MITTEILUNGEN
DER
ÖSTERREICHISCHEN
BODENKUNDLICHEN
GESELLSCHAFT

Heft 47

Wien 1993

Exkursion
der Österreichischen Bodenkundlichen Gesellschaft
15./16. September 1994

Exkursionsziel: Waldviertel

Im Rahmen der diesjährigen Exkursion ist die Besichtigung einer Reihe von repräsentativen land- und forstwirtschaftlichen Profilen des Waldviertels geplant, die zudem auch Standorte der niederösterreichischen Bodenzustandsinventur sind. Im folgenden soll der Exkursionsverlauf kurz wiedergegeben werden:

15. September

8^{00}	Abfahrt Wien West Besichtigung der Profile Mödring und Hötzelsdorf, Sabatenreith, Göpfritz und Kottschalling Nächtigung in Zwettl

16.September

8^{00}	Abfahrt, Besichtigung der Profile Purrath, Arbesbach und Schiltern sowie eines Aufschlusses
ca. 16^{30}	Ankunft Wien West

Für die Fahrt steht ein Autobus zur Verfügung, ein Exkursionsführer wird beigestellt. Der Exkursionsbeitrag beläuft sich excl. Nächtigung auf öS 400.-. Für die Übernachtung wurde pauschal reserviert, die Zimmereinteilung erfolgt während der Fahrt (Preise von öS 250.- bis öS 450.-), Anmeldeschluß ist der 16. August 1994. Wir ersuchen um Verständnis dafür, daß Anmeldungen nur in Verbindung mit der Bezahlung des Exkursionsbeitrages berücksichtigt werden können.

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Eigentümer, Herausgeber und Verleger:
Österreichische Bodenkundliche Gesellschaft
Gregor-Mendel-Straße 33, A-1180 Wien

Schriftleitung und für den Inhalt verantwortlich:
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Druck: RM - Druck- & Verlagsgesellschaft m.b.H., Graz

Gefördert durch das Bundesministerium
für Wissenschaft und Forschung in Wien

ISSN 0029-893 X

Der Großteil der Beiträge in diesem Heft ist, von einigen kleinen "Ausrutschern" abgesehen, ausnahmsweise in englischer Sprache abgefaßt, da über einen internationalen Kongreß berichtet wird.

Wir sind dennoch zuversichtlich, mit diesen aktuellen Beiträgen einen großen Leserkreis ansprechen zu können.

M. Eisenhut *O. Nestroy*

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INHALTSVERZEICHNIS

Kurzfassungen der Vorträge

Othmar Nestroy - Ehrenmitglied der ÖBG

Die ordentliche Generalversammlung der Österreichischen Bodenkundlichen Gesellschaft hat am 19. 1. 1994 über Antrag des Vorstandes beschlossen, in Anerkennung des Wirkens für die Bodenkunde und für unsere Gesellschaft Herrn tit.A.o.Univ.-Prof. Dipl.-Ing. Dr. Othmar Nestroy die Würde eines **Ehrenmitgliedes der Österreichischen Bodenkundlichen Gesellschaft** zu verleihen, die ihm anläßlich des Festaktes zum 40jährigen Bestehen unserer Gesellschaft überreicht wurde.

Geboren 1933 in Kapfenberg, absolvierte Nestroy das humanistische Gymnasium in Wien und studierte an der Universität für Bodenkultur in Wien, Landwirtschaft; 1956 schloß er dieses mit dem akademischen Grad Diplomingenieur ab.

Weitere Stationen in seinem Leben sind:

1956 - 60	Assistent am Institut für Bodenforschung (Prof. Franz), Abschluß der Dissertation über "Wirkung von Zweischichtackerung und garefördernder Fruchtfolge bei verdichteten Böden der Oststeiermark"
1960 - 70	Landwirtschaftlich-chemische Bundesversuchsanstalt (heutige Bundesanstalt für Bodenwirtschaft): Leitung des Labors, Durchführung von Kartierungen, Analyseninterpretationen, Redaktion der Bodenkartierungen
1970 - 86	Oberassistent am Institut für Geografie der Universität in Wien bei Univ.-Prof. FINK (Lehrkanzel Physische Geografie), später bei Univ.-Prof. FISCHER
1974	Habilitation in dem Fachgebiet Physische Geografie mit besonderer Berücksichtigung der Bodengeografie
1984	tit.A.o.Univ.-Prof.

1986 bis dato	Institut für Technische Geologie und Angewandte Mineralogie der Technischen Universität Graz

Nestroy verfaßte 127 Publikationen unter anderem in reviewten ausländischen Medien (z. B. European Journal of Soil Conservation).
Sein Hauptfachgebiet ist die Bodengeografie und Agrarökologie.
Aufgrund seines beruflichen Werdeganges ist er in vielen Fachgebieten der Bodenkunde zu Hause. Seine Arbeitsgebiete umfassen: Bodenkartierung, chemische und physikalische Bodenanalysen, Bodenphysik, Bodensystematik, genese und Geomorphologie. Er verfaßte spezielle Arbeiten zur Bodenerosion und Bodenökologie in den alpinen Räumen Österreichs und Griechenlands.
O. Nestroy liest auch an drei Universitäten und pendelt dabei seit Jahren zwischen Wien und Salzburg.

Unser besonderer Dank gilt seinen Verdiensten für unsere Gesellschaft.

Seit den ersten Jahren unserer Gründung steht er für die Interessen der ÖBG hilfreich zur Verfügung. Er war von 1986 bis 1989 Präsident und seit 1980 ununterbrochen Schriftleiter - nur während seiner Präsidentschaft war er formal zweiter Schriftleiter - somit verantwortlich für die Qualität des wissenschaftlichen Outputs unserer Gesellschaft. Er hat mehrere Exkursionen fast im Alleingang vorbereitet. Wir alle schätzen seinen unermüdlichen persönlichen Einsatz für die ÖBG, die sein innerstes Anliegen ist.

Die Österreichische Bodenkundliche Gesellschaft hofft, O. Nestroy durch die Verleihung der Ehrenmitgliedschaft den gebührenden Dank zum Ausdruck gebracht zu haben.

Wir wünschen Dir auch in den kommenden Jahren viel Erfolg und bitten Dich um weitere Unterstützung für unsere Gesellschaft.

E. Klaghofer

Im nun folgenden Abschnitt dieses Heftes der "Mitteilungen der Österreichischen Bodenkundlichen Gesellschaft" sind neun Arbeiten zum Thema Abbau von organischer Substanz wiedergegeben, die anläßlich des EG-workshops "Role of invertebrate and microorganisms in decomposition and soil organic matter formation" vom 22. - 24. April 1993 in Innsbruck präsentiert wurden.
Dieser workshop war der zweite workshop in der Reihe der EG-konzentrierten Aktion "Decomposition and accumulation of organic matter in terrestrial ecosystems" und hatte zum Ziel, Wissenslücken auf dem Gebiet des biologischen Abaus aufzuspüren und Forschungsprioritäten zu formulieren.

Heribert Insam

Mitt. d. Österr. Bodenkundlichen Gesellschaft, H. 47, S. 11 - 41, 1993

DECOMPOSITION OF LITTER AND SOIL ORGANIC MATTER - CAN WE DISTINGUISH A MECHANISM FOR SOIL ORGANIC MATTER BUILDUP?

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Introduction

The buildup of soil organic matter is a slow process and often spans generations of scientists, which makes it difficult to follow the buildup as well as the mechanisms controlling the buildup. The most common response to the question is that "the buildup depends on litter formation being faster than the decomposition processes". Although this is true, in principle, it seems a bit trivial and such a statement does not really provide us with an acceptable explanation for the buildup as such. It lacks a proximate mechanism and could not distinguish, for example, a difference in buildup rates between species.

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That the amount and quality of the litter formed are important factors is evident and that decomposition and disappearance of litter determines the rate of buildup of a soil organic matter layer is also evident. This knowledge is, however, not enough and additional information about causal relationships is badly needed to allow us to understand the process and to predict the outcome of the buildup in single cases.

Numerous attempts have been made to use mathematical models to describe organic matter decomposition. Among the earliest and most widely used models is the simple exponential model, first proposed by Jenny et al. (1949), but more commonly attributed to Olson (1963). Critics of this simple model have pointed out that because of the nonhomogeneous nature of litter (e.g. solubles, celluloses, lignin) (Minderman, 1968) and the various processes involved in mass loss (leaching, cellulolysis, faunal processing) (Anderson, 1973) that more complex models are required.

Field studies have confirmed the idea that organic matter decays with a decreasing instantaneous rate of decay. If the decay rate declines to zero before all of the organic matter has been decomposed the percent mass remaining over time will approach an asymptote. An asymptotic form of model for mass loss during decay has been proposed by several workers.
Howard and Howard (1974) incubated litter in glass tubes to exclude the influence of soil fauna. They concluded that an asymptotic model provided the best fit to their data. We will elaborate on their model later. Subsequently, Carpenter (1981;1982) compared several models and introduced two of his own. Although only one of the five models that he compared included a provision for an asymptote, he suggested that such an inclusion could be important for a model of long-term organic matter dynamics.

In a study on limit values for decomposition in a system where faunal activity was relatively unimportant, Berg and Ekbohm (1993) compared six sets of decomposing Scots pine needle litter with six sets of decomposing lodgepole pine needle litter and observed a highly significant difference between the two different litter types with Scots pine needles having an average limit value of 84 percent accumulated mass loss and lodgepole pine having a limit value close to 100 percent.

Support for this form of model is not limited to faunal exclusion studies or sites with minimal faunal influence. Aber et al. (1990) studied decay of a number of temperate forest litters, including some mull sites, over a period of five or more years and concluded that decomposition could be described in two phases. During the first phase, simple exponential decay provided a good fit to observed mass loss. During the second phase, mass loss became extremely slow. For the range of litters examined, they concluded that phase one (exponential decay) extended to about 20 percent mass remaining.

Further evidence that supports the asymptotice nature of organic matter decay comes from radiodated soils and chronosequence studies. Carbon-14 dating indicates ages of soil organic matter in the vicinity of 1000 to 10000 years (Paul, 1984;Stevenson, 1982). Numerous chronosequence studies support the idea that, in the absence of disturbance, amounts of organic matter continue to increase in soils over centuries or even millenia (Jenny, 1980).

It has been possible to relate the level of this estimated asymptote to initial concentrations of some nutrients such as nitrogen and manganese (Berg et al., 199X), factors that could regulate both size and composition of the microbial community, with the latter being especially crucial to decomposition of recalcitrant substances.

If it is true that different litters contribute different proportions of their mass to pools of stable organic matter it could mean that with a constant amount of litterfall we may expect very different responses in the buildup of soil organic matter depending on the chemical composition of the falling litter, both within a species and over species.

Although we clearly have a severe lack of good litterfall data as well as data on litters' chemical composition, collected in a scientifically acceptable way, we attempt, in this manuscript, to suggest a possible mechanism for soil organic matter buildup with focus on the decomposition process as such.

A BUDGET APPROACH FOR SOIL ORGANIC MATTER IN SCOTS PINE STANDS -FROM ONE SITE TO A REGION

General

In this attempt to establish a budget for organic matter we first used information from one site supplying us with extensive data. Using our site-specific model as a guideline we also attempt to generalize our observations on development of the S.O.M. layer to a region.

One site - the SWECON site, general background and site data

We have selected the forest stand of the former Swedish Coniferous Forest Project (SWECON) to create a dynamic budget for the organic soil layers. The background data and site history are well documented, thus providing a starting point for budgets in a regional perspective.

Amounts of organic matter measured in most components of the stand and average annual litter production flows (Figure 1) give a static organic-matter budget for the forest at an age of about 120 years. The numbers given are based on a good set of measurements and the original compilation of data was published by Andersson et al. (1980). Litterfall data were measured for the period 1973 - 1980 (cf Flower-Ellis, 1985;Berg et al., 1993). Root production estimates were to uncertain to be included in the present budget. Decomposition studies were long-term, using different litter components, including needles (e.g. Berg et al., 1982). The chemical composition of different litter components was also investigated (Berg, 1981). With this background information it appears reasonable to develop a budget for organic matter.

The stand was a Scots pine monoculture, about 120 years old in 1973, located at Jädraås in Central Sweden (60°49'N; 16°30'E) at an altitude of 185 m a.s.l. on a flat area of deep sand sediments. The mean annual precip. is 609 mm and the mean annual temp.is +3.8°C. Two younger stands (18 and 55 years old) on similar nutrient-poor soils were investigated for litter fall. The forest has been described in earlier papers (e.g. Axelsson and Bråkenhielm, 1980; Berg et al., 1982).

A very loose A_{OO} horizon, interwoven with living mosses and lichens, covers a combined A_{01}-A_{02} horizon of 5-10 cm. The latter organic layer was very well separated from the mineral soil. The parent mineral material as well as the whole soil is considered to be very poor in essential nutrients. In the middle part of the last century the forest at this site was burned completely, including the existing organic layers, leaving charcoal at the bottom of today's organic layer.

Litter production

Trends in total litterfall

At the site, litterfall was observed for seven to ten years in three Scots pine stands, initially 18, 55 and 120 years of age (Table 1). Some trends with time were evident within stands, and there were clear differences between them. A detailed analysis on annual fall as well as on single litter components such as needles, cones, branches, and fine litter has been given by Berg et al. (1993) and by Flower-Ellis (1985).

