptake$_{(a)}$ and uptake$_{(b)}$, respectively), the input–output balances cannot be solved for each individual compartment—only for the entire 0–40 cm profile. The corresponding soil nutrient pool budgets for the 0–10 cm topsoil (Equation 3) and underlying 10–40 cm substrate (Equation 4) compartments are

$$\begin{aligned}\Delta \mathrm{SS}_{0-10\ \mathrm{cm}}\ & \text{(kg/ha)(initial topsoil} \\ & - \textit{year 2 topsoil}_{0-10\ \mathrm{cm}}) \\ & = \text{inputs (rainfall} + \text{irrigation} \\ & + \text{input residual}_{0-10\ \mathrm{cm}}) \\ & - \text{outputs (plants uptake}_{(a)} \\ & + \text{output residual}_{0-10\ \mathrm{cm}})\end{aligned} \tag{3}$$

$$\begin{aligned}\Delta \mathrm{SS}_{10-40\ \mathrm{cm}}\ & \text{(kg/ha)(baseline overburden} \\ & - \text{year 2 topsoil}_{10-40\ \mathrm{cm}}) \\ & = \text{inputs (output residual}_{0-10\ \mathrm{cm}} \\ & + \text{input residual}_{0-10\ \mathrm{cm}}) \\ & - \text{outputs (plants uptake}_{(b)} \\ & + \text{output residual}_{10-40\ \mathrm{cm}})\end{aligned} \tag{4}$$

The overall mass balance for the 0–40 cm profile in the topsoil treatment is the combination of Equations 3

Table 1. Components of the input–output mass balance for the saline irrigation and substrate amendment plantation forestry experiment at Drayton Colliery, Hunter Valley NSW

Variable	Measurement or prediction	Assumptions
Inputs		
Irrigation water	Irrigation water quality and quantity was recorded weekly (Mercuri and others 2005). Measured weekly irrigation volumes were multiplied by average nutrient load of irrigation water.	An even distribution of water from irrigation drippers across the site is expected. There is no account for fluctuations in irrigation water quality (average weekly nutrient load only is used).
Amendment application	Initial amendment nutrient load for each amendment using Equation 2 (refer to text) multiplied by amount applied.	The percentage of fines (<2 mm) was estimated for pure amendment samples of biosolids (80%), compost (70%), and topsoil (50%) and was measured as 36.3% for overburden. An even distribution of nutrients and solid mass throughout amended treatment areas is expected.
Rainfall	Rainfall inputs were based on estimates from rainfall in Australia from a range of sources (Lockwood 1989; Keith 1997).	As rainfall inputs are generally small in comparison to other nutrient inputs (such as amendment application), it was deemed acceptable to use average values where no specific Upper Hunter Valley data were available.
Run-on/lateral flow	Not measured	Subsurface lateral flow in overburden materials was considered to be small, as there were no impermeable layers within the profile. Nutrient inputs and outputs by subsurface lateral flow were, therefore, assumed to be small and to cancel out. Similarly, any surface run on was assumed to be negligible and canceled out by any run-off.
Litterfall	Not measured	Inputs of nutrients from litterfall were not included because the volume of litter input was minor compared to the substrate nutrient pool and accumulation was not observed at this stage. Weeds were chemically controlled periodically throughout the experiment and left on the surface to decompose. Over time, visual observations indicated very little remaining biomass on the surface, suggesting decomposition and incorporation. It is assumed that uptake and subsequent decomposition would result in no net effect of weeds on the nutrient budget, therefore, weeds were excluded from the biomass uptake.
Residual input (includes weathering, mineralization, N fixation)	Predicted from rearrangement of Equation 1 (refer to text)	A positive residual value indicates net gain through nutrient inputs.

Table 1. Continued

Outputs		
Aboveground plant uptake	Standing crop aboveground biomass was multiplied by average nutrient content of tree tissues ($\sum$foliar, stem, and branches)	It was assumed that all plant nutrient uptake was sourced from the 0–30 cm substrate nutrient pools (and 0–40 cm profile in topsoil treatment) because, even though deep taproots might have developed and roots concentrated within overburden voids, the bulk of fibrous roots that take up nutrients are likely to be concentrated at the surface (Ryan 1995; Katterer and others 1995).
Belowground plant uptake	The aboveground biomass uptake was multiplied by 0.2 to estimate belowground biomass.	Belowground biomass was estimated based on published ratios of aboveground to belowground biomass and nutrient uptake from the literature in young eucalypt roots, which have an estimated ratio of 5:1 (Misra and others 1998a, 1998b; Xu and others 2002). Although there are potential errors in assuming a constant ratio between aboveground and belowground biomass (Newbould 1967), in the absence of actual data this was deemed a suitable estimate.
Runoff/lateral flow	Not measured	Subsurface lateral flow in overburden materials was considered to be small, as there were no impermeable layers within the profile. Nutrient inputs and outputs by subsurface lateral flow were, therefore, assumed to be small and to cancel out. Similarly, any surface runoff was assumed to be negligible and canceled out by any run-on.
Residual output (includes fixation, leaching, denitrification, and volatilization)	Predicted from rearrangement of Equation 1 (refer to text)	A negative residual value indicates net loss of nutrients.
Change in substrate nutrient storage (ΔSS)		
ΔSS	Difference between the final and initial nutrient pool for each treatment. Nutrient pools calculated using Equation 2.	The percentage of fines (<2 mm) was estimated for pure amendment samples of biosolids (80%), compost (70%), and topsoil (50%) and measured as 36.3% for overburden. An even distribution of nutrients and solid mass throughout amended treatment areas is expected. By the end of year 2, biosolids and compost had completely disintegrated and mixed in with the underlying 0–30 cm overburden profile. However, the topsoil amendment was applied at 10 cm and was still present at year 2, so this was regarded as a separate compartment. Calculations are explained in the text.

and 4. The output $\text{residual}_{0\text{–}10\text{ cm}}$ of Equation 3 is an input to Equation 4, thereby canceling each other and giving:

$$\begin{aligned}\Delta SS_{0-40\text{ cm}}\ (\text{kg/ha})(\Delta SS_{0-10\text{ cm}} + \Delta SS_{10-40\text{ cm}})\\ = \text{inputs (rainfall} + \text{irrigation}\\ + \text{input residual}_{0.5-10\text{ cm}} + \text{input residual}_{10-40\text{ cm}})\\ - \text{outputs (plant uptake}_{(ab)}\\ + \text{output residual}_{10-40\text{ cm}}) \qquad (5)\end{aligned}$$

