Plant and Soil **229:** 105–116, 2001.

Litter decomposition and nitrogen mineralization of soils in subtropical plantation forests of southern China, with special attention to comparisons between legumes and non-legumes

Zhi-an Li[1,4], Shao-lin Peng[2], Debbie J. Rae[3] & Guo-yi Zhou[1]
[1]*South China Institute of Botany, Chinese Academy of Sciences, Guangzhou 510650, PR China.* [2]*Guangzhou Branch, Chinese Academy of Sciences, Guangzhou 510070, PR China.* [3]*Centre for Horticulture and Plant Sciences, University of Western Sydney, Richmond, NSW 2753, Australia.* [4]*Corresponding author**

Received 4 April 2000. Accepted in revised form 13 September 2000

Key words: Acacia auriculaeformis, *Acacia mangium*, CO_2 release, *Eucalyptus citriodora*, exotic nitrogen, litter decomposition, nitrogen mineralization, *Pinus elliotii*, *Schima superba*

Abstract

Litter decomposition and nitrogen mineralization were investigated in subtropical plantation forests in southern China. The CO_2 –C release from incubated litter and the forest floor of *Acacia mangium*, *Acacia auriculaeformis*, *Eucalyptus citriodora*, *Pinus elliotii* and *Schima superba* stands were used to estimate relative rates of litter decomposition. Decomposition was not positively correlated with litter nitrogen. *E. citridora* litter decomposed most rapidly and *A. mangium* litter most slowly, both with and without the addition of exotic nitrogen. Aerobic incubation and intact soil core incubation at 30 °C over a period of 30 days were used to assess nitrogen mineralization of six forest soils. Although there were differences in results obtained using the two methods, patterns between legume and non-legume species were the same regardless of method. All soils had pH values below 4.5, but this did not prevent nitrification. The dominant form of mineral nitrogen was nitrate for legume species and ammonium for non-legume species. The nitrogen mineralization potential was highest for soils in which legumes were growing.

Introduction

Litter decomposition and nitrogen mineralization have long been regarded as important links in the nutrient cycles of ecosystems (Horner et al., 1988; Meentemeyer, 1978; Van Vuuren et al., 1993; Vitousek, 1982; Vitousek et al., 1994) and important factors in forest succession, since they largely shape the nutrient status of the forest floor (Li et al., 1996; Yu and Peng, 1995). Nitrogen and C/N ratios often have a direct impact on the decomposition process and nitrogen mineralization (Berg and Staaf, 1980; Berg et al., 1987; Coulson and Butterfield, 1978; Schlesinger and Hasey, 1981; Staaf and Berg, 1982; Taylor et al., 1989; Tian et al., 1992; Witkamp, 1966), indicating that litters with different nitrogen content decompose at different rates.

In the tropics and subtropics of China, a large scale campaign of afforestation was launched in the 1980s. More than 20 forest species were initially planted to evaluate their suitability to the climate and soils of the region. General observations of a number of characteristics including growth rates, timber quality and resistance to insect attack and environmental disturbance led to some of these species being identified as unsuitable. Today the most commonly planted species are the legumes *Acacia auriculaeformis* A.Cunn., *Acacia mangium* Willd., *Acacia confusa* Merr., *Acacia holosericea* A.Cunn.ex g Dun, *Acacia falcata* (L.) Baker ex Merr.) and non-legumes *Pinus elliotii* Engelm, *Pinus massoniana* (Lamb.), *Schima superba*

* FAX No: + 86-20-87701031. TEL No: + 86-20-87639381. E-mail: lizan@scib.ac.cn

Cardn. Et Champ, *Schima wallichii* Choisy, *Eucalyptus citriodora* Hk.f., *Eucalyptus exserta* F. Muell. and *Eucalyptus urophylla*. They are usually planted in single species stands.

Formal studies conducted on specific aspects of these forest species in southern China have compared photosynthesis rates (Zhao et al., 1995) and the structure of arthropod communities (Tan et al., 1995) between species, and investigated nutrient stocks in *A. mangium* (Li et al., 1995). Yu and Peng (1995) found precroppings of legumes resulted in the soil being more easily colonized by other forest species, and attributed this to improved soil nitrogen status. They strongly recommended using legumes as pioneering species in afforestation of degraded land to speed up the succession to ecologically stable forests. It has also been demonstrated by Yi et al. (1984) that litters having different levels of nitrogen will foster different invertebrate and microbe taxa, leading to different ecosystems on the forest floor. However, rates of litter decomposition and nitrogen mineralization have not previously been compared between species used for afforestation in southern China.

The objectives of this study were to: Compare the decomposition rates of a number of legume and non-legume litters with and without exotic nitrogen; and to compare the rates of nitrogen mineralization of soils from a number of legume and non-legume plantation forests in Southern China. CO_2 emissions (which are mainly the result of microbial processes) were used to estimate decomposition rates of forest litters. Aerobic incubation and intact soil core incubation were used to assess nitrogen mineralization.

Materials and methods

This study was conducted at the Heshan Interdisciplinary Experimental Station (112° 54′ E, 22° 41′ N), Chinese Academy of Sciences in Guangdong province, PR. China. The climate of the region is subtropical monsoon with a mean annual precipitation of 1800–2000 mm falling mainly from April to September. The period from October to January is particularly dry. The mean annual temperature is 21.7 °C with the mean maximum monthly temperature of 28.7 °C falling in July and the mean monthly minimum of 13.1 °C falling in January.