The series of observations revealed an increase in total litterfall in all three stands with stand age (cf Berg et al., 1993). The stands represented the age period of 18 - 25 years, 55 - 65 years and 120 - 130 years, giving a relatively good distribution among ages.

The methods used by Flower-Ellis (1985) to determine total litterfall was very much adapted to litter type and the reader is referred to Flower-Ellis (1985) and Flower-Ellis and Olsson (1978) for a full description of their methods.

Relation between litterfall and stand age - two models

General

A linear regression of litterfall on stand age, using data from the Jädraås site alone, gave a good relationship for the two younger stands only (18 and 55-year-old). For the 18-year-old stand, during the seven years for which data are available, use of the total amount of litter gave a significant positive relationship. The 55-year-old stand also showed a significant, positive relationship for the ten years during which it was followed (Table 1). This indicates that during the age interval concerned, the amount of litter was still increasing. However, this trend seems not to be an effect of age alone, but includes a temporal trend which is common to all three stands (Berg et al., 1993). The old stand did not fit satisfactorily into the overall picture. At an age of 120 years, it may no longer have been expanding its needle mass; but there was no indication that this was declining, during the period of observation. The average litter fall during he ten-year study (1973 - 1983) was 2212.4 kg ha^{-1} in the 55-year-old stand which was significantly higher than the average of 1621.5 kg ha^{-1} in the 120-year-old stand.

It should be noted that the 55-yr-old stand was situated on a relatively more fertile site which could explain the somewhat higher level of litter production in that stand as compared to the 120-yr-old stand. The ten-year average of 1620 kg ha^{-1} is higher than the 1420 kg ha^{-1} previously reported (Andersson, et al., 1980), due to a new estimate based on 10 years of data Berg et al. (1993).

Models

We will consider two alternative models for litterfall as a function of stand age. The first model assumes a linear increase in litter fall from 16.2 kg ha^{-1} in year 1 to 1620 kg ha^{-1} in year 100 with litter fall remaining constant thereafter. The second model assumes a logistic increase in litter production.

The logistic model can be stated as:

$$\frac{d\,LF}{dt} = g\,\frac{(Max - LF)}{LF} \qquad (1)$$

where;

LF = annual litterfall
Max = maximum ("steady- state") annual litterfall
g = constant intrinsic for rate of increase in litterfall with age

Using serial approximations to achieve the best fit to data from both the 18-yr-old and the 120-yr-old stand, the following parameters were derived.

Max = 1620
g = 0.37

These two models are illustrated in Figure 2 a and b, along with actual data points from the Jädraås litterfall studies.

Litter decomposition - maximum decomposition - limit values

Observations on the phenomenon of maximum decomposition

For most litter types it has been observed that the decomposition rate becomes lower as decomposition proceeds while the more easily metabolized components are decomposed and the recalcitrant ones remain. For some litter types in late stages it has been observed that the decomposition in late stages of decay becomes so slow that no significant mass loss has been measurable.

Howard and Howard (1974), working at Merlewood, presented estimated final values for mass loss of some leaf litter types that were decomposing in the absence of soil fauna. In their approach they found that asymptotic functions described their data well. The interpretation of their finding was (cf Figure 3) that in the absence of soil animals the decomposition could stop or at least come close to a halt and reach a limit value well below 100 percent mass loss. Using data from a nutrient-poor forest, almost free from soil animals, Berg and Ekbohm (1991) tested long-term mass-loss values for such limits. They found that for seven, chemically very different litter types, limit values could be estimated ranging from about 50 percent remaining, up to no remains at all for one specific litter type (Figure 4). A later study (Berg and Ekbohm, 1993) gave results that confirmed the tendency for some litter types to reach certain, specific limit values. They found that within two pine species (Scots pine and Lodgepole pine), each consisting of six different and independent sets of decomposing needle litters there was a considerable homogeneity in limit values. For Scots pine needles an average amount of 11 percent remained.

Method of estimation

In their study, Howard and Howard (1974) compared several statistical models for such a case and preferred an asymptotic non-linear model with three parameters. Later, Berg and Ekbohm (1991) modified their function somewhat and obtained;

$$m.l. = m(1-e^{-kt/m}), \qquad (2)$$

where m.l. is the accumulated mass loss (in %) and t is time in days. The parameter m represents the asymptotic level which the accumulated mass loss will ultimately reach and the parameter k is the decomposition rate at the beginning of the decay.

A possible interpretation of the phenomenon of limit values

Howard and Howard (1974) used an experimental environment with no soil animals present. This may be a good basic condition to the phenomenon of asymptotes as the mechanical treatment of the litter is minimized or nonexistent. Most of the analyses used for the present study were from systems which were either free from soil animals or had a soil fauna with a very low activity towards litter, at least in the early decomposition stages. It is thus possible that the asymptotes reflect where mass loss due to microbial activity comes to a halt and some other agent is needed for the process to continue. However, even if the existence of an asymptote may be attributed to the absence of soil animals other studies, cited above indicate the asymptotic nature of mass loss in a variety of forested ecosystems. A set of investigations were thus carried out by Berg et al. (199X) comparing and synthesizing the data available.

In an analysis, Berg et al. (199X) used existing data and compared the litters' initial chemical composition and limit values for decomposition. All limit values were compared to the initial nitrogen concentrations which gave a highly significant, negative relation. For example, all data combined gave the negative relation r= -0.706; n=28;p<0.001 (Figure 5a). In this analysis both data from field studies and those from the Merlewood microcosms (Howard and Howard, 1974) were used and they seemed to fit to the same line. The higher the initial nitrogen level the more organic material should be stored.

There may be a causal relationship behind this observation. As decomposition of the litter proceeds, the concentrations of lignin and lignin-like substances increases. This is due to two phenomena. First, because these substances are highly resistant to decay, they are decomposed more slowly than more easily degradeable substances, resulting in an increase in their concentration. Second, lignin-like compounds (and other forms of stabilized organic matter) are synthesized during the decay procss. Many of the enzymes responsible for lignin degradation can, under appropriate conditions, catalyze the polymerization of lignin-like substances (e.g. Kondo et al., 1990; Lobarzewski et al., 1982). Furthermore, nitrogen containing compounds have been shown to react with aromatic substances in the soil, yielding recalcitant humic compounds (Nõmmik and Vahtras, 1982;Bollag et al.,1983; Liu et al., 1985).

The composition of the microbial community is very much dependent on the chemical input into the system. This includes not only the availability of the energy source but also the concentrations of different nutrients. The interaction of nutrients may be complex and although, for example, a good supply of nitrogen may stimulate the initial decomposition rate of litter, an abundant supply of nitrogen may have a reversed effect in somewhat later stages. Properties such as nutrient content or organic composition can affect the composition of the microbial community. Relatively few species of microorganisms can decompose lignin completely. The members of this lignin-degrading group are generically referred to as white-rot organisms. Keyser et al. (1978) showed that a white-rot fungus did not synthesize its lignin-degrading enzyme system in the presence of low molecular, nitrogen-rich compounds such as ammonium and amino acids. Fungi with such properties would naturally be hampered in their development when growing on a far-decomposed, lignin-rich litter which was also rich in nitrogen. As a good number of white-rot fungi have been found to have this property (P. Ander, pers. comm.), a high nitrogen level could retard lignin degradation and consequently also the decomposition of far-decomposed litter in several soil systems.

When testing the relation between initial concentrations of manganese and the limit values all available data gave the highly significant, positive relation r=0.757;n=17;p<0.001 (Figure 5b).
This nutrient has been demonstrated to be essential to the activity of manganese peroxidase, a lignin degrading enzyme (Perez and Jeffries, 1992). It is also involved in the regulation of other lignolytic enzymes including laccase (Archibald and Roy, 1992) and lignin peroxidase (Perez and Jeffries, 1992;Paszcynski, et al., 1985). Although the causal relationships are unclear, there was a significant relation, suggesting that manganese concentrations may influence limit values.

The fact that a relationship (although mainly empirical) exists between limit values and initial nitrogen concentrations, as well as to those of manganese, is good support for a strong rate-retarding mechanism.

Do the estimated limit values indicate a halt in the decomposition process ?

Although it has been possible to estimate significant limit values for litter mass loss by using asymptotic functions we cannot conclude that such limit values necessarily reflect completely undegradeable remains of the litter. The estimated remains could equally well reflect a fraction that would be fairly stabilized and thus be decomposing at a very low rate. However, this would not make the phenomenon any less interesting, especially if we could connect this recalcitrance to properties of the of the litter or its environment, e.g. lignin concentration, nutrient status or climate. The fact that allophanic organic matter exists shows that an "infinite" storage is possible and although allophanic organic matter may be regarded as "extreme" the level of stabilizing components necessary for a halt in the decomposition process is today not defined (e.g. Paul, 1984).

Soil organic matter data for the stand

In a measurement of the amounts of soil organic matter in the 120-year-old stand Staaf and Berg (1977) estimated - with the use of large-diameter samplers - the amount of "humus" in the combined A_{01} - A_{02} layer to be 15400 kg ha^{-1}. This did not include visible litter remains.

A model for a budget

With regard to the importance of the litter´s chemical composition (cf Figure 5) it is important to note that the litter produced from the pines was chemically similar from all parts of the trees (with the exception of the rough bark)(Berg, 1981) and that the main part of the shrub litter had a chemical composition reasonably close to that of pine (Berg, 1981). In addition, the litter formation from the pines dominated the litter input in this stand and we have thus assumed that the decomposition process in the late stages was similar among all litters leaving recalcitrant or stabilized remains of about 11 percent of the litter fall. Using Eq. 1 we estimated that it takes 8 years for fresh Scots pine nedle litter to be reduced to within 1 percent of the asymptotic level. First we calculated litter input for each of the 112 years (last 8 years have yet to contribute stable organic matter). Then, assuming that 11 percent of each year's input becomes stable, we calculated the annual amount of stable organic matter each year. Summation over the 112 years gave an estimate of accumulated organic matter. For the linear model we obtained a figure of 11370 kg ha^{-1} (Figure 6). Using the logistic model, described above, to predict litterfall, for the same period of time we estimate an accumulation of 16670 kg ha^{-1} of soil organic matter (Figure 6). Neither of these estimates included belowground litter inputs but compared favourably with the direct measurements of 15400 kg ha^{-1} (Staaf and Berg, 1977).

The larger estimate produced by the logistic model is due to the fact that the logistic model predicted much higher litter input in the early yars of stand development. The logistic model predicts that the stand reaches maximum litter production after only 30 years whereas the linear model assumed that maximum was not a attained until year 100 (see Figure). When the former stand at this site burned in the middle of the 19th century, practically all of the organic layers were burned too, leaving just charcoal remains. The fact allows us to easily identify and date the organic matter. The model does not include input due to root litter and ground vegetation or output due to downward transport of organic matter into the mineral soil. Although these two processes would, to a degree, counteract one another, neither is known with enough precision to be included in the present model. The difference between the estimated and the determined amount was 26 percent lower for the linear model and about 8 percent higher for the logistic model. Such differences are reasonable.

Budget estimates for soil organic matter in Scandinavia

Litterfall

A transect of Scots pine sites across Scandinavia

The litterfall data, originated from 33 Scots pine sites. For three plots litterfall was followed for at least ten years and for most plots up to three years. The geographical range for these sites varied from north of the Arctic Circle (66°32'N) to south Sweden (58°07'N) which covers about half of the north-south distribution of Scots pine in Europe. For all of these sites information was presented about needle litter fall (Albrektsson, 1988;Johansson, 1986;Flower-Ellis, 1985) and about half of them also had total litterfall measured (Johansson, 1986;Flower-Ellis, 1985). A synthesis of these data was made by Berg et al. (1993).

In an attempt to construct a regional model we compared annual fall of needle litter as well as total litter, to stand age, basal area, site index and latitude, using available data (Tables 2 and 3).

The variation in mean annual needle litterfall ranged from 530 kg ha^{-1} to 2312 kg ha^{-1}. The lowest amounts were found at nutrient-poor sites, generally with sediment soil, in the north. At more nutrient-rich sites with till deposits or clays, litterfall mass was higher. Among sites of similar fertility, needle litterfall was lower for sites situated in the north than for sites with a more southern situation. This is evident when needle litterfall at two extreme sites, at latitude 66°32'N and latitude 58°04'N, are compared. These sites had nearly identical site indices (17 m and 16 m, respectively) and basal areas (17.5 m^2 ha^{-1} and 18.3 m^2 ha^{-1}, respectively) but the needle litterfall at the northern site (608 kg ha^{-1} yr^{-1}) was only about one third of the amount obtained at the site located in the south (1571 kg ha^{-1} yr^{-1}).

The mean annual total litterfall (larger twigs and branches excluded) varied from 735 kg ha^{-1} to 4198 kg ha^{-1}. The proportion of needle material in the total litterfall, which comprises needles and fine litter, varied between 33 and 75 percent in the stands studied. However, in most of the stands the needle litter fraction accounted for about 50 to 60 percent of the total annual litterfall. According to

Bonnevie-Svendsen and Gjems (1957) and Mälkönen (1974), needle litterfall is approximately 75 percent of the total litterfall in Scots pine stands. Their values are higher than ours, despite the fact that larger twigs and branches are not included in our figures. Although the methods used in the two different transect studies were not ideal for e.g. determination of the quantification of cones and branches litter we have used data for estimates in a regional budget (below) to indicate trends.