As the change in substrate storage of nutrients (ΔSS) over the experiment was known, the unmeasured (residual) components of the balance can be estimated by rearranging Equation 1, thereby making the unknown components the focus. The exact nature of these unmeasured components differs for each element but represents a balance between a positive input and a negative output. Conclusions can be drawn about the dominating processes involved in generating these residual values. The residual value was estimated for the overburden, fertilizer, biosolids, and compost treatments whereby:

$$\begin{aligned}\text{Residual}_{0-30\text{ cm}}(\text{kg/ha})\ (\text{input residual}\\ - \text{output residual})\\ = \Delta SS_{0-30\text{ cm}} + \text{plant uptake} \qquad (6)\\ - \text{inputs (rainfall} + \text{irrigation}\\ + \text{nutrient addition from amendments})\end{aligned}$$

For the topsoil treatment (0–40 cm), the equation incorporates both the 0–10 cm topsoil amendment and the underlying 10–40 cm substrate, whereby:

$$\begin{aligned}\text{Residual}_{0-40\text{ cm}}(\text{kg/ha})(\text{input residual}_{0-10\text{ cm}}\\ + \text{input residual}_{10-40\text{ cm}}\\ - \text{output residual}_{10-40\text{ cm}}) \qquad (7)\\ = \Delta SS_{0-40\text{ cm}} + \text{plant uptake}\\ - \text{inputs (rainfall} + \text{irrigation})\end{aligned}$$

The residual value is the combination of unmeasured inputs and outputs. Therefore, a positive residual value indicates net gain to the substrate subsystem through nutrient inputs, whereas a negative residual value indicates a net loss. Nutrient balance information as inputs (rainfall, irrigation, and amendment inputs), outputs (accumulation in aboveground and belowground biomass), measured change in soil storage, and estimated residual values (net residual inputs–outputs) are presented for exchangeable Ca, exchangeable Mg, exchangeable K, exchangeable Na, total N, total P, and extractable P.

Statistical Analyses

Analysis of variance (ANOVA) was used to test the difference between treatments using S-PLUS (Insightful Corporation, 2001). Dependent variables were the change in substrate storage (ΔSS) for each nutrient variable. A log transformation was performed on data when the assumptions of normality were not met. Where significant differences were detected, the contrast functions in S-PLUS were employed to determine the significance between levels within treatments. The change in soil nutrient storage was negative for some treatments. To overcome this in the ANOVA, values were converted to tons and the minimum values (highest decrease) were subtracted from each value. To assess if the soil nutrient contents at baseline were significantly different to contents at year 2, paired *t*-tests in S-PLUS (Insightful Corporation, 2001) were used for individual subtreatments (Irrigation × Amendment).

Standard errors of the means are presented for the predicted residual values. As mentioned earlier, the mass balances were determined for each individual treatment, but are presented for the interaction of irrigation and amendment only. Therefore, mean values incorporate the subplot effect of species and replicates, thereby giving eight observations per treatment. The random variability of each model component was calculated separately, as they were independent.

Results

Inputs and Outputs

Cation inputs (particularly Na) to irrigated treatments were much higher than rain-fed treatments (Table 2). Inputs of cations from irrigation water at this magnitude (e.g., an additional 327 times the Na received from rainfall alone) highlight the necessity for determining the impact of applied salts on the substrate. In comparison, the inputs of N and P from irrigation were low.

Nutrients supplied from amendments were substantial, with the highest inputs of Ca, K, and N from compost, whereas biosolids contributed larger loads of Mg, Na, and P (Table 3). In comparison, the fertilizer treatment supplied very little N and P relative to compost and biosolids.

The accumulation of cations, N, and P in the aboveground and belowground biomass was calculated for the standing crop and was confounded by trends in survival and biomass for treatments and species (Table 4). Due to a poor growth response in the top-

Table 2. Total amount of each element applied (kg/ha) to the two main treatment effects (rain-fed and irrigated) after 2 years of the experiment

Element	Rain-fed (rainfall only)	Irrigated (irrigation + rainfall)
Ca	8.0	2488
Mg	4.0	2974
K	6.0	424
Na	20.0	6560
N	8.0	15.7
P	0.32	1.5

soil treatment, there was significantly less nutrient accumulation in trees growing in the topsoil treatment, and *E. occidentalis* showed a significantly higher accumulation of nutrients due to its larger biomass (Mercuri and others 2005).

Measured Change in Substrate Storage of Nutrients

The storage of Ca in rain-fed compost treatment significantly increased by 2050 kg/ha at year 2 (Table 5). The fertilizer treatment had increased Mg, and the biosolids treatment showed increased K in both the rain-fed and irrigated treatments from baseline conditions. The storage of Na significantly decreased in rain-fed biosolids, compost, and topsoil treatments by year 2 (Table 5). The substrate storage of N and total P increased significantly by year 2 in the rain-fed and irrigated biosolids, compost, and topsoil amendment treatments. Extractable P was significantly higher from baseline conditions in rain-fed and irrigated biosolids and compost.

In comparing the difference among amendments, compost had a significantly higher ΔSS value for Ca (1170 kg/ha, $P < 0.01$) and N (2840 kg/ha, $P < 0.05$) than the other treatments. All treatments, except irrigated biosolids and irrigated topsoil, showed negative ΔSS for Na. The biosolids treatment had a significantly higher ΔSS value for the total-P pool than the other treatments (910 kg/ha, $P < 0.01$). Both biosolids and compost had a higher ΔSS value for extractable P (51.6 and 68.9 kg/ha, $P < 0.01$) compared to the very small variation over time in the other three amendment treatments.

Estimated Residual for Overburden, Fertilizer, Biosolids, and Compost Amendments

Residual values for all rain-fed amendments were positive for Ca, Mg, and K, whereas they were generally negative for irrigated amendments, except K in biosolids (Table 6). The highest positive residual value was in the rain-fed compost treatment for Ca (ranging between 1290 and 1960 kg/ha) (Table 6). All irrigated treatments had negative residual values for Ca and Mg that were orders of magnitude higher than those estimated in rain-fed treatments. Of the irrigated treatments, overburden, fertilizer, and compost had negative K residual values, with irrigated compost having the highest negative value (ranging between –575 and –251 kg/ha) (Table 6). Na residual values for all rain-fed and irrigated treatments were negative; however, irrigated treatments had negative residual values that were orders of magnitude higher than rain-fed treatments (Table 6). With a total of 6560 kg/ha Na added in irrigation over the 2-year experiment (Table 2), the Na residual values in the irrigated treatments (ranging between −6310 and −6750 kg/ha) suggested that between 96% and 103% of Na was lost by leaching (assuming the residual value accounts predominantly for losses). Therefore, very little of the large amount of Na applied in the saline mine water remained in the 0–30 cm profile.