The experimental area is typical of the region with low hills (peak elevation of 98 m) and small catchments (each having an area of about 5–8 ha). The soil is an oxisol developed from sandstone, with a pH of about 4.0. In 1984, six adjacent catchments vegetated only with grass were chosen for a scientific study on the basis of their similarity. A different forest type was randomly allocated to each catchment and trees planted on a 2.5 m × 3 m grid (Li et al., 2000; Yu and Peng, 1995). Among them, five catchments are single species stands of *Acacia mangium*, *Acacia auriculaiformis*, *Eucalyptus citriodora*, *Pinus elliotii* and *Schima superba*, and one catchment is a mixed forest of legumes including *A. mangium*, *Acacia confusa* and *Acacia holosericea*. A 2 hectare area which was not afforested has been left unchanged for comparison. These forests have been protected for scientific research since their establishment. Unfortunately this design has inherent limitations because site and forest type effects are confounded. However, in this particular situation, any differences found between species are considered to be almost entirely due to the forest type because of the homogenous nature of the experimental area prior to afforestation.

When this study commenced in 1997, the canopy closure of *A. mangium*, *S. superba* and mixed forest were about 90%, while closure for *A. auriculaiformis* and *P. elliotii* were 50–60% and *E. citriodora* about 30%. Corresponding to the canopy closure, undergrowth was most dense in the *E. citriodora* stand and most sparse in the *A. mangium* stand. In 1997, the litter stocks on the ground were 42.3 T per ha for *A. mangium*, 18.3 T per ha for *A. auriculaiformis*, 1.7 T per ha for *E. citriodora*, 18.7 T per ha for *P. elliotii* and 8.4 T per ha for *S. superba*.

Collection of samples

Litter samples were collected in square metal traps (1 m × 1 m × 0.3 m) with plastic screen mesh (1 × 1 mm^2) bases during the dry season to minimize leaching from the litter. Ten litter traps were randomly placed within a 1 ha area in each of the five single species stands. Litter was collected from the traps every second week from October 1 1997 to January 31 1998, placed into labeled resealable plastic bags, oven dried at 60 °C and then stored at room temperature. At the end of the collection period, litter from each of the five species was bulked, mixed and cut into 2 cm pieces with scissors in preparation for the decomposition test.

For the nitrogen mineralization test, 30 soil cores were randomly sampled from the inner part of each of the six forest stands. As the effect of different forest litter composition is most strongly reflected in the top-

soil (Carlyle and Malcolm, 1986), the top 5 cm of soil was sampled after leaf litter was removed, using 5 cm tall steel cylinders with a total volume of 100 cm^3. Stones and plant residues were removed from samples by hand before being bulked, thoroughly mixed and air dried. All remaining material which passed through a 2 mm sieve was placed into plastic bags and stored at 4 °C for 1 week prior to the commencement of incubation experiments. Meanwhile, soils were resampled using the same method to measure the field water holding capacity (Nanjing Soil Institute, 1980) to provide a base for water adjustment.

For the soil core incubation experiment, five soil cores from each forest were collected as for nitrogen mineralization, except the soil was kept undisturbed in each cylinder by covering each end of the cylinder with plastic lids. The cylinders were stored at 4 °C for 1 week until incubation commenced.

Soil used in the litter decomposition experiment was sampled from the unforested comparison area, using 20 cm tall steel cylinders with an inner diameter of 2.5 cm. Stones and plant residues including tree roots were removed from samples by hand before being thoroughly mixed, passed through a 2 mm sieve and air dried for 5 days prior to the commencement of the experiment.

Litter decomposition

A laboratory incubation method similar to that described by Angers and Recous (1997) was used to determine the rate of CO_2 –C release from different forest litters with or without the addition of nitrogen. Resealable plastic bags fitted over cylindrical wire frames (d=9 cm, h=20 cm) were used as incubation containers. For each of the five different forest litter types, 2 g of litter thoroughly mixed with 200 g of air dried soil were placed into each of six incubation containers. Six control samples were prepared in the same way, without litter. To test the effect of added nitrogen, urea solution at a rate of 100 mg N kg^{-1} soil (20 mg N of urea solution/incubation container) was added to half of the samples (3) in each treatment and the control. All bags were sealed and incubated at 30 °C for 1 day to allow the decomposition system to stabilize. Initially, all bags were opened and a beaker containing 10 ml of 1 m NaOH solution placed inside each to absorb CO_2 emitted from decomposing litter. Bags were resealed and incubated at 30 °C. On each sampling date, the hours of absorbance were recorded, bags were opened and the beaker of NaOH replaced. Soil moisture was checked and adjusted to field holding capacity when necessary. Carbonate in the NaOH samples was precipitated with excess $BaCl_2$, remaining NaOH titrated with HCl to a pH of 8.2 using phenopthalein as the indicator and trapped CO_2 –C calculated from NaOH consumption. The sampling interval was 3 days at the beginning of the experiment, but increased over time. After 45 days the bags were opened and free gas exchange allowed for 2 months, while incubation at 30^oC continued. NaOH was then replaced, bags sealed and NaOH placed in for 4 days and again after 30 days (109 and 139 days after the commencement of the experiment, respectively).

A 200 g subsample of litter from each forest type was retained from litter prepared for the decomposition experiment to determine the initial litter nutrient status (C, N, P, K, Ca, Mg, Fe and Mn). The litter was milled to pass a 1 mm sieve and stored in glass bottles. Dry litter powder was used to determine C and N using a CHN Analyzer (Perkimm II 2400). Approximately 1 g samples digested with concentrated sulfuric acid and 60% $HClO_4$ using the wet-ashing procedure were used to determine P colorimetrically and K by flame emission spectroscopy. Another digestion using a solution of $H_2SO_4 – HNO_3 – HClO_4$ was used to determine Ca, Mg, Fe and Mn by atomic absorption spectroscopy.