Linear relations of litterfall in Scandinavia to stand age, latitude, basal area,and site index

Latitude; We obtained a negative relationship between latitude and litterfall (both total litterfall with r^2=0.39;n=17;p<0.01 (Table 2) and needle litterfall (r^2=0.48;n=33;p<0.001) (Table 3). This may be compared with the results of Albrektson (1988), who also found a negative relationship for needles (r^2=0.46;n=16).

Site index; Site index values were available for all the Scandinavian sites. All needle litter data combined, gave an r^2 value of 0.42 (n=33; p<0.001;Table 2). Total litter gave an r^2 value of 0.50;n=17;p<0.01 (Table 3).
Stand age; When we compared all available data for needle litter to stand age we obtained an r^2 value of 0.27 (n=33;ns) for needle litter (Table 2). For total litter the r^2 value was 0.19 (n=17: ns) (Table 3).

Basal area; A comparison of litterfall to basal area gave a coefficient of determination (r^2) of 0.52 for total litterfall(n=17;p<0.001)(Table 3). When all available data for needle litter fall were combined (Table 2), basal area gave a highly significant, but poorer relationship with (r^2=0.26;n=33; p<0.01).

Multiple linear relations for litter fall of both needles and total litter, using all Scandinavian data

Relationships on latitude and site index were highly significant (n=33) and gave r^2 values of 0.48 and 0.42 respectively. In multiple linear relationships, the combined factors gave significant or highly significant relationships with r^2 values from 0.49 for site index plus basal area, to 0.71 for latitude plus basal area. A three-factor model using the variables, latitude, site index and basal area, gave an r^2 value of 0.73, thus explaining about 3/4 of the variation in needle litterfall (Table 2). For total litter fall an r^2 value of 0.77 was reached when latitude, site index and basal area were included (Table 3).

Soil organic matter

No quantitative data for amounts of soil organic matter existed in these transects.

Decomposition of litter and limit values

For Scots pine needle litter decomposing in the transect of Johansson, limit values were estimated for the whole climatic range of sites (M.B. Johansson, pers. comm.). Although some trends in limit values apparently were dependent on climate and the sites' nutrient status (B.Berg, M.B. Johansson, unpubl.) they did not seem to be of a magnitude worth considering in the present budget estimate. We have therefore used the value of 11 percent litter remaining, estimated for the Jädraås site for the whole transect range.

Growth of soil organic matter layers on a regional level

Using the data presented above and the equations of Tables 2 and 3 we can estimate the growth of organic matter layers over a larger region. We can illustrate this with the use of a change in latitude from, say northern Scandinavia to central Germany or south Poland. Let us take first needle litter and then total litter. The needle litter fall would be about 810 kg ha^{-1} at a latitude of 66° and at 49° (cf Figure 7) it would be 3316 kg ha^{-1}, meaning that the SOM layer may grow about 4 times faster in the south just due to the needle component. For total litter we would have a figure of 3.5 times as fast assuming then that the limit values would be similar in both cases and that the stands would have similar nutrient status.

If we extend the calculations and use the assumption that the stand in the south would grow faster and reach a maximum canopy in, say, 40 years as compared to 110 years in the north we can modify the estimates. Using the linear model for litter production versus stand age, (Figure 2) and a remaining recalcitrant component of 11 percent. We estimate that in the northern stand the needle component in litter fall would have added about 4900 kg ha^{-1} to the soil organic matter layer in 110 years. In the south the stand would have added 7500 in 40 years, in both cases until the canopies reached a maximum. Such an estimate can be considered as conservative when compared with the estimates by Romell (1932) for a Norway spruce forest. He measured the amounts of soil organic matter to a depth of 100 cm and found 300000 kg ha^{-1}. If we assume that this site and its liter quality are similar to that of the Jädraås site we can estimate how long it would take to accumulate 300 000 kg ha^{-1} or organic matter. As Rommel's masurements extended to one meter we have doubled litter inputs from 1620 to 3240 kg yr^{-1} to account for the belowground input. Using the limit value for Scots pine needles of 11 percent and assuming constant litter input it would take 840 yars. Fires, diseases and storms or other disturbances may reduce litter input and increase the time required for the measured accumulation. Perhaps more important, a different limit value for the litter input could influence the time required to accuire the accumulation. The available data, limited as they are, while not providing support for the asymptotic model, clearly do not reject the model.

This kind of calculations may be extended, elaborated, and applied to specific questions. However, it should be emphasized that we have been presenting studies from boreal pine forests and the results obained and the conclusions drawn are not necessarily applicable to other systems.

CONCLUDING REMARKS AND RECOMMENDATIONS

We can now ask whether we have an appropriate method to estimate the growth rate of the soil organic layer in mor forests. This approach is, of course, sensitive to the accuracy of litter input data (both root litter and litter fall data) as well as to the limit value for litter decomposition.

The extension of Scandinavian data to the northern half of continental Europe is an exercise that lacks validation data. The results predict a considerable difference in the growth rate of the soil organic matter layer as a consequence of a difference in latitude. The magnitude may seem alarmingly large with respect to the possibility of a corresponding storage of nutrients.

The examples given above focus on a certain group of Scots pine stands. We may expect even more pronounced differences between north and south when considering other species e.g. spruce. Deciduous trees could also be expected to differ. It is also important, though, to emphasize that the studies have dealt with litterfall of leaf and needle litters which are more nutrient rich than wood components. The effects described do not therefore necessarily describe the effects of wood components on the organic matter buildup.

We make the following recommendations for research efforts necessary to fill vital gaps in our knowledge.

- It appears self evident that a good basic knowledge about lignin-degrading organisms and their ecology is absolutely vital for further progress in the field of soil organic matter dynamics.
- As budgets of dead organic matter make up the basic starting point for budgets of nutrients in forest stands some basic approaches are necessary;
  (i) well coordinated litter input measurements in selected stands in transects
  (ii) a generalization of the concept of limit values to a range of litter and forest types as well as an understanding of the causal relationships for their magnitude.

REFERENCES

ABER,J.D., MELILLO, J., and McCLAUGHERTY,C. 1990. Predicting long-term patterns of mass loss, notrogen dynamics, and soil organic matter formation from initial fine litter chemistry in temperate forest ecosystems. Canadian Journal of Botany 68:2201-2208.

ALBREKTSSON, A. 1988. Needle litterfall in stands of *Pinus sylvestris* L. in Sweden, in relation to site quality, stand age, and latitude. Scandinavian Journal of Forest Research 3:333-342.

ANDERSON,J. 1973. The breakdown and decomposition of sweet chestnut (*Castanea sativa* Mill.) and beech (*Fagus sylvatica* L.) leaf litter in two deciduous woodland soils. I. Breakdown, leaching and decomposition. Oecologia 12:251-274.

ANDERSSON,F., AXELSSON, B., LOHM, U., PERTTU, K., and ÅGREN, G. 1980. Skogen som miljö.Naturvetenskapliga Forskningsrådets årsbok 1979 - 1980. Forska för livet. Naturvetenskapliga forsknings-rådet. Stockholm. pp 142-179. (In Swedish).

ARCHIBALD, F., AND ROY, B. 1992. Production of manganic chelates by laccase from the lignin-degrading fungus *Trametes (Coriolus) versicolor*. Applied and Environmental Microbiology. 58: 1496-1499.

AXELSSON, B., and S. BRÅKENHIELM. 1980. Investigation sites of the Swedish Coniferous Forest Project. Biological and Physiographical features. Ecological Bulletins (Stockholm) 32: 25-64.

BERG,B., and G. EKBOHM. 1991. Litter mass-loss rates and decomposition patterns in some needle and leaf litter types. Long-term decomposition in a Scots pine forest VII. Canadian Journal of Botany 69:1449-1456.

BERG, B. 1981. Litter decomposition studies within SWECON. Data on weight loss and organic chemical composition. Swedish Coniferous Forest project. Internal Report No 104.

BERG, B., and G. EKBOHM. 1993. Decomposing needle litter in lodgepole pine (Pinus contorta) and Scots pine (*Pinus silvestris*) monocultural systems - is there a maximum mass loss? Scandinavian Journal of Forest Research (in press).

BERG, B., K. HANNUS, T. POPOFF and 0. THEANDER. 1982. Changes in organic-chemical components of needle litter during decomposition. Long-term decomposition in a Scots pine forest. I. Canadian Journal of Botany 60:1310-1319.

BERG,B., EKBOHM,G., JOHANSSON, M.-B., MCCLAUGHERTY, C., RUTIGLIANO, F., and VIRZO DE SANTO, A. 199X. Do some forest ecosystems have a maximum limit for litter decomposition? A synthesis of data. Canadian Journal of Botany (submitted).

BERG,B., BERG,M., FLOWER-ELLIS, J.G.K., GALLARDO, A., JOHANSSON,M., LUNDMARK, J.-E. and MADEIRA,M. 1993. Amounts of litterfall in some European Coniferous Forests. In: A. Breymeyer (Ed). Proceedings from Scope Seminar. Geography of Carbon Budget Processes in Terrestrial Ecosystems. Szymbark, Aug. 17-23, 1991. (in press).

BOLLAG, J.-M., MINARD, R.D., and LIU, S.-Y. 1983. Cross-linkage between anilines and phenolic constituents. Environmental Science Technology 17:72-80.

BONNEVIE-SVENDSEN,C., and GJEMS,O. 1957. Amount and chemical composition of the litter from Larch, Beech, Norway spruce and Scots pine stands and its effect on the soil. Meddelelser fra det Norske skogsforsöksvesen 14(48):115-168.

CARPENTER, S.R. 1981. Decay of heterogeneous detritus: A general model. Journal of Theoretical Biology 89:539-547.

CARPENTER, S.R. 1982. Comparisons of equations for decay of laf litter in tree-hole ecosystems. Oikos 39:17-22.

FLOWER-ELLIS, J.G.K. 1985. Litterfall in an age series of Scots pine stands; Summary of results for the period 1973-1983. Swedish University of Agricultural Sciences, Department of Ecology and Environmental Research Report 19:75-94.

FLOWER-ELLIS,J.G.K., and OLSSON,L. 1978. Litterfall in an age series of Scots pine stands and its variation by components during the years 1973 - 1976. Swedish University of Agricultural Sciences, Swedish Coniferous Forest Project, Technical Report 15.

HOWARD, P.J.A., and HOWARD, D.M. 1974. Microbial decomposition of tree and shrub leaf litter. Oikos 25: 311-352.

JENNY, H., GESSELS, S.P., and BINGHAM, F.S. 1949. Comparative studies of decomposition rates of organic matter in temperate and tropical regions. Soil Science. 68:419-432.

JENNY, H. 1980. The Soil Resource. Origin and Behaviour. Springer Verlag, New York.

JOHANSSON, M.-B. 1986. Decomposition rates of Scots pine needle litters in relation to site properties and litter quality. In: M-B. Johansson (Dissertation) Chemical composition and decomposition pattern of leaf litters from forest trees in Sweden with special reference to methodological aspects and site properties. Reports in Forest Ecology and Forest Soils, Swedish University of Agricultural Sciences, Report No 56, 154 pp.

KEYSER, P., T.K. KIRK and J.G. ZEIKUS. 1978. Ligninolytic enzyme of *Phanerochaete chrysosporium*: synthesized in the absence of lignin in response to nitrogen starvation. Journal of Bacteriology 135: 790-797.

KONDO, R., IIMORI, T., IMAMURA, H., and NISHIDA,T. 1990. Polymerization of DHP and depolymerization of DHP-glucoside by lignin oxidizing enzymes. Journal of Biotechnology. 13:181-188.

LIU, S.-Y., FREYER, A.J., MINARD, R.D., and BOLLAG, J.-M. 1985. Enzyme-catalyzed complex-formation of amino esters and phenolic humus constituents. Soil Science Society of America Journal 49:337-342

LOBARZEWSKI, J., TROJANOWSKI,K., and WOJTAS-WASILEWSKA M. 1982. The effects of fungal peroxidases on Na-lignosulfonates. Holzforschung 36: 173-176.

MÄLKÖNEN, E. 1974. Annual primary production and nutrient cycle in some Scots pine stands. Communicationes Instituti Forestalis Fenniae 84.5. 87 pp.

MINDERMAN, G. 1968. Addition, decomposition, and accumulation of organic matter in forests. Journal of Ecology 56:355-362.

NÖMMIK, H., and K. VAHTRAS. 1982. Retention and fixation of ammonium and ammonia in soils. In: Nitrogen in agricultural soils. Edited by F.J. Stevenson. Agronomy monographs, No 22. Agronomy Society of America. Madison, WI. pp 123-171.

OLSON, J.S. 1963. Energy storage and the balance of producers and decomposers in ecological systems. Ecology 44: 322-331.

PASZCYNSKI,A., HUYNH, V.-B., and CRAWFORD,R. 1985. Enzymatic activities of an extracellular, manganese-dependent peroxidase from *Phanerochaete chrysosporium*. FEMS Microbiology Letters 29: 37-41

PAUL, E. 1984. Dynamics of organic matter in soils. Plant and Soil. 76:275-285.

PEREZ, J., and JEFFRIES, T.W.1992. Roles of Manganese and organic acid chelators in regulating lignin degradation and iosynthesis of peroxidases by *Phanerochaete chrysosporium.* Applied and Environmenal Microbiology 58: 2402-2409.

ROMELL, L.G. 1932. Mull and duff as biotic equilibria. Soil Science 34: 161-188.