All treatments had positive residual values for N (suggesting dominance of N inputs). The estimated N residual value was highest for the irrigated compost treatment (3340 kg/ha) (Table 6). The estimated N residual values for the fertilizer treatments were highly variable and fluctuated around zero, suggesting a balance between unaccounted inputs and outputs.

The estimated residual for total P was positive for all treatments except rain-fed biosolids, but the high degree of variation for both biosolids treatments suggested no net input or outputs (values fluctuate around zero) (Table 6). Positive residual values were estimated for all other treatments, with the irrigated compost treatment having the highest residual value for total P (ranging between 447 to 762 kg/ha). The residual value for extractable P indicated negative values for rain-fed and irrigated biosolids and rain-fed compost (Table 6). The residual values estimated for biosolids treatments showed a dominance of losses of extractable P from the substrate subsystem. The irrigated compost treatment residual value ranged across zero (between −12.6 and 86.4 kg/ha). The overburden and fertilizer treatments showed slightly positive residual values, suggesting extractable-P net accumulations.

Estimated Residual for the Topsoil Treatment

The topsoil treatment mass balances refer to changes over time for the top 40 cm, incorporating the topsoil (0–10 cm) and the underlying substrate (10–40 cm) (Equation 7). Under rain-fed conditions, estimated residual values for the exchangeable cations were positive for Ca and Mg and negative for K and Na

Table 3. Amount of each element applied (kg/ha) as a result of substrate amendment application

Element	Overburden	Fertilizer	Biosolids	Compost
Ca	—	—	64.7	705
Mg	—	—	68.8	15.6
K	—	—	6.3	521
Na	—	—	68.9	18.4
N (total)	—	23.2	770	1141
P (total)	—	27	1047	228

Table 4. Plant elemental (Ca, Mg, K, Na, N, and P) storage in the aboveground and belowground biomass (kg/ha) for the main treatment, effects of irrigation, amendment, and species at the end of year 2 (June 2002)

Treatments	Ca	Mg	K	Na	N	P
Irrigation						
Rain-fed	178a	141a	190a	86a	260a	51a
Irrigated	351a	222a	308a	125a	451a	94a
Amendment						
Overburden	239a	178a	232a	115a	405a	40a
Fertilizer	224a	170a	208a	75a	278a	73a
Biosolids	321a	216a	305a	120a	413a	92a
Compost	398a	245a	358a	133a	525a	124a
Topsoil	141b	99b	142b	83b	155b	33b
Species						
C. maculata	165b	121b	156b	58b	228b	54b
E. botryoides	218b	185b	148b	57b	263b	60b
E. tereticornis	238b	146b	161b	52b	333b	43b
E. occidentalis	438a	274a	531a	255a	597a	133a

Note: Means with the same character within each year are not significantly different.

(Table 7). Irrigated topsoil treatments showed a net loss, with negative values for all exchangeable cations by orders of magnitude compared to rain-fed treatments. Saline mine-water irrigation added a total of 6560 kg/ha of Na (Table 2), and the average Na residual value for irrigated topsoil was −6340 kg/ha. This suggested that 96.6% of applied Na was lost to leaching (assuming the residual value accounts for losses only), similar to the other treatments. The N residual values for topsoil were similar between rain-fed (average 446 kg/ha) and irrigated (average 498 kg/ha) treatments, but were highly variable (Table 7). The total-P residual value estimated for the rain-fed topsoil treatment fluctuated around zero and was lower than that for the irrigated treatment. The extractable-P residual values had a higher variance and fluctuated around zero, indicating no net input or outputs.

Discussion

The use of saline mine-water irrigation and substrate amendments were evaluated, not only for their potential to increase tree growth (Mercuri and others 2005) but also for their potential environmental impacts. The calculation of nutrient input–output mass balances provided information to determine the degree to which salts applied in irrigation (e.g., exchangeable cations) are accumulating in the substrate, the retention of nutrients applied in the amendments, and potential losses (e.g., leaching of salts and nitrogen to the groundwater).

Nutrient Budgets for Irrigated Treatments

The input of Ca, Mg, K, and especially Na from the saline mine-water irrigation, compared with those from rainfall, was substantial (Table 2). The application of substrate amendments also contributed additional exchangeable cations to the substrate subsystem (Table 3), resulting in a higher input of cations from irrigated and nutrient-amended treatments.

The estimated residual values for Ca, Mg, K, and Na in all irrigated treatments (except irrigated biosolids for K) were negative, demonstrating a net loss from the substrate subsystem. These losses were dominated by cations applied in irrigation water, although some of the residual value must be attributed to other inputs, such as weathering and cation exchange with subsequent leaching. This finding highlights the high

Table 5. Change in substrate storage of each element (kg/ha) for the 0–30-cm profile[a] from baseline (June 2000) to year 2 (June 2002) for each amendment

	Rainfed						Irrigated				
	Baseline (June 2000)		Year 2 (June 2002)				Baseline (June 2000)		Year 2 (June 2002)		
Treatment	Mean	SE	Mean	SE	Sig	Treatment	Mean	SE	Mean	SE	Sig
Ca											
O	3000	323	3150	258		O	2740	230	2200	292	
F	2940	316	3280	190		F	2790	211	3070	163	
B	3020	281	3400	465		B	2970	256	3300	297	
C	3090	473	5140	428	*	C	2198	185	2500	298	
T	3180	225	3690	232		T	3490	198	4200	300	
Mg											
O	3360	377	3550	375		O	2680	356	2320	269	
F	2640	171	3140	117	*	F	2470	107	2930	119	*
B	3090	285	3190	189		B	2370	120	2740	142	
C	2734	130	3360	207	*	C	2110	275	1460	134	
T	3220	268	3360	187		T	2710	203	3280	199	
K											
O	344	85.9	363	41.0		O	387	27.4	431	83.2	
F	245	16.6	502	36.6	*	F	508	92.6	428	21.2	
B	306	−49.8	542	34.2	*	B	429	36.7	629	35.9	*
C	364	51.2	1050	137	*	C	371	13.6	515	110	
T	341	36.7	354	22.1		T	455	41.7	557	28.1	
Na											
O	902	144	656	125		O	961	281	650	171	
F	718	97.2	528	82.7		F	513	98.7	375	144	
B	660	83.7	305	61.0	*	B	584	149	715	129	
C	746	34.9	464	52.5	*	C	932	220	613	95.3	
T	941	59.3	667	67.0	*	T	464	87.6	603	121	
N											
O	2480	293	2260	265		O	1814	359	2510	515	
F	1690	538	1970	110		F	2310	283	2200	158	
B	1480	127	3110	196	*	B	1640	181	2820	255	*
C	2221	215	4460	341	*	C	2160	234	6450	889	*
T	1226	139	2060	235	*	T	1330	145	2240	252	*
Total P											
O	400	60.7	554	59		O	659	158	920	201	
F	440	61.2	534	35		F	429	53.0	522	57.5	
B	360	57.2	1160	156	*	B	437	30.6	1530	216	*
C	382	21.4	852	55	*	C	498	43.9	1180	153	*
T	357	29.4	567	30	*	T	379	54.1	801	116	*
Extractable P											
O	2.0	0.31	2.7	0.50		O	5.2	1.3	3.9	0.84	
F	2.4	0.36	2.6	0.37		F	2.4	0.26	2.8	0.17	
B	2.4	0.43	40.9	10.8	*	B	3.5	0.49	68.0	16.6	*
C	2.4	0.36	30.5	4.2	*	C	3.7	0.39	113	27.2	*
T	1.7	0.20	2.0	0.18		T	5.6	1.0	3.8	0.35	