The amount of CO_2 –C emitted from the forest ground over 24–48 h was also determined in the field during October 1998. In each forest type, CO_2 –C release was determined at four depths; above litter, the soil surface (litter removed), at 5 cm (upper 5 cm soil removed) and 10 cm depth (upper 10 cm soil removed). Three sites at each depth were prepared in each of the five single species forests, and then left undisturbed for 5 days to allow the system to stabilize. A beaker containing 10 ml of 1 M NaOH was then placed on each site, a plastic bag fitted over a cylindrical wire frame (d=12.5 cm, h=20 cm) placed over the site and the opening fixed firmly to the ground using four metal pegs (l=10 cm). After 24 h, the cylinders were opened and beakers of NaOH taken to the field station laboratory where emitted CO_2 –C was determined as described above. The experiment was then repeated for another 24 h, but only at 5 and 10 cm depths.

Nitrogen mineralization

A laboratory incubation procedure similar to that described by Carlyle and Malcolm (1986) and Frazer et al. (1990) was used to determine the mineral nitrogen content of soil samples. For each of the five single

species forests and the mixed forest, 25 conical flasks with 20 g of bulked soil (adjusted to field water holding capacity) were sealed with rubber stoppers and incubated aerobically at 30 °C. Every day, flasks were removed from the incubator, weighed and opened for 30 min to allow aeration. Distilled water was added when necessary to maintain soil moisture. The mineral nitrogen content of bulked soil was determined from five flasks from each forest type after 0, 5, 10, 20 and 30 days of incubation. Mineral nitrogen content was also determined for five intact soil cores from each forest type after incubation at 30 °C for 30 days. Mineral nitrogen (NH_4^+–N and NO_3^-–N) was extracted by shaking samples for 1 h with 2 M KCl solution at a solid:liquid ratio of 1:5. Extracts were centrifuged and supernatants were analyzed colorimetrically for NH_4^+–N and NO_3^-–N using an Auto Ion analyzer (Lachat Instruments, Milwaukee, WI, USA).

Statistical methods

The percentage difference between CO_2 –C release rates from litter with (+N) or without (-N) the addition of nitrogen were calculated using the following equation:

$$PD = (X_{+N} - X_{-N}) * 100/X_{-N} \quad (1)$$

where X_{+N} is the rate of release with added nitrogen and X_{-N} is the rate without added nitrogen.

Cumulative loss of CO_2 –C from litter over time was described by fitting the following first order kinetics equation to data (Olson, 1963):

$$\ln(x/x_0) = -kt \quad (2)$$

where x_0 is the original carbon mass of litter, x is the amount of carbon remaining after time t, and k is the decomposition constant (d^{-1}).

$$\text{Half decay} = -\ln(1/2)/k \quad (3)$$

The fit of data to the equations were assessed using the coefficient of determination (R^2).

CO_2 –C release rates at 3, 5, 8, 12, 16, 27, 36 and 45 days were linearly regressed against initial litter quality parameters (N, P, K, Ca, Mg, Fe, Mn, C/N, C/P, C/K, N/P, N/K) and correlation coefficients calculated.

To determine the effect of different forest species and soil depth on CO_2 –C emission from the forest floor, data were analyzed using general factorial ANOVAs. Only data collected in the first 24 h was used in one model with the factors being forest species and soil depth. The second model had the same factors (with only the two deepest soil depths) and included a covariate for time. Homogeneity of variance tests and residuals plots indicated the data were homoscedastic and normal. There were no significant interaction effects. The significance of differences among treatment means were examined using deviation contrasts (which compare the mean for each level of a factor with the overall treatment mean), with significance levels adjusted using the Scheffé method (Sokal and Rohlf, 1981).

To determine the effect of the incubation method on mineral nitrogen content of forest soil, data were initially analyzed using 2-way ANOVAs. For each of the dependent variables $NH_4^+$and NO_3^-, three models were used. Factors in the models were species and time (before and after incubation) or species and incubation method. As all models resulted in significant interaction effects, un-planned multiple comparisons using the Scheffé method were used to test differences among species means separately for each soil sampling method or time. $Log_{10}(x + 1)$ transformations were required to stabilize variances for NH_4^+ and NO_3^- data collected after soil incubation. When transformations were required, back-transformed means are presented.

Results

Litter decomposition

The addition of N in the form of urea to *A .mangium* and *A. auriculaeformis* litter generally decreased the rate of CO_2 –C release (Table 1). For the three non-legume species, the addition of N stimulated the emission of CO_2 –C from litter at the beginning of the incubation period, but this trend was reversed after about a month. By the end of the incubation period, percentage difference in CO_2 –C release between +N and -N treatments were generally high and reached a maximum of −42.3% for *P. elliotii*. However, the absolute differences between +N and -N treatments were small because of the low rates of CO_2 –C emission at this late stage of decomposition (Figure 1).