STAAF, H., and BERG, B. 1977. A structural and chemical description of litter and humus in the mature Scots pine stand at Ivantjärnsheden. Swedish Coniferous Forest Project. Internal Report No 65.

STEVENSON, F.J. 1982. Humus chemistry. Genesis, composition, reactions. John Wiley and Sons, New York.

Table 1. Linear relations for annual amounts of total litterfall and needle litterfall as dependent on stand age in Scots pine monoculture stands at the SWECON site Jädraås. Basic data from Berg et al. (1993) and Flower-Ellis (1985).

Initial stand age		slope ($kg\ ha^{-1}$ / yr)	intercept ($kg\ ha^{-1}$)	r^2	n	p<
18-year-old stand						
	total litter	89.11	-1041.6	0.716	7	0.05
	needle litter	54.83	- 456.7	0.506	7	ns
55-year-old stand						
	total litter	128.78	-5450.0	0.622	10	0.01
	needle litter	79.81	-3273.2	0.517	10	0.05
120-year-old stand						
	total litter	56.22	-5377.6	0.573	10	0.05
	needle litter	75.12	- 832.2	0.358	10	ns
18 plus 55-year old stand						
	total litter	37.39	11.1	0.837	17	0.001
	needle litter	21.99	211.9	0.773	17	0.001
18, 55, and 120-year old stand						
	total litter	4.76	1284.6	0.099	27	ns
	needle litter	1.44	1011.6	0.018	27	ns

Table 2. Simple and multiple linear relations for average annual amounts of needle litterfall
(kg ha^{-1}) as dependent on latitude, stand age, site index and basal area in Scots pine stands in Scandinavia. All available average values for Scots pine were used (data from Albrektsson 1988;Flower-Ellis 1985;Johansson 1986;cf Berg et al. 1993). ns stands for not significant.

	coefficient	intercept	r^2	n	p<
Simple linear relations					
Latitude	-147.16	10526.81	0.48	33	0.001
Site index	93.49	-676.41	0.42	33	0.001
Age	-9.01	2105.20	0.27	33	0.01
Basal area	46.32	281.91	0.26	33	0.01
Multiple linear relations					
Latitude plus	-140.60	9030.30	0.71	33	0.001
basal area	42.42				0.001
Site index plus	75.55	-924.01	0.49	33	0.001
basal area	25.77				0.001
Latitude plus	-122.17	7487.37	0.73	33	0.001
site index plus	25.31				ns
basal area	36.05				0.001

Table 3. Simple and multiple linear relations for annual amounts of total litter-fall (kg ha^{-1}) as dependent on latitude, stand age, site index and basal area in Scots pine stands. All available data (from Flower-Ellis 1985; Johansson 1986; cf Berg et al. 1993); ns stands for not significant.

	coefficient	intercept	r^2	n	p<
Simple linear regressions					
-latitude	-212.51	15387.38	0.39	17	0.01
-site index	153.40	-1154.73	0.50	17	0.01
-age	-19.28	3587.65	0.19	17	ns
-basal area	102.68	132.71	0.52	17	0.001
Multiple linear regressions					
-latitude plus	-152.70		0.71	17	0.01
-basal area	83.39	9924.33			0.01
-site index plus	93.60		0.64	17	0.05
-basal area	66.63	-1224.16			0.05
-latitude plus	-130.86		0.77	17	0.05
-site index plus	70.47				ns
-basal area	59.01	7501.89			0.05

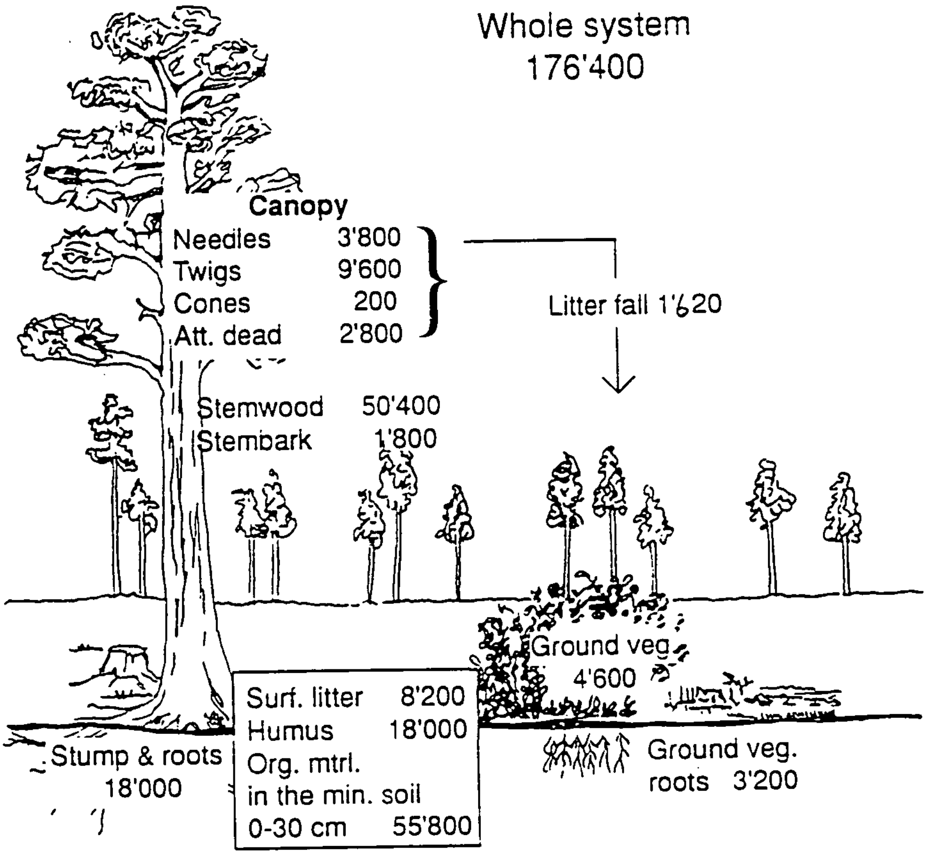


Figure 1. A budget for organic matter in a mature (120-year-old) Scots pine monoculture (SWECON site). Based on data from Andersson et al. (1980) and a litterfall of 1620 kg per ha. Units are in kg of organic matter per ha.

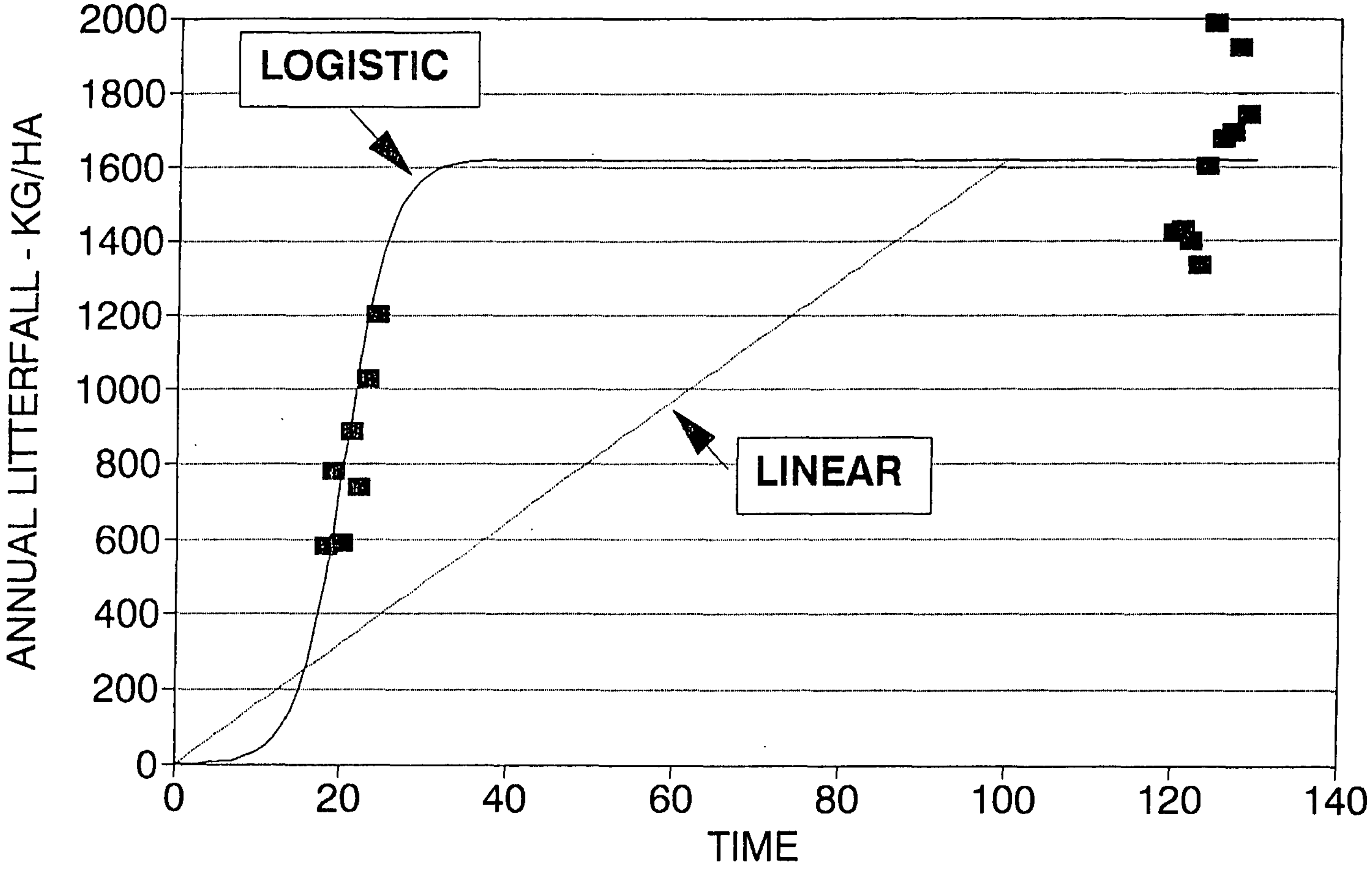


Figure 2. Two simplified models for litterfall with stand age. A. assuming a linear increase in litterfall with age, up to 100 years, followed by a constant litterfall. B. A logistic, nonlinear model fir to the Jädraås data for the 18-25 and the 120-130 yr-old stand.

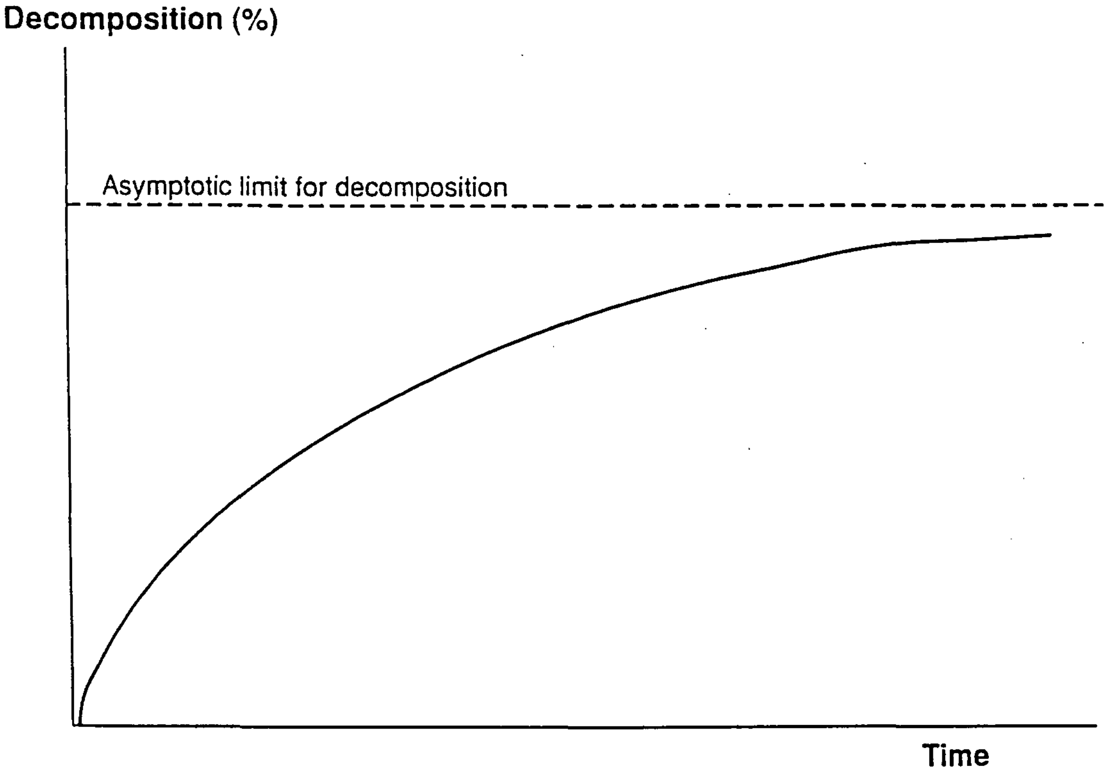


Figure 3. Asymptotic model for mass loss during decomposition. The model, using extrapolation, can be used to estimate a quasi-steady state or limit for accumulated mass loss, indicating the percentage of litter input that is ultimately stabilized. Limit value indicated by a dashed line.