Note: (O = overburden, F = fertilizer, B = biosolids, C = compost, and T = topsoil) under rain-fed (RF) and irrigated (I) conditions. sig = significance whereby the asterisk denotes a significant difference ($P < 0.05$) between baseline and year 2.

[a]For the topsoil amendment, substrate storage at year 2 is for the 10–40 cm profile.

proportion of salts in irrigation water lost by leaching and the lack of salt accumulation in the substrate to 30 cm, thereby not hindering early tree growth in the short term. The fate of leached salt is a concern regarding pathways and likely contribution to groundwater and will require research attention in the future.

Table 6. Measured inputs (rainfall, irrigation, and amendments), outputs (plant uptake), Δ substrate storage (time 2 – time 0), and residual values (kg/ha) for Ca, Mg, K, Na, N, total P, and extractable P over the 2-year experiment for each amendment (except topsoil) under rain-fed and irrigated conditions.

		Outputs		Δ SS		Residual				Outputs		Δ SS		Residual	
Treatments	Inputs	Mean	SE	Mean	SE	Mean	SE	Treatments	Inputs	Mean	SE	Mean	SE	Mean	SE
Ca															
Rain-fed								Irrigated							
O	8	211	55.2	146	181	349	182	O	2490	268	69.3	−537	316	−2760	325
F	8	112	26.3	342	215	446	219	F	2490	336	71.1	282	215	−1870	218
B	72.7	183	43.9	379	287	489	341	B	2550	460	126	336	222	−1760	304
C	713	294	103	2040	329	1620	336	C	3190	501	128	306	259	−2390	332
Mg															
Rain-fed								Irrigated							
O	4	185	39.1	186	206	367	210	O	2970	170	39.3	−369	317	−3170	319
F	4	95.6	17.9	494	113	586	114	F	2970	245	53	457	69.3	−2270	87.2
B	72.8	145	33.1	93	170	166	174	B	3010	286	88.6	364	101	−2360	135
C	19.6	204	63.3	623	161	807	173	C	3040	286	80.3	−650	170	−3410	188
K															
Rain-fed								Irrigated							
O	6	228	60.3	18.2	70.3	241	92.6	O	424	236	77.2	44.1	88	−144	117
F	6	107	29.8	257	39.8	358	49.8	F	424	308	82.2	−80	87.2	−196	120
B	12.3	192	56.8	236	58	416	81.2	B	430	417	155	200	43.1	187	161
C	527	328	122	685	161	486	202	C	945	388	121	144	107	−413	162
Na															
Rain-fed								Irrigated							
C	20	111	38.7	−247	93.7	−156	101	O	6260	119	43.1	−311	151	−6750	157
F	20	40.7	12.6	−190	47.2	−169	48.8	F	6560	110	30.9	−138	104	−6590	109
B	88.9	86.6	33.8	−355	36.4	−357	49.6	B	6590	153	45.6	131	100	−6310	110
C	38.4	107	44.3	−283	60.9	−214	75.3	C	6570	159	51.3	−320	251	−6730	256
N															
Rain-fed								Irrigated							
O	8	377	114	−223	111	146	159	O	15.7	432	133	700	290	1195	776
F	31.3	151	29.1	280	472	342	547	F	39	404	118	−112	410	167	300
B	778	270	58.2	1560	308	1040	131	B	786	557	184	1110	266	971	362
C	1150	376	117	1850	428	1310	298	C	1160	672	240	3822	730	3340	982
$P_{(T)}$															
Rain-fed								Irrigated							
O	0.3	40.9	12.4	154	20.1	195	23.6	O	1.5	39.2	10.8	260	72.6	298	73.4
F	27.3	44.4	12.8	94.3	45.4	111	47.1	F	28.5	101	20.9	92.1	24.6	165	32.3
B	1050	58.7	15.3	800	183	−188	183	B	1050	124	30.2	1090	180	170	182
C	228	92.2	31.2	470	54.7	334	63	C	230	157	42.4	678	152	605	158
$P_{(Ex)}$															
Rain-fed								Irrigated							
O	0.3	40.9	12.4	0.7	0.4	41.3	12.4	O	1.5	39.2	10.8	−1.4	1.5	36.3	10.9
F	27.3	44.4	12.8	0.1	0.4	17.2	12.8	F	28.5	101	20.9	0.4	0.2	72.9	20.9
B	1050	58.7	15.3	38.6	10.3	−950	18.5	B	1050	124	30.2	64.5	15.7	−860	34
C	228	92.2	31.2	28.1	4	−108	31.4	C	230	157	42.4	110	25.4	36.9	49.5

Note: (O = overburden, F = fertilizer, B = biosolids, C = compost, and T = topsoil) under rain-fed (RF) and irrigated (I) conditions. sig = significance whereby the asterisk denotes a significant difference ($P < 0.05$) between baseline and year 2.

The Drayton experimental site was relatively confined with little runoff (Simpson, personal communication, 2001). The mass balance data suggest significant "leakiness" under irrigation, with a large proportion of the applied cations being lost below 30 cm of the overburden substrate by leaching. On the basis of site hydrological data collected over the 2-year experiment (Joyce 2003), much of the applied water

Table 7. Ca, Mg, K, Na, N, and P (P_T- total and P_{Ex} -extractable), inputs, outputs, Δ substrate storage (time 2 − time 0), and residual values (kg/ha) over the 2-year experiment for the topsoil amendment (0–40 cm) under rain-fed and irrigated conditions.