The rate of CO_2 –C release from litter differed most markedly between species in the first 16 days of the decomposition experiment (Figure 1). Without additional nitrogen, the rate of CO_2 –C emission was highest for *E. citriodora* followed by *S. superba* (Figure 1 A). Low but similar rates of CO_2 –C were released from *A. mangium*, *A. auriculaeformis* and *P.*

Table 1. Percentage difference between CO_2 –C release rates from litter with additional nitrogen (N+) and without additional nitrogen (N-). Positive figures indicate acceleration and negative figures decelaration of CO_2 –C release as a result of additional nitrogen

Litter species	Incubation period (days)									
	3	5	8	12	16	27	36	45	109	139
A. mangium	−0.6	5.1	−7.9	3.1	−10.4	−2.9	−8.9	−13.3	−11.9	−6.7
A. auriculaeformis	−0.6	7.7	−2.5	−12.2	−13.1	0.1	−6.6	−21.6	−18.7	−9.7
E. citriodora	12.2	19.8	7.4	−1.7	−7.3	0.0	−15.6	−18.2	−16.6	−12.0
P. elliotii	8.2	12.7	−0.6	5.6	21.7	3.6	−10.4	−14.7	−26.0	−42.3
S. superba	6.0	9.4	0.3	2.6	5.5	−0.3	−6.4	−16.7	−17.1	−16.6

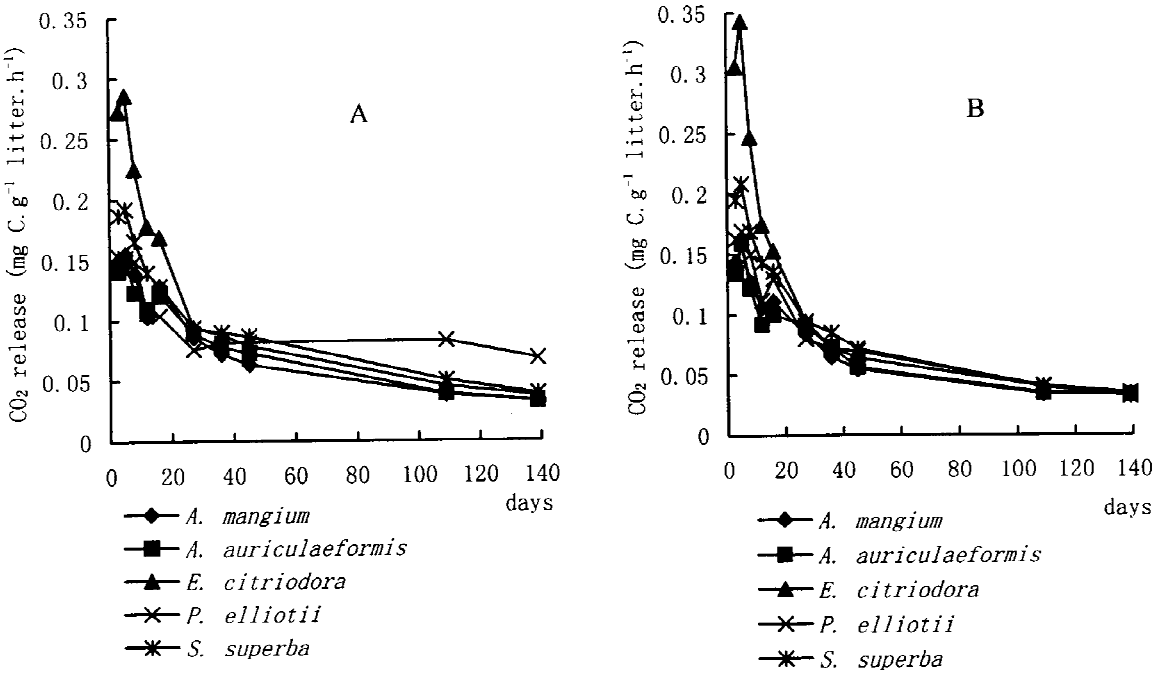

Figure 1. CO_2 –C release rates from litters incubated without (A) and with (B) additional nitrogen.

elliotii litters during the first half of the decomposition experiment. However, at 109 and 139 days, *P. elliotii* litter decomposed at a higher rate. When nitrogen was added, similar patterns of decomposition were found (Figure 1 B), but the differences between species were enlarged because additional nitrogen promoted the decomposition of *E. citriodora* and *S. superba* litter and retarded the decomposition of *A. mangium* and *A. auriculaeformis* litter. After 5–8 days of incubation, the rate of CO_2 –C release from *E.citriodora* litter was almost double the rate from the two legume species. However, all release rates dropped to a similar level at the end of the experiment.

CO_2 –C release rates at different times were generally not correlated with the initial litter quality parameters of N, P, K, Ca, Mg, Fe and Mn. The only significant correlation (p=0.05) was found between CO_2 –C release rate and N at day 45. However, at a lower probability level (p=0.1), CO_2 –C release rates were correlated with Fe on three occasions (days 8, 12, 36) and with Mg on day 27.

Litter from legume species released lower levels of CO_2 –C than non-legume species both with and without the addition of nitrogen (Figure 2) In the first 45 days of litter incubation without additional nitrogen, the cumulative amounts of CO_2 –C released were highest for *E. citriodora* then *S. superba* followed by the other three species (Figure 2 A). However, by day 109, the cumulative CO_2 –C release from *P. elliotii* litter had risen to a similar level to that of *E. citro-*