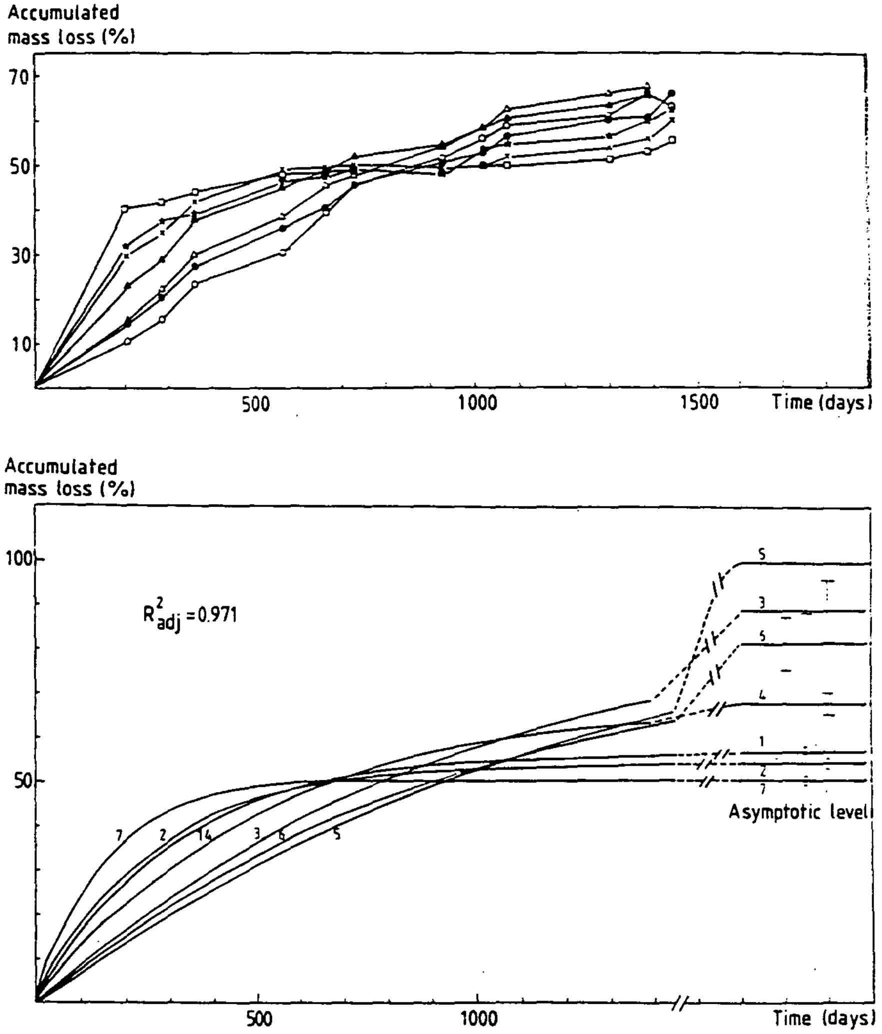


Figure 4. Accumulated mass loss for seven litter types incubated in a 120-year-old Scots pine forest;(x,-1) brown leaves of white birch, (✱,-2) green leaves of white birch, (△,-3) brown Scots pine needle litter, (▲,-4) green Scots pine needles, (●,-5) green lodgepole pine needles, (○,-6) brown lodgepole pine needle litter, (□,-7) grey alder leaves. A. Measured litter mass loss. B. Estimated initial litter mass-loss and asymptotic values for accumulated mass loss. From Berg and Ekbohm (1991).

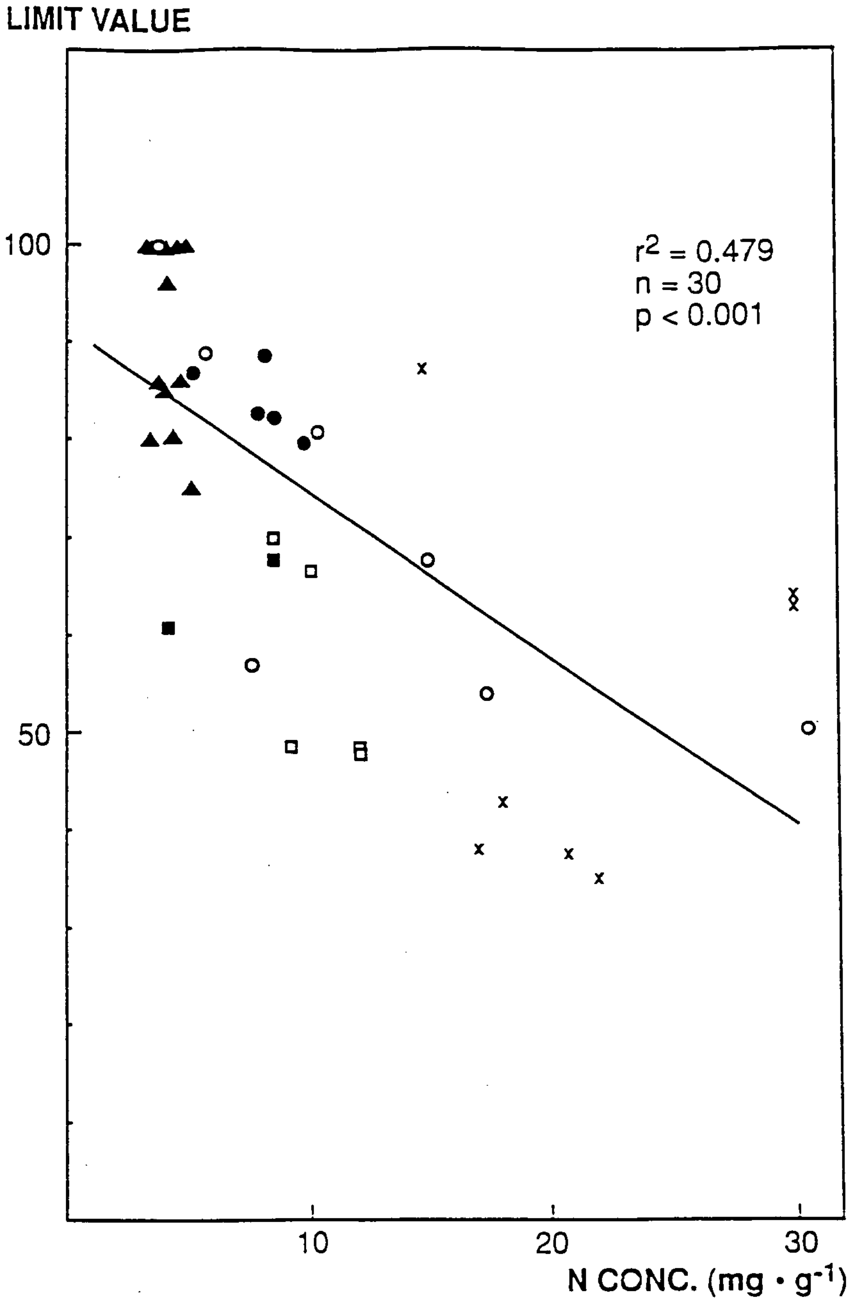
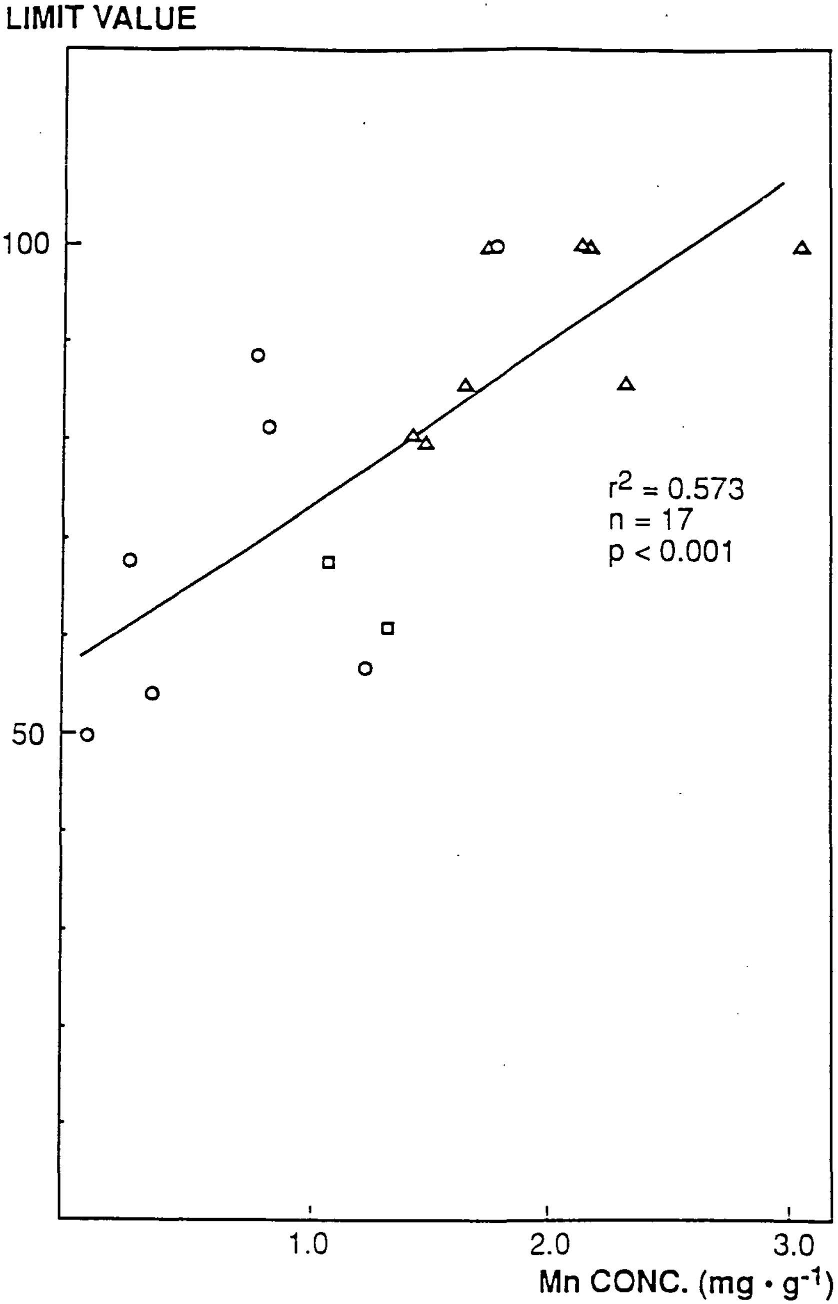


Figure 5. Comparison of limit values and initial concentrations of some nutrients in litter. A. Nitrogen, B. Manganese. Figures from Berg et al. (199X).


LIMIT VALUE
100
50
1.0
2.0
3.0
Mn CONC. (mg • g-1)
r2 = 0.573
n = 17
p < 0.001

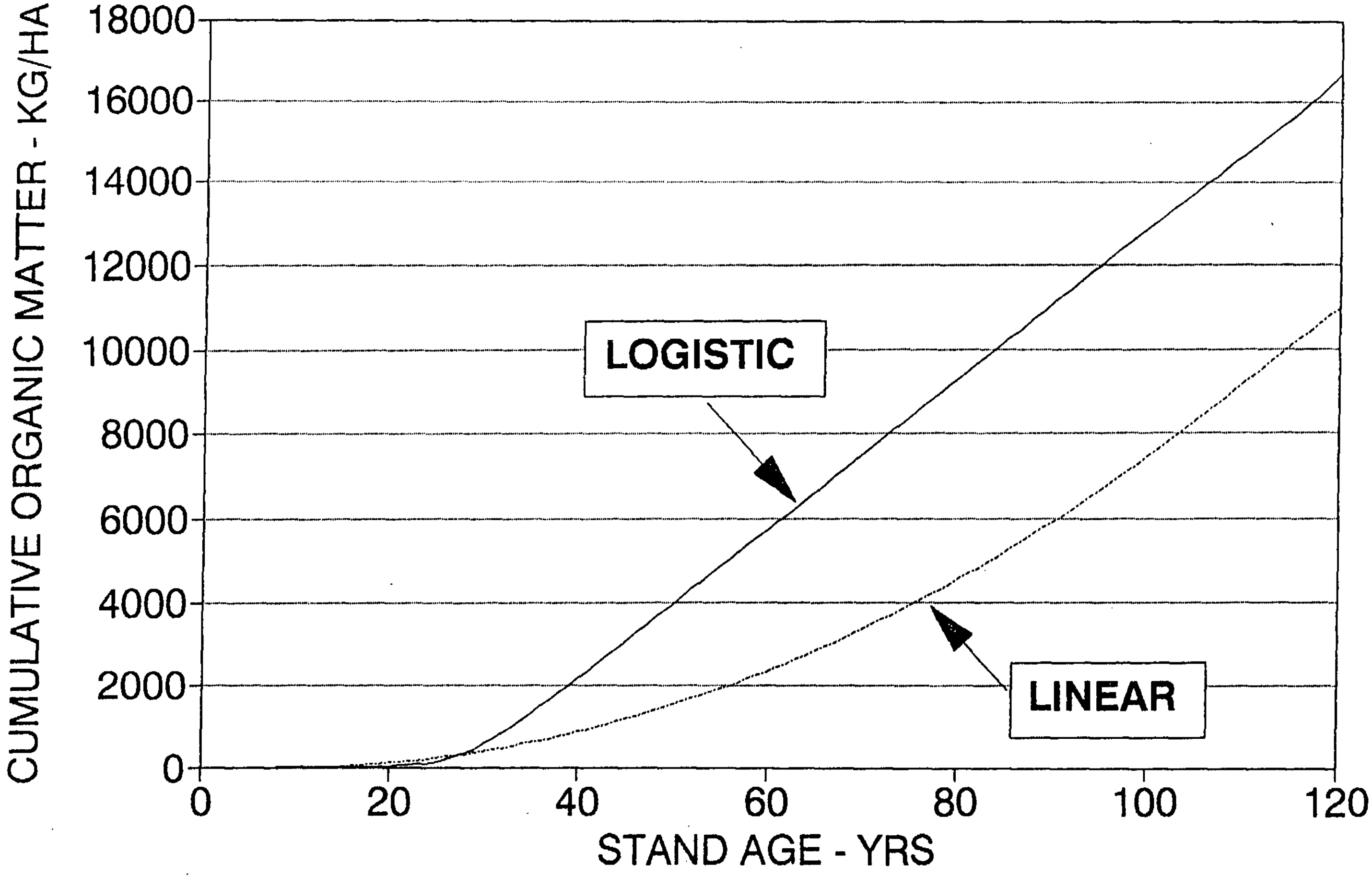


Figure 6. Accumulation of soil organic matter as estimated by a linear model and by a logistic model. In th former case the accumulated amount was 11140 kg ha^{-1} and in the latter one 16670 kg ha^{-1} after a period of 112 years.