		Outputs (U)		Δ Substrate Storage		Residual	
Treatment	Inputs (RF + I)	Mean	SE	Mean	SE	Mean	SE
Ca Rain-fed	8.0	91.0	33.2	884	173	927	176
Ca Irrigated	2520	190	73.6	1110	310	−1220	319
Mg Rain-fed	4.0	74.6	27.7	317	140	372	143
Mg Irrigated	2990	123	45.2	769	235	−2100	240
K Rain-fed	6.0	94.5	43.8	−246	56.1	−159	71
K Irrigated	426	190	93.7	−130	49.9	−367	106
Na Rain-fed	20.0	43.2	20.1	−281	94.3	−258	96
Na Irrigated	6560	82.7	46.4	138	70.9	−6340	85
N Rain-fed	8.0	122	42.4	333	254	446	257
N Irrigated	15.7	189	70.7	324	305	498	314
P_T Rain-fed	0.32	18.7	6.33	−18.7	34.2	−0.35	34.8
P_T Irrigated	1.52	48.2	21.8	157	98.3	203	101
P_{Ex} Rain-fed	0.32	18.7	6.33	0.27	0.34	18.6	35.3
P_{Ex} Irrigated	1.52	48.2	21.8	−1.86	0.91	44.8	103

drains deeper into the overburden profile, possibly returning to groundwater. Of future environmental concern is the volume of water and associated dissolved salts reaching groundwater and their impact on regional groundwater quantity and quality.

Residual values for the divalent cations were positive for rain-fed conditions, demonstrating that accumulation occurred (Tables 6 and 7). The most likely principal source of the residual input was from weathering of overburden and release of cations to exchange sites. Although the release of nutrient elements in a mature soil by the weathering of primary silicate minerals is generally slow (Keith 1997), excavated overburden materials can weather rapidly, as they are exposed to oxygen, wetting and drying cycles, and plant roots. Weathering rates in overburden have been reported as being extremely high (Wilden and others 2001; Dang and others 2002) compared to agricultural soils (Sverdrup and Rosen 1998). Large fluxes in total Ca and Mg have been reported for lignite mine soils up to 1 ton/ha/year for Ca, and this is believed to be related to the weathering of minerals such as calcite (Wilden and others 2001; Gast and others 2001). Calcite was observed in the overburden materials and, therefore, would have potentially contributed to the high Ca concentrations recorded. The degree to which irrigation might have influenced this process is unknown, but the residual values for Ca and Mg in irrigated overburden suggest significant losses of these cations, in addition to those supplied through irrigation (Tables 6 and 7). Further investigations of the weathering patterns of irrigated overburden material are required.

Net Na losses were recorded in all rain-fed treatments but were 20–30-fold less than under irrigation, demonstrating that Na was leached from the profile, even under rain-fed conditions. With the input of Ca and Mg ions from weathering, displacement of Na from exchange sites will occur, followed by rapid leaching (Brady and Weil 2002).

Nutrient Budgets for Substrate Amendments

Some N and P accumulation was noted in unamended conditions (Table 5). This was not significant; it was a reflection of the variability of overburden. High spatial variation in overburden chemical properties has been widely reported (Kimber and others 1978; McFee and others 1981; Johnson and others 1982; Hawkins 1998). In addition, coal from the Upper Hunter Valley Greta Coal Measures has an N content of 1.5% (Power Survey Sectional Committee 1955), so the presence of coal fragments in analytical samples might influence variability in the final result. However, the increase in substrate storage of nutrients (mainly N and P) reported for this experiment was high under amended conditions. The average increase in total N from baseline conditions for compost was 3265 kg/ha, and for biosolids, it was 1405 kg/ha, demonstrating these amendments were effective in sustaining a substantial increase in substrate N content after 2 years.

Nitrogen residual values might include contributions by N fixation and losses by volatilization and denitrification. Estimates for denitrification and volatilization from an effluent-irrigated pasture have been previously reported as low (0.91–1.33 kg/ha/year and 0.51–0.72 kg/ha/year, respectively), although the rates were influenced by the redox potential (Lund and others 1981). Although most residual N values were positive, suggesting a dominance of N inputs, there might have been periods of anoxia, especially under irrigation, when losses by these processes affected the substrate subsystem's mass balance. The residual values measured in the current experiment for overburden and fertilizer were highly variable and ranged across zero, and nitrogen accumulation on colliery spoil has been previously reported as low (8 kg/ha/year) (Palmer and Chadwick 1985). Therefore, predictions of the likely processes contributing to the positive N residual values can only be discussed for the substrate amendments of biosolids, compost, and topsoil.

Biosolids and compost amendment treatments showed a significant increase in total N, total P, and extractable P after 2 years (Table 5). Topsoil amendment significantly increased total N and P after 2 years. Many authors have noted increased availability of P as a result of biosolids amendment to soils (Brockway and others 1986; Lutrick and others 1986; Robinson and Polglase 1996). Substrate nutrient increases were reported by Albaladejo and others (1994) after 3 years in a compost amendment trial, and elevated N values were sustained 1 year after application to a burned forest soil (Guerrero and others 2001) and 2 years after application to forest soils (Borken and others 2002). Increased nutrient content in these amendments suggest that the nutrients (N and P) supplied by the amendments were sustained 2 years after application. However, the lack of difference in nutrient uptake (Table 4) suggests that nutrients might have been available in excess of tree requirements.

The mean N residual values for amendments of biosolids, compost, and topsoil treatments under both rain-fed and irrigated conditions were positive, indicating an accumulation of total N. Accumulations of this magnitude (e.g., 477–2738 kg/ha) are difficult to explain ecologically. Reported values for N fixation in the literature include 14 kg/ha/year in a forested ecosystem (Likens and Bormann 1995), up to 300 kg/ha for legume crops (Herridge and Bergersen 1988) and up to 103 kg/ha for N fixation not associated with legumes (Gibson and others 1988). Although residual values in the current experiment are representative of 2-year data, it is unlikely that N fixation could solely explain the residual N values detected for biosolids, compost, and topsoil. It is possible that these residual values result from unaccounted inputs from the amendments. Nutrients inputs to the mass balance were calculated based on average spreading depth for each amendment (Equation 2). Amendments were applied using heavy machinery and it is unlikely that an even and uniform spread was achieved, especially for biosolids and compost where the application depths were 0.98 cm and 3.0 cm, respectively. As residual values were highly positive for N and P, it is likely that inputs were underestimated.

The mass balance calculations for N are for total N, as available forms were not measured. However, the high input of N from amendment applications (Table 3) does not reflect plant availability. The C:N ratio for compost was 24:1 (Joyce 2003), indicating potential mineralization and immobilization by soil microbes, thereby limiting plant availability and uptake.