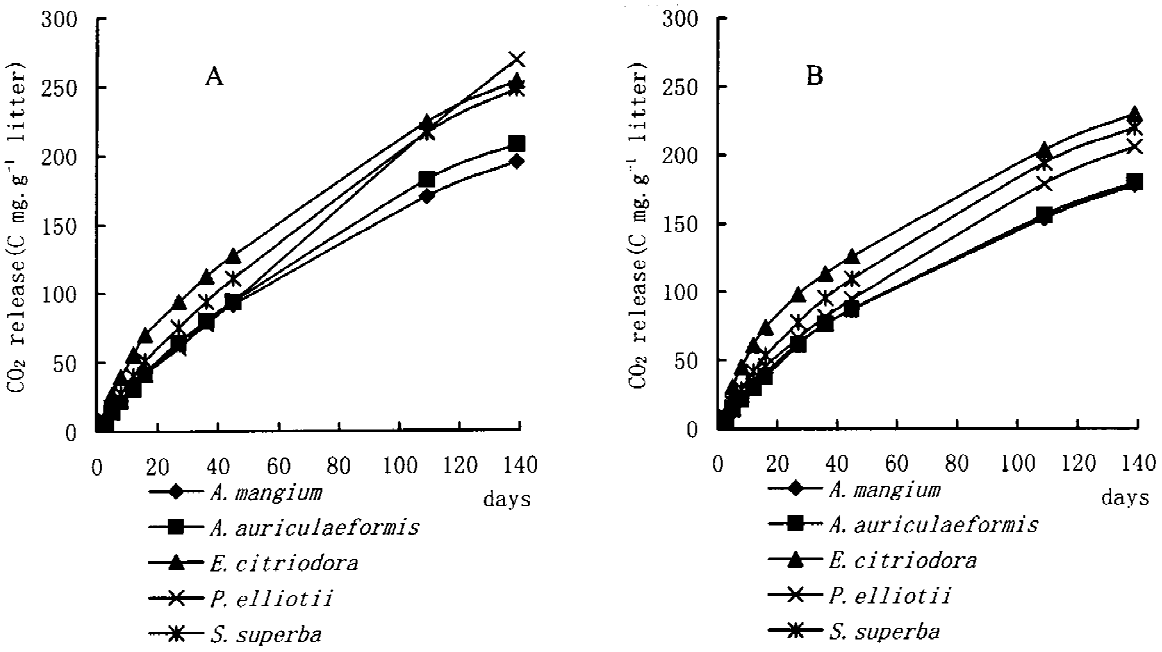

Figure 2. Cumulative CO_2 –C release from litters incubated without (A) and with (B) additional nitrogen.

Table 2. Parameter values for the first order kinetics equation used to describe cumulative loss of CO_2 –C from litter over time

Litter species	Without additional nitrogen (N-)			With additional nitrogen (N+)		
	k (d^{-1})	Half decay (d)	R^2	k(d^{-1})	Half decay (d)	R^2
A. mangium	−0.0017	416	0.9661*	−0.0015	461	0.9522*
A.auriculaeformis	−0.0018	392	0.9786*	−0.0015	459	0.9594*
E.citriodora	−0.0023	300	0.9363*	−0.0021	330	0.8684*
P.elliotii	−0.0022	313	0.9991*	−0.0018	396	0.9699*
S.superba	−0.0023	295	0.9814*	−0.0021	333	0.9517*

* $p<0.01$.

dora and *S. superba*. By the end of the experiment, *P. elliotii* litter had released the most CO_2 –C, and the two legume species the least. With the addition of nitrogen to litter, the pattern of cumulative CO_2 –C release remained the same between species for the whole 139 day decomposition period. *E. citriodora* released the highest levels, followed by *S. superba*, *P .elliotii* and the two legumes (Fig 2 B). Cumulative CO_2 –C release modeled with the first order kinetics equation provided a significant fit to the data for all species with or without nitrogen (Table 2). The k values fell between −0.0015 for *A. auriculaeformis* (+N) and −0.0023 for *S. superba* (-N).

In the field, the amount of CO_2 –C released from the forest floor over 24 h differed significantly between species ($F_{(4,40)}$=6.13, p=0.001) and also between soil depths ($F_{(3,40)}$=6.78, p=0.001). When averaged across all depths, significantly more CO_2 –C was released from the *E. citriodora* forest floor and significantly less from the *A. mangium* forest floor than average (Table 4). When adjusted for the different species, significantly more CO_2 –C was released from the surface of the soil with litter removed and significantly less from 10 cm below the soil surface than average. When CO_2 –C release from the two lower soil depths was compared between species and across days there was a significant difference between species ($F_{(4,49)}$=9.48, $p<0.001$), but not between soil depths ($F_{(1,49)}$=2.97, p=0.091). The amount of CO_2 –C released also tended to increase on the second day ($F_{(1,49)}$=4.66, p=0.036).

Table 3. CO_2 –C emission from the forest ground (mg C h^{-1} m^{-2}). Values are means (standard error), and differences between species and depths are indicated by letters and numbers respectively (Scheffé deviation contrasts)

Depth	Forest species				
	A. mangium^{c}	*A. auriculaeformis*^{b}	*E. citriodora*^{a}	*P. elliotii*^{b}	*S. superba*^{b}
24 hour comparison					
Above soil surface ^{2}	51.74 (5.66)	65.38 (7.27)	65.99 (2.08)	66.75 (6.28)	59.77 (5.12)
At soil surface ^{1}	51.18 (2.84)	57.84 (6.12)	77.42 (5.02)	70.56 (5.92)	64.31 (2.94)
5 cm ^{2}	49.01 (1.85)	57.84 (4.88)	60.27 (1.08)	55.56 (1.83)	57.32 (6.59)
10 cm ^{3}	49.82 (2.23)	49.73 (2.55)	60.78 (3.65)	50.43 (4.67)	52.38 (5.42)
24–48 hour comparison					
5 cm ^{2}	45.05 (5.63)	61.16 (4.24)	80.45 (1.78)	59.49 (4.00)	57.01 (3.50)
10 cm ^{2}	46.65 (5.29)	52.26 (5.06)	68.43 (2.67)	57.42 (6.79)	59.73 (8.64)

Letters denote differences between CO_2 –C emissions from each species and the overall treatment mean and numbers denote differences between CO_2 –C emissions from each depth and the overall treatment mean. a, *1* – significantly higher; b, *2* – not significantly different; c, *3* - significantly lower.