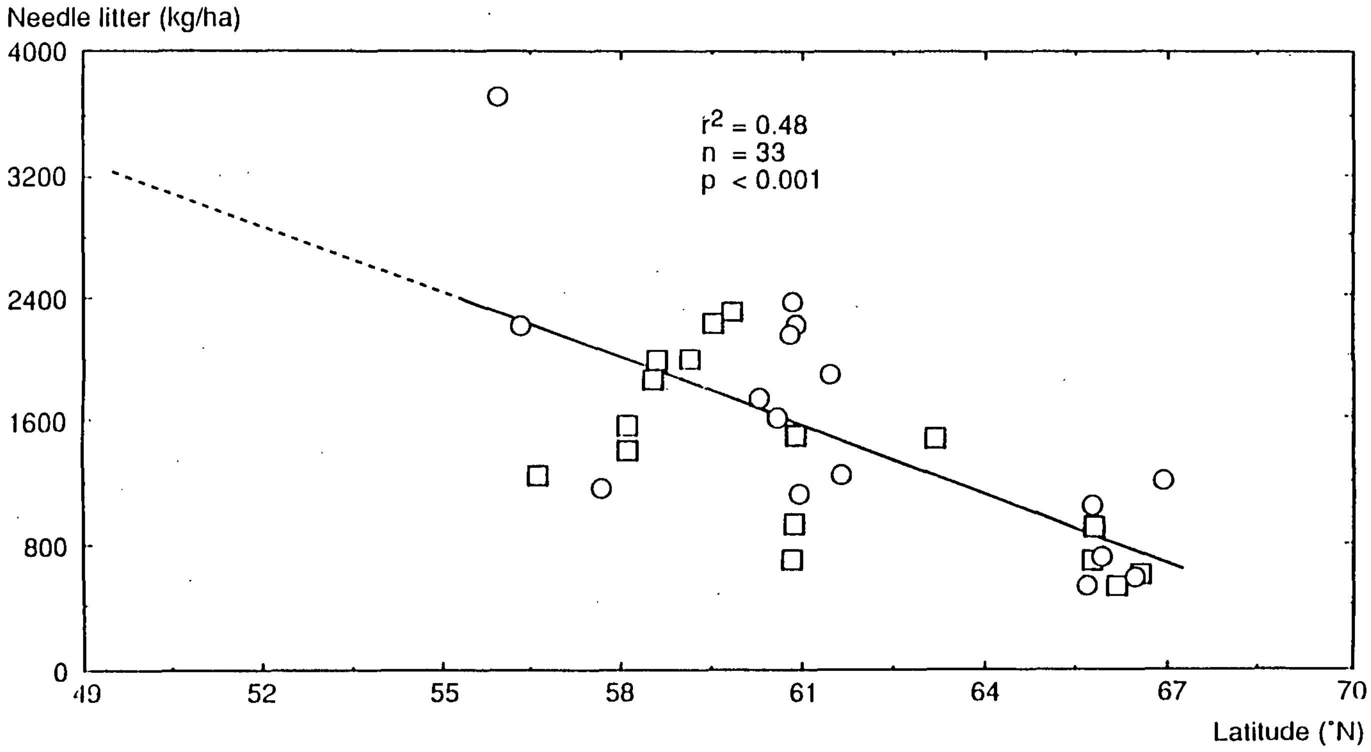


Figure 7. A linear regression giving needle litterfall as compared to latitude. The dashed line gives the extrapolation of data to Central Europe.

Mitt. d. Österr. Bodenkundlichen Gesellschaft, H. 47, S. 43 - 45, 1993

DECOMPOSITION AND SOIL FAUNA

M. M. Coûteaux*

The first general question to be raised can be 'Is it a priority to differentiate what is to be attribute to the soil fauna on the general functioning of the decomposition'. The models (CENTURY, JENKINSON, NCSOIL) on the turnover of the soil organic matter seems to be validated without any identification of the soil fauna contribution. This contribution is integrated in other regulating factors like climate or type of soil. In other models, the turnover of the organic matter is analyzed as the tranfer from one biological compartments to another through the food web but the link with decomposition is not clear.

There are a lot of informations in the literature on the contribution of soil fauna to decomposition, most of them based on micro- or mesocosmes experiments, on physical or chemical exclusion in laboratory and in field concluding to either the stimulation or the inhibition of the fluxes according to the interactions of various basic process.

The priority concerning the fauna contribution to decomposition should focus on informations which can reach to a general concept. That is not easy because of the large range of responses linked to the biodiversity which leads to specific response of two by two relationships (predation, competition, mutualism...) and can be complexified by crossed interactions in the network.

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What is needed to know on order to characterize the role of organisms on a function is: (1) what is it? (identification), (2) where is it? (localisation), (3) how many are they? (density), (4) what are they doing? (activity), (5) how does it interact with other organisms? (interactions).

At all these levels, the gaps in our knowledge are numerous.

1) Identification: Even if there are many works describing soil fauna and their seasonal dynamic, these works concern a group of species which is linked to the skill of the authors. A complete soil fauna description which allow to build up the food web is rarely performed (cf. Amsterdam team) because it is needed to concentrate the activity of different specialists at the same place and time for at least a one-year period.

Nevertheless, this kind of information is needed for all the main ecosystems through the life-zones because the food web structure reflects the biological interactions and is characteristic of the decomposition process.

2) Localisation: At a spatial level, the soil fauna is generally identify with randomized sampling procedure which indicate the level of spatial variability but not their distribution according to spatial distribution of the habitats. This spatialisation is now possible using geostatistic methods. The role of animal by moving and migrating is generally accepted. A quantification of their role in the tranfer of C and nutrient through the profile is needed.

3) Density: This point is often studied with the identification of the faunal communities and their seasonal dynamic. The density of a group and its consumption potential determine its grazing pressure. The grazing pressure is often expressed as the difference between ungrazed and grazed systems or by the ratio between grazers biomass and the grazed biomass. These expressions do not take in account the growth of the grazed biomass which can hid the grazing effect if there is no limitation of nutrient for the grazed populations. It is probably the reason why some relation between grazing pressure and activity are not obvious.

4) Activity: The general assumption is that grazing and comminuting modify fluxes processes. There are many experimental examples of enhancement of carbon and nitrogen mineralization, other examples of reduction such as phosphorus uptake by plant through the grazing of micorhizae. Nevertheless, most results concerns the short term (for instance, the shift in the composition of the microbial communities in the faeces does not last very long). Should we consider that the role of the soil fauna is an addition of numerous induced flushes on the microbial activity and does this addition give a significant effect? Some results show that in the long term, the difference between grazed and ungrazed systems disappear. It can be that the role of the animal is to modify the speed of the processes thus in the late stages where the processes are slow down, their effect is less obvious.

5) Interactions: That is probably the field where the gaps are the most numerous because of the high diversity of possible relationships and our bad knowledge of microbial communities. Some new methods for identifying the microbial diversity should be linked to soil fauna studies to reach a better understanding of their interactions.

Knowing all theses gaps and probably others which are not listed here, priorities are to be defined. Two points have to be considered first: the feasability and the possibility for an information to be included in a model. So in a first step, considering the characterisation of the ecosystems, instead of trying to provide a complete knowledge of the soil fauna composition, a focus should be given on the large groups of the macrofauna. At the spatial level, how they contribute to C and N tranfer through the soil profile?. At a temporal level, what is their role in the long term?

Priority should also be given to studies which use new and acurate techniques (^{13C,} ^{15}N labelling, ^{13}C, ^{15}N natural abundance, new techniques for measuring the microbial diversity...).

Mitt. d. Österr. Bodenkundlichen Gesellschaft, H. 47, S. 47 - 56, 1993

THE C:N RATIO OF THE SOIL MICROBIAL BIOMASS IN SOILS OF DECIDUOUS FORESTS

R. G. Joergensen*

Introduction

The size and composition of the soil microbial biomass is related to the annual C input, to different environmental factors (e.g. climate, relief, water regime) and the parent soil material (e.g. clay content, cation-exchange-capacity, base saturation or pH). Pedogenesis in the humid climate of Central Europe is accompanied by increasing soil acidification (Wolters and Joergensen, 1991). There is evidence that the composition of the decomposer community in soil is partly a function of soil acidity (Schaefer, 1990). It is unknown to what extent the degree of soil acidification affects N storage and the C:N ratio of the soil microflora in forest soils. In agricultural soils, the C:N ratio of microbial biomass is only slightly affected by soil factors (Joergensen *et al.*, 1992). This may be completely different in forest soils where activity and growth of the soil microflora is often limited by nutrient availability, especially N and also P. Changes of soil conditions by acidification alters microbial performance and affects the turnover rate of biomass N and its role in the supply of mineral N to the ecosystem. As a start to increase actual knowledge about these central points, the following simple questions are adressed in this paper:

- What is the size of the microbial biomass N pool in forest soils ?
- What percentage of total N is represented by the microbial biomass ?
- Do soil conditions, especially the pH, influence the C:N ratio of the microbial biomass ?

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Materials and Methods

The 38 forest soils and all analytical methods are described by Wolters anc Joergensen (1991), Joergensen and Wolters (1994) and Joergensen *et al.* (1994) Analysis of soils was carried out using standard methods; for details see Wolters anc Joergensen (1990). Soil microbial biomass C and N were measured by fumigation extraction (Brookes *et al.*, 1985; Vance *et al.*, 1987). Moist soil (50 g, on ar oven-dry basis) was split into two parts, 25 g for the fumigated and 25 g for non-fumigated treatment and extracted with 100 ml 0.5 *M* K_2SO_4. Organic C in the extracts was measured using a Dohrman DC 80 automatic analyzer. Soil microbial biomass C was estimated from the relationship: biomass C = 2.22 D_C (Wu *et al.*, 1990), where D is [(organic C extracted from fumigated soil) minus (organic C extracted from non fumigated soil)]. Total N was measured after oxidation to NO_3 colorimetrically using ar automated system (Skalar). Soil microbial biomass N was estimated from the relationship: biomass N = 2.22 E_N (Jenkinson, 1988), where D_N is [(total N extractec from fumigated soil) minus (total N extracted from non-fumigated soil)]. Ammonium wa measured colorimetrically using an automated system (Skalar). The ammonium flusl after fumigation was calculated from D_{NH4} = [(NH_4 extracted from fumigated soil) minu (NH_4 extracted from non-fumigated soil)]. The results are the means of triplicat analysis and are all expressed on an oven-dry basis (about 24 h at 105°C).

Results

Microbial biomass N

Cation-exchange-capacity (CEC), soil organic C, total N, total P, microbial biomass C biomass N and the biomass C:N ratio were all logarithmically normally distributed i the group of 38 forest soils. In contrast, soil pH, the ratios of soil organic C to total biomass N to total N and the ammonium flush D_{NH4} could not be transformed to normal distribution. Microbial biomass N and biomass C were closely correlate (Pearson, r = 0.88, P < 0.0001, log-transformed data) despite the enormous variation between soils (Fig. 1, non-transformed data). The soil conditions pH, CEC, total N an P had significantly positive relationships with biomass N and the ratio soil organ C:total N was negatively correlated (Spearman rank correlations).

Ratio biomass N:total N

The ratio of microbial biomass N to total N showed a distribution with two peaks centering on 3.5% and 1.7% total N (Fig. 2). The ratios biomass N:total N and biomass C:soil organic C were closely correlated (Spearman, $r = 0.79$; $P < 0.0001$). The 38 soils were divided into two groups (Joergensen *et al.*, 1994), group 1: biomass C:soil organic C > 1.4% ($n = 23$) and group 2: biomass C:soil organic C < 1.2% ($n = 15$). The 38 soils could also be divided into two groups using the ratio of microbial biomass N to total N as parameter, group 1: biomass N:total N > 3.0% ($n = 24$) and group 2: biomass N:total N < 2.8% ($n = 14$). With one exception, the groups were separated by the two ratios in the same way. The soils of group 2 differed markedly from those of group 1 in nearly all parameters presented in Table 1, except total P. The ratios of soil organic C to total N and biomass C:N were higher, whilst the soil pH was lower and the ammonium flush (D_{NH4}) extremely low. The biomass N:total N ratio had the closest correlation to soil pH which itself suggested the formation of two groups (Fig. 3). The lower the pH, the smaller was the biomass N to total N ratio (Table 2). Loose but significant correlations of biomass N:total N existed also with CEC, soil organic C and the soil organic C:total N quotient (Table 2).