Total-P residual values were positive for all treatments except rain-fed biosolids (Table 6). Mass inputs or outputs of P are difficult to explain ecologically, although the large accumulation might be attributed to differences in the spreading efficiency of amendments. The estimated extractable-P residual values were positive but low (in relation to total P) for overburden, fertilizer, and irrigated compost (Table 6) and topsoil (Table 7). Extractable-P inputs could result from dissolution of phosphate compounds or desorption. Reported mass balances for forest ecosystems (Likens and Bormann 1995) and on afforested mine sites (Wilden and others 2001) showed very little change in extractable P over time. Yet, P availability is dynamic and is influenced by plant uptake, desorption, and fixation properties of soils, and the extractable-P pool in the substrate can be highly volatile. It is unlikely that the residual values for extractable P are an indication of net extractable-P accumulation, but, rather, a result of sample variability.

Extractable-P residual values demonstrated overall losses from biosolids and rain-fed compost treatments (Table 6). A negative residual value for extractable P might be the result of inorganic P becoming fixed into unavailable compounds (precipitated out of solution or sorbed to mineral surfaces) or leaching loss. Leaching of phosphate is unlikely (or would be quite low), as, for comparison, phosphorus leaching losses have been estimated to be 0.33 kg/ha/year in northern forests (Johnson 1992). The fixation of extractable P into unavailable forms, resulting from alkaline conditions and moderately high Ca levels, is a more likely process, resulting in negative residual values. Calcium and Fe-bound P compounds have also been previously shown to increase with biosolids addition (Folle and others 1995). Another mechanism that might contribute to a net loss in extractable P is through rapid plant uptake. Although uptake is accounted for in mass balance calculations, there is a possibility that the P uptake by the belowground biomass was underestimated. It has been suggested that the P requirements for the effective function of eucalypt roots is more important than for foliage (Kirschbaum and others 1992; Misra and others 1998b).

Residual values for Ca, Mg, and K were positive under rain-fed conditions (Table 6), indicating accumulation from weathering and release of exchangeable forms of these cations. The overburden contained calcite and dolomite, which can release available forms of Ca and Mg upon dissolution. Increased substrate K might have come from the weathering of K-containing aluminosilicate minerals (Cravotta 1998). The clay mineral illite was abundant in shales and mudstones, and the crystal structure contains interlayer K ions, which are relatively easily released by weathering.

Model Assumptions

Potentially, some of the values in the input–output model might be overetimated or underestimated. Nutrient inputs from rainfall were small compared to those added from irrigation or stored in the substrate and, therefore, an overestimate or underestimate of inputs would be unlikely to affect the overall nutrient budget. There is potential for the atmospheric input of N to be influenced by the nearby Bayswater and Liddell power stations. Research has shown that acidic depositions in areas surrounding power stations in the Hunter Valley are small (EPA 1997) compared to world standards (Alastuey and others 1994). The predominant northwesterly wind direction (Muswellbrook Shire Council 1998) in relation to the location of the experimental site means that this was unlikely to have influenced the estimated rainfall inputs.

The reported biomass of roots of young eucalypts varied in the literature, but due to a lack of data, an estimate of 20% of the aboveground biomass was used across all treatments (Table 1). Increased root nutrient uptake has been previously noted in response to fertilizer addition (Misra and others 1998b). A study by Xu and others (2002) reported decreases in the proportion of root biomass with increasing P fertilizer. The use of a single estimate across the substrate amendments ignores the possibility of amendment and species differences in root biomass and nutrient concentration. As the concentration of nutrients changed between treatments in the aboveground biomass (Table 4), it could be assumed that a similar change would be found in the root biomass. Therefore, in the absence of actual measurements, the estimated contribution of nutrient storage in the root biomass to the input–output budgets was considered acceptable. In addition, the assumption that tree nutrient uptake was sourced entirely from the 0–30-cm profile might have been an overestimate (i.e., deeper root penetration resulting in a reduced uptake value). Similarly, a sensitivity analysis suggested this was unlikely to affect the overall balance (Joyce 2003).

The nutrient content of the weedy understory component was not measured in the current experiment. However, weed management throughout the experiment was such that the dead biomass was left *in situ* to decompose, thereby releasing nutrients back into the substrate. Grigg and Mulligan (1999) noted high N and P contributions from the understory grasses to a site nutrient budget on rehabilitated coal mine overburden in central Queensland. The contribution of forest floor litter to the ecosystem nutrient pools has been shown by other authors to be relatively small, compared to soil and tree nutrient storage (Feller 1980; Frederick and others 1985a,1985b,1985c; Smethurst and Nambiar 1995). Further investigation into the nutrient content of the weedy understory might provide insight into the contribution of these plants to the mass balance of nutrients in the substrate. However, if, as assumed, weeds take up and then release nutrients on decomposition, they might provide an efficient mechanism for reducing nutrient losses by leaching.

Short-term fluctuations in nutrient content were not detected by the techniques used in this study. Although some treatments appear to be relatively stable (inputs equal outputs), there might have been times when the balance was tipped toward accumulation (e.g., periods of high evaporative demand after irrigation) or losses (e.g., after a heavy rainfall event). Therefore, the prediction of mass balances over time might underestimate or overestimate variables such as substrate cation concentration (Likens and Bormann 1995), and balances might not reflect variations in cations during the irrigation season (Tedschi and others 2001). However, the results reported here are indicative of longer-term trends in accumulation or losses from the substrate subsystem.

Conclusions

Estimates of nutrient cycling within the experimental treatments provided information on the potential hazards associated with saline mine-water irrigation and substrate amendment. The substantial negative residual values for exchangeable cations demonstrated that the bulk of the cations supplied in irrigation water were lost from the overburden substrate (0–30 cm) through leaching and did not accumulate in the upper part of the profile. Although the fate of these salts beyond this depth is not known, it can be concluded that irrigation with saline mine water has little impact on substrate cation accumulation during the early stage of plantation establishment examined. Accumulations of N and P were detected for the substrate amendments, suggesting that these organic input products (particularly compost) have a good retention of applied nutrients, with very little associated loss of nutrients through leaching.

Acknowledgments

Muswellbrook Shire Council funded the research program and postgraduate scholarship for this project in collaboration with Ecosystem Management in the School of Environmental Sciences and Natural Resources Management at the University of New England. Drayton Colliery generously provided the land, facilities, and infrastructure support for the study. Pam Simpson from Drayton Colliery provided valuable assistance and support during the project. We wish to thank Dr. Ian Davies from the School of Mathematics, Statistics and Computing Sciences, UNE for statistical advice. Marion Costigan and Judi Kenny provided valuable technical assistance in the laboratory, and Trevor Stace assisted in many fieldwork components.