Table 4. The mean (standard error) mineral nitrogen content of soil from six different forest types before and after incubation using two methods (mg kg^{-1} soil). Values followed by the same letter within a column are not significantly different from one another (Scheffé multiple comparisons)

Forest type	Mineral nitrogen before incubation			Aerobic incubation for 30 days			Soil core incubation for 30 days		
	NH_4^+	NO_3^-	NH_4^+ + NO_3^-	NH_4^+	NO_3^-	NH_4^+ + NO_3^-	NH_4^+	NO_3^-	NH_4^+ + NO_3^-
A. mangium	13.81 *a*	4.83 *a*	18.64	19.82 *c*	26.08 *b*	45.90	3.55 *c*	10.52 *a*	14.07
	(0.77)	(0.87)		(1.60)	(0.51)		(0.59)	(2.52)	
A.auriculaeformis	13.82 *a*	5.35 *a*	19.17	19.62 *c*	24.47 *b*	44.09	6.65 *bc*	14.05 *a*	20.70
	(1.21)	(0.34)		(1.84)	(1.14)		(1.68)	(1.28)	
Mixed forest	12.68 *ab*	4.36 *a*	17.04	5.76 *d*	54.35 *a*	60.11	5.38 *c*	11.74 *a*	17.12
	(1.37)	(0.54)		(0.38)	(2.39)		(1.04)	(0.93)	
P. elliotii	13.73 *a*	4.88 *a*	18.61	33.81 *a*	4.19 *d*	38.00	17.60 *a*	3.10 *b*	20.70
	(1.27)	(0.30)		(0.94)	(0.35)		(2.94)	(0.04)	
E. citriodora	8.50 *ab*	4.64 *a*	13.14	23.97 *bc*	6.47 *c*	30.44	17.51 *a*	4.13 *b*	21.64
	(0.60)	(0.35)		(0.50)	(0.50)		(2.05)	(0.51)	
S. superba	7.28 *b*	4.04 *a*	11.32	31.98 *ab*	4.24 *d*	36.22	15.12 *ab*	4.90 *b*	20.02
	(0.91)	(0.49)		(1.53)	(0.06)		(0.58)	(0.55)	

The patterns of release between species across days were the same as in the first 24 h (Table 3).

Nitrogen mineralization

Before incubation, the mineral nitrogen content of soils for all six different forest types was relatively low. Initial ammonium levels were significantly lower for *S. superba* than *A. mangium*, *A. auriculaeformis* and *P. elliotii*, but did not differ significantly between *S. superba*, *E .citriodora* or the mixed forest (Table 4). Initial nitrate levels did not differ between soils (Table 4). After commencement of the incubation, ammonium levels increased rapidly for all soils (Figure 3), then continued to increase gradually in non-legume soils but decreased in the legume soils (Figure 3A). A different pattern was found for nitrate dynamics (Figure 3 B), with nitrate increasing in legume soils to a maximum level of 54 mg kg^{-1} soil in the mixed forest, while the nitrate in non-legume soils remained low and relatively unchanged.

At the end of the incubation period of 30 days, the mineral nitrogen content of soils differed both between species and between incubation methods with the pat-

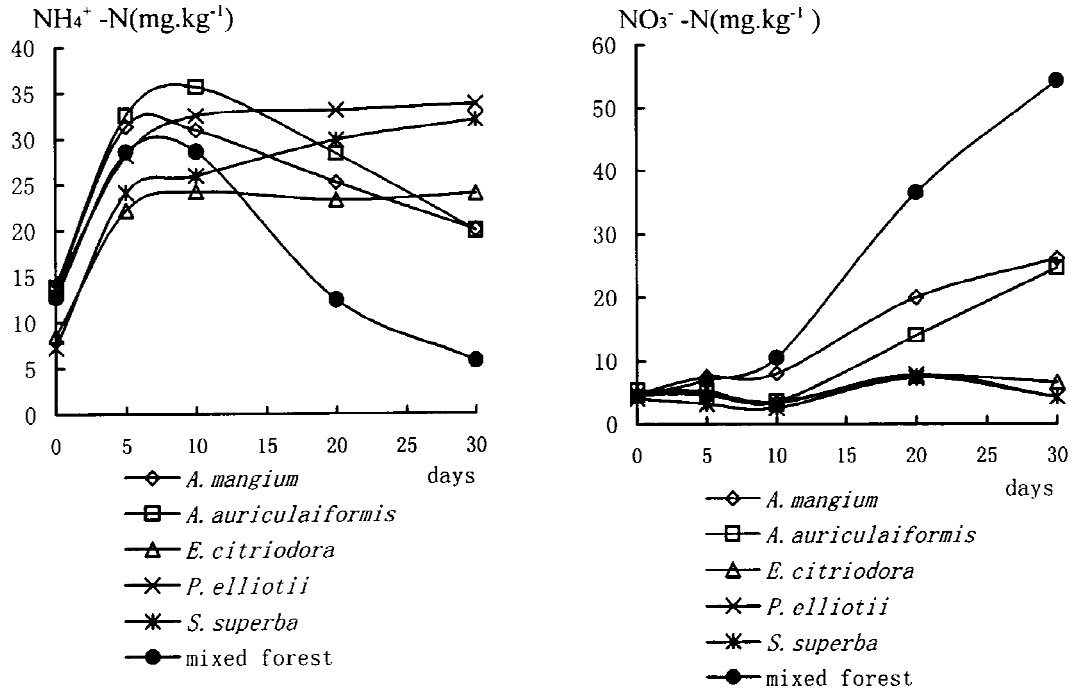

Figure 3. Dynamics of mineralized nitrogen (NH_4^+–N and NO_3^-–N) with time.