Ratio biomass C:N

The biomass C:N ratio was always approximately half of the soil C:total N ratio (Table 1). The biomass C:N ratio has the closest correlation with the soil C:total N ratio, followed by soil C:total P ratio (Table 2). The negative correlation of the biomass C:N ratio with the soil pH was also significant at $P = 0.05$ (Fig. 3). The soil with the lowest pH had also the smallest C:N ratio. On the other hand, the soils with the largest C:N ratio all had a pH below 4. The biomass C:N ratio of the 38 forest soils could be estimated only slightly better by a combination of different linear variables in a multiple regression model than by a single variable. A combination of soil organic C, total N and biomass P could explain 36.3% of the total variance (Table 3).

Discussion

Microbial biomass N and biomass C were as closely correlated in this set of forest soils as reported for agricultural soils (Jenkinson, 1988). Similarly, the biomass N:total N ratio was nearly doubled in comparison to the biomass C:soil organic C ratio, emphasizing the importance of the soil microbial biomass as a reservoir for nutrients. The mean biomass C:N ratio 7.7 of forest soils was markedly higher than the mean values of 5.5 (fumigation incubation, Jenkinson, 1988) and 4.7 (fumigation extraction, Joergensen *et al.*, 1992) found in agricultural soils. In particular to the range of the biomass C:N ratio of 11.9 (5.4 - 17.3) in the 38 forest soils is much wider than that of 5.1 (2.3 - 7.4) previously reported in agricultural soils (Joergensen *et al.*, 1992).

Increasing acidification of forest soils by leaching of calcium caused no systematic change in the biomass C:N ratio. Two other small groups of soils could be separated from the entire group of 38 forest soils (not identical with group 1 or group 2 in Table 2 and Fig. 2). In the soils of the Göttinger Wald, which are mainly developed from the solution residue of the underlying lime stone or from non-hydromorphic loess (Wolters and Joergensen, 1991), the biomass C:N ratio declined with decreasing pH (Fig. 3). The clay minerals of these soils are predominantly illitic (Thöle and Meyer, 1978). However, the role of clay quality has not been proved yet and the relation in Fig. 3 may be coincidental. A pH decrease in these soils was combined with increasing depth and bulk density, which increases the water storage capacity of the profile. A decline in pH could also positively affect the availability of nutrients, such as phosphorus, so that a microbial population is supported with a smaller C:N ratio. Some soils with very low total P content have large biomass P:total P ratios, sometimes reaching more than 40% (Joergensen *et al.*, 1994). In the soils with a biomass P:total P ratio larger than 22%, the biomass C:N ratio shows an extremely steep increase with declining pH (Fig. 3).

In most of the soils, the biomass C:N ratio could not be explained by single factors such as pH, clay quality or the ratios of soil C:total N or biomass P:total P. The development of a soil specific biomass C:N ratio presumably depends on a complex combination of different factors: population structure (age, species), C input (amount and quality), nutrient availability, presence of toxic substances such as heavy metals. An increased biomass C:N ratio with increasing biomass P content (Table 3) and soil C:total N ratio indicates changes in the population structure, e.g. mycorrhiza fungi, in the situation of declining C and N availability. The C:N ratio of fungi can vary widely, from about 4 to 15, that of bacteria, less so, from about 3 to 5 (Paul and Clark, 1989). It is not known whether the biomass C:N ratio shows seasonal fluctuations in forest soils however, it is probable that the biomass C:N ratio is relatively constant and expresses a soil specific population structure developing under definite soil and environmental conditions.

References

Brookes P.C., Landman A., Pruden G. and Jenkinson D.S. (1985) Chloroform fumigation and the release of soil nitrogen: A rapid direct extraction method for measuring microbial biomass nitrogen in soil. *Soil Biology & Biochemistry* **17**, 837-842.

Jenkinson D.S. (1988) The determination of microbial biomass carbon and nitrogen in soil. In *Advances in Nitrogen Cycling in Agricultural Ecosystems* (Wilson, J.R., Ed.), pp. 368-386. CAB International, Wallingford.

Joergensen R.G. and Wolters V. (1994) C and N relationships of the soil microbial biomass in soils of deciduous forests. Biology and fertility of Soils (submitted

Jörgensen R., Meyer B. and Müller T. (1992) Zeitliche Veränderung der mikrobiellen Biomasse in der Ackerkrume einer mitteleuropäischen Lößparabraunerde. *Göttinger Bodenkundliche Berichte* **100**, 1-137.

Joergensen R.G., Kübler H., Meyer B. and Wolters V. (1993) Microbial biomass phosphorus in soils of deciduous forests. biology and fertility of Soils (submitted)

Paul E.A. and Clark F.E. (1989) *Soil Microbiology and Biochemistry.* Academic Press, San Diego.

Schaefer M. (1990) The soil fauna of a beech forest on limestone: Trophic structure and energy budget. *Oecologia* **82**, 128-136.

Thöle R. and Meyer B. (1978) Bodengenetische und ökologische Analyse eines Representativ-Areals der Göttinger Muschelkalk-Scholle als landschaftsökologische Planungsgrundlage. *Göttinger Bodenkundliche Berichte* **59**, 1-230.

Vance E.D., Brookes P.C and Jenkinson D.S. (1987) An extraction method for measuring soil microbial C. *Soil Biology & Biochemistry* **19**, 703-708.

Wolters V. and Joergensen R.G. (1991) Microbial carbon turnover in beech forest soils at different stages of acidification. *Soil Biology & Biochemistry* **23**, 897-902.

Wu J., Joergensen R.G., Pommerening B., Chaussod R. and Brookes P.C. (1990) Measurement of soil microbial biomass C - an automated procedure. *Soil Biology & Biochemistry* **22**, 1167-1169.

Table 1: Mean (geometric), minimum and maximum of soil and microbial parameters in the A horizon (0-10 cm) of 38 deciduous forest soils; group 1: biomass C:soil C > 1.4%; group 2: biomass C:soil C < 1.2%

				Group 1	Group 2
	Mean	Minimum	Maximum	Mean	Mean
pH-H_2O	5.5	3.5	8.3	6.7	4.1
CEC[μeq g^{-1}]	195	63	696	272	117
Soil C [mg g^{-1}]	63.0	18.2	180.5	52.7	85.3
Total N [mg g^{-1}]	4.0	1.0	10.9	3.7	4.5
Total P [mg g^{-1}]	0.46	0.18	1.77	0.47	0.45
Soil C:total N	15.7	12.0	26.6	13.9	18.9
Soil C:total P	133	34	423	110	179
DNH4	15	0	44	17	2
Biomass C [μg g^{-1}]	830	317	2116	958	668
Biomass N [μg g^{-1}]	108	30	347	140	72
Biomass C:N	7.7	4.5	17.3	6.8	95.3
Biomass C:soil[%]	1.3	0.5	2.3	1.9	0.8
Biomass N:total[%]	2.7	0.6	5.4	3.8	1.6

Table 2: Spearman rank correlation coefficients between microbial biomass and soil parameters

	pH	CEC	Soil C	Total N	Total P	Soil C: total N	Soil C: total P
Biomass C	0.48[b]	0.73[d]	0.49[b]	-0.02	0.49[b]	-0.47[b]	0.08
Biomass N	0.63[c]	0.79[d]	0.24	0.56[c]	0.53[b]	-0.62[c]	-0.15
Biomass C:N	-0.33[a]	-0.34[a]	0.25	0.01	-0.23	0.44[b]	-0.41[a]
Biomass C:soil C	0.73[d]	0.55[c]	-0.37[a]	-0.30	0.32	-0.70[d]	-0.53[b]
Biomass N:total N	0.70[d]	0.45[b]	-0.41[b]	-0.20	0.03	-0.47[b]	-0.41[a]

[a] P < = 0.05; [b] P < = 0.01; [c] P < = 0.001; [d] P < = 0.0001.

Table 3: Multiple linear correlation of the biomass C:N-ratio with different variables

	Coefficients of variances
Constant	-0.70
Total N	-0.63[b]
Soil organic C	0.62[b]
Biomass P	0.26[a]
r^2	36.3

[a] P < = 0.05; [b] P < = 0.01.

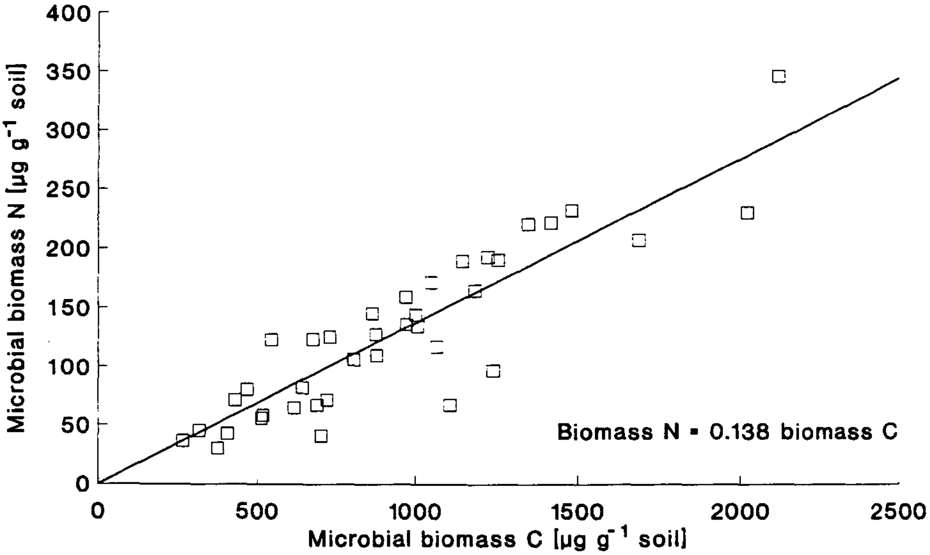
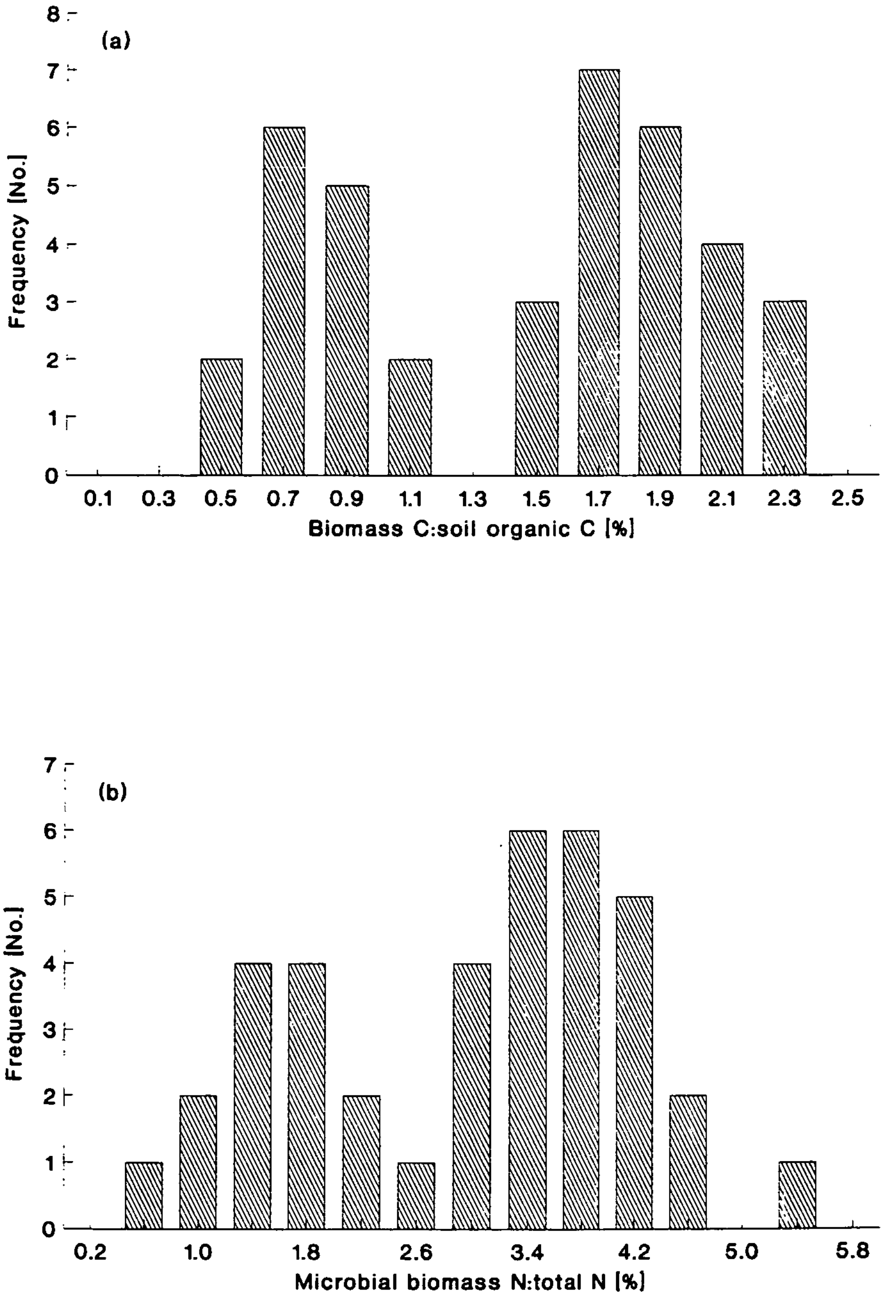