Literature Cited

Alastuey, A., X. Querol, A. Chaves, C. R. Ruiz, A. Carratala, and A. Lopez-Soler. 1994. Bulk deposition in a rural area located around a large coal-fired power station, northeast Spain. *Environmental Pollution* 106:359–367.

Albaladejo, J., M. Stocking, E. Diaz, and V. Castillo. 1994. Land rehabilitation by urban refuse amendments in a semi-arid environment: Effect on soil chemical properties. *Soil Technology* 7:249–260.

Anderson, D. I., and L. J. Henderson. 1986. Sealed chamber digestion for plant nutrient analysis. *Agronomy Journal* 78:937–938.

N. Andrews(ed.). 1999. Synoptic plan: Integrated landscapes for coal mine rehabilitation in the Hunter Valley of New South Wales. Department of Mineral Resources, Sydney.

Bindraban, P. S., J. J. Stoorvogel, D. M. Jansen, J. Vlaming, and J. J. R. Groot. 2000. Land quality indicators for sustainable land management: proposed method for yield gap and soil nutrient balance. *Agriculture, Ecosystems and Environment* 81:103–112.

Borken, W., A. Muhs, and F. Beese. 2002. Changes in microbial and soil properties following compost treatment of degraded temperate forest soils. *Soil Biology and Biochemistry* 34:403–412.

Bormann, F. H., and G. E. Likens. 1967. Nutrient cycling: Small watersheds can provide invaluable information about terrestrial ecosystems. *Science* 155:424–429.

Brady, N. C., and R. R. Weil. 2002. The nature and properties of soils. 13th ed. Prentice-Hall, Englewood Cliffs, NJ.

Brockway, D. G., D. H. Urie, P. V. Nguyen, and J. B. Hart. 1986. Wastewater and sludge nutrient utilization in forest ecosystems. Pages 221–224 *in* D. W. Cole, C. L. Henry, and W. L. Nutter (eds.), The forest alternative for treatment and utilization of municipal and industrial wastes. University of Washington Press, Seattle.

Burns, M. W. 1987. Reafforestation of open-cut coal mines using direct seeding techniques. New South Wales Coal Association, Newcastle.

Cravotta, C. A. 1998. Effect of sewage sludge land application to northern hardwood forests. *Ecological Applications* 5:53–62.

Dang, Z., C. Liu, and M. J. Haigh. 2002. Mobility of heavy metals associated with the natural weathering of coal mine spoils. *Environmental Pollution* 118:419–426.

EPA (Environment Protection Authority). 1997. State of the environment report. Environmental Protection Agency, Sydney.

Feller, M. C. 1980. Biomass and nutrient distribution in two eucalypt forest ecosystems. *Australian Journal of Ecology* 5:309–333.

Folle, F., J. W. Shuford, R. W. Taylor, A. Mehadi, and W. Tadesse. 1995. Effect of sludge treatment, heavy metals, phosphate rate, and pH on soil phosphorus. *Communications in Soil Science and Plant Analysis* 26: 1369–1381.

Frederick, D. J., H. A. I. Madgwick, G. R. Oliver, and M. F. Jurgensen. 1985a. Dry matter and nutrient content of 8-year-old *Eucalyptus saligna* growing at Taheke Forest. *New Zealand Journal of Forestry Science* 15:251–254.

Frederick, D. J., H. A. I. Madgwick, M. F. Jurgensen, and G. R. Oliver. 1985b. Dry matter content and nutrient distribution in an age series of *Eucalyptus regnans* plantations in New Zealand. *New Zealand Journal of Forestry Science* 15:158–179.

Frederick, D. J., H. A. I. Madgwick, G. R. Oliver, and M. F. Jurgensen. 1985c. Dry matter, energy and nutrient content of 8-year-old stands of *Eucalyptus regnans*, *Acacia dealbata* and *Pinus radiata* in New Zealand. *New Zealand Journal of Forestry Science* 15:142–157.

Gast, M., W. Schaaf, J. Scherzer, R. Wilden, B. U. Schneider, and R. F. Huttl. 2001. Element budgets of pine stands on lignite and pyrite containing mine soils. *Journal of Geochemical Exploration* 73:63–74.

Gee, G. W., and J. W Bauder. 1986. Particle-size analysis. Pages 383–411 in A. Klute (ed.). Methods of soil analysis (Part 1): Physical and mineralogical properties, 2nd ed. American Society of Agronomy, Madison, Wisconsin.

Gibson, A. H., M. M. Roper, and D. M. Halsall. 1988. Nitrogen fixation not associated with legumes. Pages 66–88 *in* J. R. Wilson (ed.), Advances in nitrogen cycling in agricultural ecosystems. C.A.B. International, Wallingford, UK.

Grigg, A. H., and D. R. Mulligan. 1999. Biometric relationships for estimating standing biomass, litterfall and litter accumulation of *Acacia salicina* on mined land in central Queensland. *Australian Journal of Botany* 47:807–816.

Guerrero, C., I. Gomez, R. Moral, J. Mataix-Solera, J. Mataix-Beneyto, and T. Hernandez. 2001. Reclamation of a burned forest soil with municipal waste compost: Macronutrient dynamics and improved vegetation cover recovery. *Bioresources Technology* 76:221–227.

Gupta, S. C., and W. E. Larson. 1979. A model for predicting packing density of soils using particle-size distribution. *Soil Science Society of America Journal* 15:1633–1635.

Hannan, J. C., and R. M. Gordon. 1996. Environmental management of coal mines in the Hunter Valley, New South Wales. Pages 265–290 *in* D. R. Mulligan (ed.), Environmental management in the Australian minerals and energy industries. University of New South Wales Press, Sydney.

Hawkins, J. W. 1998. Hydrogeologic characteristics of surface-mine spoil. Pages 3.1–3.11 *in* K. B. C. Brady, M. W. Smith, and J. Schueck (eds.), Coal mine drainage prediction and pollution prevention in Pennsylvannia. Department of Environment Protection, Pittsburgh, PA.

Herridge, D. F., and F. J. Bergersen. 1988. Symbiotic nitrogen fixation. Pages 46–65 *in* J. R. Wilson (eds.), Advances in nitrogen cycling in agricultural ecosystems. C.A.B. International, Wallingford, UK.

Insightful Corporation. 2001. S-PLUS 6 for Windows-Professional. Insightful Corporation, Seattle, WA.

Johnson, D. W. 1992. Base cations: Introduction. Pages 233–235 *in* D. W. Johnson and S. E. Lindberg (eds.), Atmospheric deposition and forest nutrient cycling: A synthesis of the integrated forest study. Springer-Verlag, New York.