tern of change across the species differing between incubation methods. For the aerobic incubation, ammonium levels were highest for *P. elliotii* and *S. superba. E .citriodora, A. mangium* and *A. auriculaeformis* had intermediate levels, with the mixed forest soil having a significantly lower level of ammonium than any of the other species (Table 4). However, for the soil core incubation, *P. elliotii* and *E. citriodora* had the highest ammonium levels with *S. superba* being lower but not significantly different from these two species. *A. auriculaeformis* was significantly lower than *P. elliotii* and *E .citriodora*, but not significantly different from*S. superba*. The mixed forest and *A. mangium* had the lowest levels of ammonia. Nitrate levels from the aerobic incubation fell into four distinct groups (Table 4). Nitrate levels in the mixed forest were highest and significantly different from *A. auriculaeformis* and *A. mangium* which ranked second highest. *E. citriodora* had the 4th lowest nitrate level which was significantly higher that that of *P. elliotii* and *S. superba*. However, in the soil core incubation, the levels of nitrate did not differ significantly between *A. mangium, A. auriculaeformis* and the mixed forest, but these three legumes were significantly higher than non-legume species (*S. superba, E .citriodora* and *P. elliotii*; Table 4). When comparing mineral nitrogen levels obtained using the two incubation methods, the greatest variation in results occurred for the mixed forest. Estimated levels of ammonia were most similar between methods in this forest type, while estimated levels of nitrate were most different.

Discussion

Litter decomposition

In this study, the rate of CO_2 –C released from incubated leaf litters was used to estimate relative decomposition rates between plantation species. As the release of CO_2 –C is mainly a result of microbial processes, it cannot and was not intended to estimate total decomposition rates in the field. However, as the first order kinetics equation of Olson (1963) provided an equally good fit to the CO_2 –C release data as to mass loss data from studies estimating total decomposition using litter bags (Aerts and De Caluwe, 1997; Gallardo and Merino, 1993), it appears that the component of decomposition measured in this study does reflect the overall pattern of decomposition. Additionally, the relative rates of CO_2 –C release between species were the same in the laboratory incubation experiment and when measured directly from the forest

floor. This indicates that the proportion of CO_2 –C released by each species as a result of microbial activity also reflects the overall pattern of decomposition as a result of abiotic factors, mesofauna and microbial action.

Different leaf litters decomposed at different rates, particularly in the initial stages of incubation. However, the lower nitrogen litters of *E. citriodora* and *S. superba* decomposed more rapidly than those from *A. mangium* and *A. auriculaeformis* forests. This is contrary to the hypothesis that short-term litter decay is controlled by nutrients, and nitrogen in particular (Berg, 1984, 1986; Gosz et al., 1973). The addition of exotic nitrogen did appear to increase initial decomposition rates for *E. citriodora* and *S. superba* litters, but it had the opposite effect on *A. mangium* and *A. auriculaeformis* litters. Again, this is in contrast to studies that reported accelerated decomposition rates with either higher naturally occurring nitrogen, or as a result of the addition of exotic nitrogen (Coulson and Butterfield, 1978; Hunt et al., 1988; Sundarapandian & Swamy, 1999; Tian et al., 1992). Titus and Malcolm (1987) also found the addition of urea decreased the decomposition rate of spruce litter. They suggested that in their study microbial activity may be limited by N, even with the addition of urea, as this nutrient was still retained in the litter relative to litter weight loss. There was also a lack of positive correlations between decomposition rates and litter quality in terms of other nutrient levels (N, P, K, Ca, Mg, Fe, Mn) or ratios. Although N or nitrogen related indices (Berg and Staaf, 1980; Coulston and Butterfield, 1978; Flanagan and Van Cleve, 1983; Taylor et al., 1989; Tian et al., 1992; Witkamp, 1966) and P or phosphorus related indices (Berg et al., 1987; Coulston and Butterfield, 1978; Schlesinger and Hasey, 1981; Staaf and Berg, 1982; Vitousek et al., 1994) have often been found to be the major factors governing the decomposition process, they are not the only determinants. Many other factors which are site and species dependent influence decomposition. Those which have been found to have a direct relationship with decomposition include holocellulose (Berg and Staaf, 1980; Fogel and Cromack, 1977; Herman et al., 1977; Meentemeyer, 1978), lignin (Gallardo and Merino, 1993; Melillo et al., 1982), cutin (Gallardo and Merino, 1993), phenolics (Aerts and De Caluwe, 1997) and physical leaf toughness (Gallardo and Merino, 1993). As many of these factors are likely to vary considerably between the species we investigated, their influence on decomposition in plantation forests in southern China warrants further investigation.

It was expected that the effects of adding urea would be most obvious during early stages of incubation since urea undergoes rapid hydrolysis, immobilization and remineralization (Overrein, 1968). However, it was not expected that the rate of CO_2 –C release would be depressed from *A. mangium* and *A. auriculaeformis* litters early in the experiment, or from *P. elliotii* litter during the later stages of the incubation. It has been suggested that an imbalance of nutrients exacerbated by exotic nitrogen could slow down decomposition rates (Aerts and Caluwe, 1997), and this would be most likely for the higher nitrogen legume species. Some long term effects of urea on the total nitrogen levels of humus have been reported (Williams, 1972) and this may have played a role in the reduction of decomposition rates in *P. elliotii* litter. However, it is not clear why only *P. elliotii* was affected in this way.

The relative contribution of the component of decomposition measured in this study to total decomposition is not known. However, low estimated k values and the longer estimated time to half decay than observed in the field confirm that microinvertebrates do make an important contribution to decomposition in this system. Other studies have shown that the proportion of decomposition which occurs as a result of microinvertebrates can vary between 4% to 50% (see Table 1 in Seastedt, 1984; Vossbrinck, 1979). Litter type and quality can also strongly influence the species and population structure of soil invertebrates (Thomas, 1968; Yi et al., 1984). Results presented here indicate that the component of decomposition measured in the incubation studies did reflect the overall pattern of litter decomposition between forest species in the field, without the addition of nitrogen. However, it is possible that the pattern of change in decomposition rates observed with the addition of urea do not reflect the total decomposition rates. Different microarthropod communities present in the litter of different forest species may vary in their response to added nitrogen, resulting in their contribution to decomposition overshadowing that of microbes in some litters but not others. Coulson and Butterfield (1978) found that the animal contribution to decomposition varied much more between different plant substrates than the microbial contribution. Aerts and De Caluwe (1997) also suggested that the composition of the decomposer community may change when litter has been experimentally enriched.

The effect of different plant species on nutrient cycling is determined by both the nutrient release rate from litter and by the total amount of litter that is produced per unit ground area (Aerts and De Caluwe, 1997). Previous studies conducted in the Heshan forests reported that *A. mangium* soils had a deep layer of dark coloured humus, while the humus was shallow in *E.citriodora* soils (Li et al., 1996). Litterfall also varies greatly between these species, with 11.1 t/ha being produced annually by *A. mangium* compared to 2.8 t/ha for *E.citriodora* (Li et al., 2000). When this information is combined with the relative rates of decomposition determined in this study, it indicates that the dense litter accumulation found in *A. mangium* forests is due to a combination of high levels of litter production and low rates of decomposition. Similarly, the scarce litter accumulation in *E.citriodora* forests is contributed to by both low levels of litter production and more rapid decomposition.

Theoretically, CO_2 –C release from upper layers of soil should exceed that from lower layers, since CO_2 –C emitted from lower levels will also be trapped in upper layers. However, the highest rate of CO_2 – C release across all species occurred from the surface of the soil, rather than from litter on the soil surface. Li et al. (1990) also found that the fastest rates of decomposition occurred at the interface between soil and litter. It has also been found that disturbing the soil can accelerate the decomposition of litter (Li et al., 1996). Li et al. (1990) attributed the higher decomposition rates at the interface between the soil and litter to a non-limiting C source and suitable moisture conditions.

Nitrogen mineralization

It is generally considered that levels of nitrification in acid forest soils are low (Keeney, 1980), because autotrophic nitrification is inhibited. It has also been demonstrated that microbial nitrification is significantly decreased with pH levels below 6, and becomes negligible below a pH of 5 (Alexander, 1977). Many workers noted little or no nitrification in incubated samples of forest floors and soils (Geist, 1977; Heilman, 1974; Theobald and Smith, 1974). However, in this study, the results from both aerobic incubation and soil core incubation experiments indicate that the dominant form of nitrogen was governed more by the forest type than the pH value of the soil. The pH values for soils from all forest types were below 4.5, but the nitrate content was high in soils from both single species legume forests and particularly so in soils from the mixed legume forest. In contrast, the dominant form of nitrogen in the soils from non-legume forests was ammonium (Table 4 and Figure 3). Although soils were not examined for acid-tolerant heterotrophic nitrifiers, which may be able to convert organic N to nitrate (Focht and Verstraete, 1977), a positive relationship was found between diversity of decomposer organisms and rate of N conversion in these legume forests (Ding et al., 1996; Li et al., 1996). These findings are in agreement with those of Paschke (1989) who found that nitrate was the major form of mineral N available in mixed plantings of black walnut with N-fixing species (*Elaeagnus umbellata* Thunb), whereas ammonium was the major form of mineral N available where there were very few or no N-fixing trees. Our results indicate that the role of nitrification in the process of nitrogen transformation may be as important in subtropical forests as it has been found to be in temperate forests (Paschke, 1989).

Both methods used to estimate nitrogen mineralization have limitations. The much higher estimated level of nitrogen mineralization using the aerobic soil incubation method indicated that it might overestimate actual levels in the field. Frazer et al. (1990) reported similar results and warned of the danger of using such laboratory results to predict field behavior, although the method is widely accepted (Keeney, 1980). Increased aeration and disturbance during the treatment of samples is mainly responsible for the enhanced level of nitrogen mineralization. Therefore, the estimates of nitrogen mineralization using the soil core incubation may be closer to reality, although these samples were also manipulated in the laboratory. Despite the differences in the patterns of nitrogen mineralization of soils from different forest types between incubation methods, the gross patterns between legume and non-legume species were the same regardless of the method.

Our results clearly demonstrate that both the higher nitrogen levels of legume litters and their relatively low rates of decomposition are important factors in the build up of nitrogen stocks in the soils of legume forests. The mineralization experiment demonstrated that soils in which legume species were grown had a higher nitrogen mineralization potential than those in which non-legumes were grown. The improved nitrogen fertility of soils where legumes have grown can in turn result in the soil being more easily colonized by other forest species (Yu and Peng, 1995).

Acknowledgements

We gratefully acknowledge the support of the National Natural Science Foundation of China (Projects 39899370 and 39928007), Chuang Xin Project, Chinese Academy of Science (Project KZ951-B1-110), Heshan Experimental Station Open Foundation and South China Institute of Botany Foundation (Chinese Academy of Science).

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