Johnson, F. L., D. J. Gibson, and P. G. Risser. 1982. Revegetation of unreclaimed coal strip-mines in Oklahoma. *Journal of Applied Ecology* 19:453–463.

Joyce, A. M. 2003. Plantation early growth responses to saline mine water irrigation and nutrient amendment on coal mine spoil in the Upper Hunter Valley, NSW. PhD thesis, University of New England, Armidale, NSW, Australia.

Katterer, T., A. Fabiao, M. Madeira, C. Ribeiro, and E. Steen. 1995. Fine-root dynamics, soil moisture and soil carbon content in a *Eucalyptus globulus* plantation under different irrigation and fertilisation regimes. *Forest Ecology and Management* 74:1–12.

Keith, H. 1997. Nutrient cycling in eucalypt ecosystems. Pages 195–226 *in* J. E. Williams and J. C. Z. Woinarski (eds.), Eucalypt ecology: Individuals to ecosystems. Cambridge University Press, Cambridge.

Keolsch, R., and G. Lesoing. 1999. Nutrient balance on Nebraska livestock confinement systems. *Journal of Animal Science* 77:63–71.

Kimber, A. J., I. D. Pulford, and H. J. Duncan. 1978. Chemical variation and vegetation distribution on a coal waste tip. *Journal of Applied Ecology* 15:627–633.

Kirschbaum, M. U. F., D. W. Bellingham, and R. N. Cromer. 1992. Growth analysis of the effect of phosphorus nutrition on seedlings of *Eucalyptus grandis. Australian Journal of Plant Physiology* 19:55–66.

Likens, G. E., and F. H. Bormann. 1995. Biogeochemistry of a forested ecosystem. 2nd ed. Springer-Verlag, New York.

Lockwood, P. V. 1989. Element cycling in forest ecosystems with particular reference to weathering rates. PhD thesis, University of New England, Armidale, NSW Australia.

Lund, L. J., A. L. Page, C. G. Nelson, and R. A. Elliot. 1981. Nitrogen balances for an effluent irrigation area. *Journal of Environmental Quality* 10:349–352.

Lutrick, M. C., H. Riekerk, and J. A. Cornell. 1986. Soil and slash pine response to sludge applications in Florida. *Soil Science Society of America Journal* 50:447–451.

McFee, W. W., W. R. Byrnes, and J. G. Stockton. 1981. Characteristics of coal mine overburden important to plant growth. *Journal of Environmental Quality* 10:300–308.

Mercuri, A. M., J. A. Duggin, and C. D. Grant. 2005. The use of saline mine water and municipal wastes to establish plantations on rehabilitated open-cut coal mines, Upper Hunter Valley, NSW Australia. *Forest Ecology and Management* 204:195–207.

Misra, R. K., C. R. A. Turnbull, R. N. Cromer, A. K. Gibbons, and A. V. LaSala. 1998a. Below- and above-ground growth of *Eucalyptus nitens* in a young plantation I. Biomass. *Forest Ecology and Management* 106:282–293.

Misra, R. K., C. R. A. Turnbull, R. N. Cromer, A. K. Gibbons, A. V. LaSala, and L. M. Ballard. 1998b. Below- and above-ground growth of *Eucalyptus nitens* in a young plantation II. Nitrogen and phosphorus. *Forest Ecology and Management* 106:295–306.

Muswellbrook Shire Council. 1998. State of the environment report. Muswellbrook Shire Council, Muswellbrook, Australia.

Myers, B. J. 1992. Effluent loading rates for irrigated plantations: A water balance model. *Australian Forestry* 55:39–47.

Newbould, P. J. 1967. Methods for estimating primary production of forests. 2nd ed. International Biological Programme, London.

Page, A. L., R. H. Miller, and D. R. Keeney (eds.). 1982. Methods of soil analysis, Part 2: Microbiological and biochemical properties, 2nd ed. American Society of Agronomy and Soil Science, Madison, Wisconsin.

Palmer, J. P., and M. J. Chadwick. 1985. Factors affecting the accumulation of nitrogen in colliery spoil. *Journal of Applied Ecology* 22:249–257.

Pare, D., P. Rochan, and S. Brais. 2002. Assessing the geochemical balance of managed boreal forests. *Ecological Indicators* 1:293–311.

Polglase, P. J., and B. J. Myers. 1999. Saline water and amendments for tree plantations on rehabilitated mine sites in the Upper Hunter Valley: Design for trial at Drayton Colliery, A report to the Muswellbrook Shire Council. CSIRO Forestry and Forest Products, Canberra.

Power Survey Sectional Committee. 1955. A report on the coal resources of the Commonwealth of Australia. The Standards Association of Australia, Sydney.

Ranger, J., and M. Turpault. 1999. Input–output nutrient budgets as a diagnostic tool for sustainable forest management. *Forest Ecology and Management* 122:139–154.

Rayment, G. E., and F. R. Higginson. 1992. Australian laboratory handbook of soil and water chemical methods. Inkata Press, Melbourne.

Robinson, M. B., and P. J. Polglase. 1996. Release and leaching of nitrogen from biosolids applied to a pine plantation. Pages 26–31 *in* P. J. Polglase and W. M. Tunningley (eds.), Land application of wastes in Australia and New Zealand: research and practice. Proceedings of the 14th New Zealand land treatment collective. CSIRO Forestry and Forest Products, Canberra.

Ryan, P. J. 1995. Factors affecting the establishment and management of tree stands on rehabilitated coal mines in the Hunter Valley, New South Wales. State Forests of New South Wales Research Division, Sydney.

Smethurst, P. J., and E. K. Nambiar. 1995. Changes in soil carbon and nitrogen during the establishment of a second crop of *Pinus radiata*. *Forest Ecology and Management* 73:145–155.

Sverdrup, H., and K. Rosen. 1998. Long-term base cation mass balances for Swedish forests and the concept of sustainability. *Forest Ecology and Management* 110:221–236.

Tedeschi A., A. Beltran, and R. Aragues. 2001. Irrigation management and hydrosalinity balance in a semi-arid area of the middle Ebro river basin (Spain). *Agricultural Water Management* 49:31–50.

Wilden, R., W. Schaaf, and R. F. Huttl. 2001. Element budgets of two afforested mine sites after application of fertilizer and organic residues. *Ecological Engineering* 17:253–273.

Xu, D., B. Dell, N. Malajczuk, and M. Gong. 2002. Effects of P fertilisation on productivity and nutrient accumulation in a *Eucalyptus grandis* × *E. urophylla* plantation in southern China. *Forest Ecology and Management* 16:89–100.