Fig. 1: Linear relationship between biomass C and biomass N



ig. 2: Frequency distribution of the ratios (a) biomass C:soil organic C and (b) biomass N:total N in the 38 forests soils (0-10 cm)

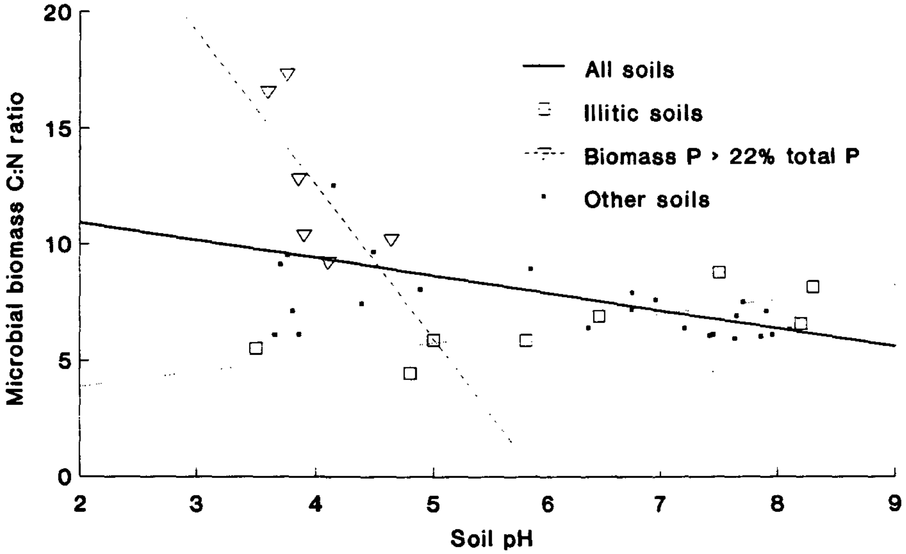


Fig. 3: Linear relationships of biomass N and pH: (1) all soils, (2) illitic soils and (3) soils with a biomass P:total P ratio > 22%

Mitt. d. Österr. Bodenkundlichen Gesellschaft, H. 47, S. 57 - 60, 1993

DECOMPOSITION OF ORGANIC MATTER IN TERRESTRIAL ECOSYSTEMS. MICROBIAL COMMUNITIES IN SOIL.

A.Kjøller, S. Struwe*

AIM

This proposal emphasizes the need of a research-programme on the microbial processes during decomposition of organic matter in terrestrial ecosystems. It is of importance to investigate microbial populations and variations in the decomposition patterns influenced by natural environmental events and by perturbations introduced by man. Elevated greenhouse gas emission leading to increased atmospheric concentrations (global warming), acid rain (acidification), xenobiotics (pesticides etc.) are some of the "stress factors" influencing the microbial processes in the soil.

BACKGROUND

Background for investigation of microbial processes in terrestrial ecosystems: During the last twenty years appropriate methods have been innovated in microbial ecology, especially for use in aquatic (incl. sediment) environments. Soil is a much more complex substrate and the development of methods has advanced less. Therefor knowledge about the microbial processes in soil is more limited and investigations with modern techniques are needed.

The principle of microbial infallibility implies that all biological compounds are decomposed by microorganisms. The different organic components are consequently decomposed but not necessarily with the same rate. The actual decomposer equilibrium depends on the active decomposer organisms and their abundance.

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In some ecosystems and at specific seasons organic matter is accumulated while other systems are decomposing and mineralizing. This is of particular interest in agro-systems where nutrient availability is critical in the growing season, but also in natural systems.The two major microbial populations in the decomposer system are the bacteria and the microfungi, bacteria playing a secondary role despite their high numbers; microfungi are the most active decomposers of plant material. Bacteria have special functions in the mineralization of nitrogen compounds.

Previous studies have focused on isolation and identification of soil microorganisms but not often on the combination of functional and taxonomic studies which are needed for a full understanding of the decomposer community.

Biodiversity during decomposition of organic matter and the role of especially microfungi in nature is only partly known.

NEW CONCEPTS

Quantification of soil and litter fungi have been carried out for many years using very time-consuming techniques. Recent biomass methods are based on substrate - induced respiration and selective inhibition of bacteria and fungi, respectively.

Many factors influence the rate of decomposition and thereby variation in the microbial biomass, e.g. moisture, temperature, pH, substrate quality. The effect of some of these environmental factors on bacteria and microfungi is probably not identical. Studies of the relative activity of these main groups of microorganisms and an analysis of the community variation are important for predicting the ecosystem function.

There are two different reasons for quantifying fungal decomposer populations; one is to assess the decomposition of the organic matter in order to determine the mineralization rate, the other is to obtain a description of the pool of immobilized nutrients stored for the life-time of the decomposer fungi. The entire decomposer chain therefor serves as a link between nutrient supply and nutrient demand.

Some of the soil humus fractions are inaccessible to microbial attack while fresh organic matter is readily attacked. The distinction between autochtonous and zymogenous organisms relates to this difference but seems not to be valid for fungi. In stead, the search for fungi with different growth strategies is important in order to define the nutrient regimes where specific species or strains of fungi may prevail.

In future work it will also be important to include molecular techniques. The methods are developed during the last years and much work is still to be done. It will be possible to identify specific groups of bacteria and fungi and to determine their size, position and activity in situ. Ribosomal RNA based studies of natural microbial diversity and ecology, environmental nucleic acid extracts to analyse microbial communities, fatty acid analysis to identify natural isolates are some examples of modern techniques in microbial ecology. These and other molecular methods need to be innovated for use in decomposition studies.

Besides biodiversity and potential role of microorganisms during decomposition in soil, the microbial processes have to be studied more intensively. Both aerobic and anaerobic metabolic products are produced, CO_2 and CH_4, N_2O, NO, respectively. The potential for the emission of these gasses in various terrestrial ecosystems influenced by different environmental events needs to be investigated, e.g. the physiology and ecology of the bacteria involved in methan production and denitrification.

CONCLUSION

It is consequently proposed to formulate a research programme developing lines of microbial research to analyze microbial communities and their function. The results gained from this activity will be helpful in predicting the effect of some environmental events, temperature increase, changes in humidity and oxygen tension, mineralization of organic material and in addition, effects of human activity.

RESEARCH PRIORITIES

- Decomposer patterns on organic matter - variations caused by antropogenic influence.
- Accumulation of organic matter - accumulation and turnover of microorganisms - Mineralization index.
- Microbial abundance and activity. Linkage between presence and function.
  In situ activity.
- Respiration studies, substrate-induced respiration, selective inhibition, key compound response.
- Development of a test compound response method in natural or controlled experiments to determine microbial growth and activity.
- Development of selected strain fate methods in natural or controlled experiments for establishment and activity response.
- Growth strategies of key species under favourable and stressed conditions.

- Sensitivity of greenhouse gas production to environmental conditions. Substrate dependency.
- Molecular methods for detection of activity in natural environments.

Mitt. d. Österr. Bodenkundlichen Gesellschaft, H. 47, S. 61 - 67, 1993

DECOMPOSITION OF ORGANIC RESIDUES IN SOILS : LITTER QUALITY AND SPATIAL DISTRIBUTION OF DECOMPOSITION PRODUCTS AND MICROBIAL COMPONENTS

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Introduction

Reliable descriptions of decomposition processes in simulation models largely depend on the definition of conceptual pools of both residue components and soil organic matter. These fractions are considered to behave in an independent way and are treated as separate entities with different kinetic behaviour upon decomposition. Although the "ideal" conceptual fraction is also one that can experimentally be verified, attempts to do so have had variable success so far. On the one hand, residues can be fractionated into separate pools differing in their degree of extractability with a range of solvents with increasing extracting power. On the other hand, prediction of their decomposition or mineralization seems often confounded by other unidentified quality parameters. In addition, the independence of the established fractions can be questioned as the decomposition rate of so-called resistant or recalcitrant fractions might be influenced by the presence of "active" components.

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Along another line of thinking and perhaps on a longer time perspective, decomposition processes have been linked to the spatial distribution of various organic components amongst soil aggregates. Association of organics with mineral particles has regularly been mentioned as crucial in determining their resistance to decomposition. Unravelling the association between organic components and soil mineral particles has been performed by various physical disruption and fractionation schemes. These have been put forward as an alternative to the more straightforward chemical fractionation and characterization methods thatnevertheless lack sound relationships with soil organic matter dynamics or soil fertility.

In the present paper three lines of research are highlighted. At first, an investigation towards the quality parameters of various residues will be given, together with a more detailed study on the validity of such practice. Secondly, the physical separation of so- called "light" litter fractions will be addressed and illustrated by an example of organic residue decomposition in alley cropping systems. Thirdly, physical fractionation of stable humus components and microbial components into different size classes will be presented. Integration of these three lines should be aimed at to obtain a complete picture of decomposition processes.

Description of residue quality in relation to decomposition/mineralization processes

Organic residue inputs constitute important and in some cases essential components of soil fertility. Proper management of organic residues thus must be based on accurate knowledge of the principles governing the dynamics of the components involved. One of these principles is that decomposition rate or nutrient release rate is different for various plant species depending on a multitude of quality parameters. In a first approach, quality of the organic matter is assessed by determining the distribution of dry matter over lignin, cellulose, hemicellulose and soluble components (Van Soest and Wine, 1968). Lignin represents a recalcitrant fraction, (hemi)cellulose a resistant one, and the soluble components are typified as "active" or labile.

The validity of this fractionation concept can elegantly be demonstrated by using homogeneously labelled residues. We used ^{14}C-labelled maize residues (Merckx e al., 1986) and characterized them according to Van Soest and Wine (1968). The characteristics of the material and the results of the fractionation are given in Table 1

In the subsequent incubation experiment two treatments were envisaged. Either the complete, unmodified residue (MAI treatment) was added to 50 g samples of preincubated soil or only the resistant and recalcitrant material (NDF treatment). As such, in the MAI treatment 209 mg of dried material were added, whereas in the NDF treatment only 102 mg were added. The rationale behind this experiment was that such procedure should allow to study the decomposition of the resistant and recalcitrant fractions with or without the presence of the so called "active" fraction, all other parameters kept equal. At specified time intervals during the incubation, respired CO_2 and ^{14}C-CO_2, microbial biomass C and ^{14}C, mineral N and residual ^{14}C were determined.

Table 1. Selected characteristics of homogeneously labelled maize leaves.

Fraction	size %	%C	%N	C/N	Spec.Act. Bq/ mg C
Soluble (Active)	48.0	37.8	1.4	27.0	4977
(Hemi)cellulose (Resistant)	46.7	40.0	0.6	66.7	4636
Lignin (Recalcitrant)	1.0	57.5	nd	nd	4556
Ash	4.3	na	na	na	na
Maize leaves	100	37.4	0.9	41.6	4800

na...not applicable, nd.... not determined

The results of the respired carbon are given in Table 2. From the evolution in specific activity it can be deduced that in the MAI treatment most of the CO_2 in the initial phases is derived from the residue. Decomposition of the NDF fraction clearly is delayed, until at day 9 a specific activity approaching that of the added material is reached. Taking for granted the original hypothesis that fractions behave independently, the subtraction of the decomposition curves of the NDF from those of the MAI should yield the decomposition curve of the active fraction. Doing so, however, we arrive at the impossible situation of a decreasing ^{14}C-CO_2 evolution at day 9 in a curve which by definition should be cumulative. It seems therefore that the decomposition of the NDF materials is enhanced in the presence of the active fraction. In fact, the decomposition of the NDF materials takes an earlier start in the presence of active components, possibly due to a real priming effect.

Apart from these, perhaps academic questions, relating to the independent behaviour of residue fractions, another implication can be inferred from similar laboratory incubations with a range of (tropical) plant species of varying age. An example is to be inferred from Table 3 where combinations of two different incubation methods (with or without preincubation of the soil) and four different species are listed. For three species we used three different batches of a different "age", that is the time since the previous pruning. As the variable for decomposition the net amount of CO_2 evolved after 20 weeks of incubation at 25°C is chosen. To explain the variance in the measured CO_2 evolution we tested the C/N ratio, the lignin-to-N ratio, the (lignin + polyphenol)-to-N ratio, the "active" fraction after a Van Soest fractionation and the water soluble fraction.

Table 2. Activity and specific activity of released CO_2 in an incubation experiment with homogeneously labelled maize leaves or with their resistant and recalcitrant fractions (NDF).

Day	Cumulative Activity of evolved CO_2 (kBq/kg dry soil)			Specific activity (Bq/mg C)	
	MAI	NDF	MAI - NDF	MAI	NDF
0	0	0	0	na	na
2	1905	4	1901	4443	571
4	2698	123	2575	4200	2593
9	3476	1401	2075	3721	4137
16	3989	1729	2260	4385	3872
23	4301	1828	2473	3207	2519
40	4601	1915	2686	2119	1334

na = not applicable

Regression analyses between the amount of CO_2 produced and various quality factor showed that the best correlation was obtained with the L/N ratio ($r^2 = 0.78$) and wit the L+PP/N ratio (r^2 = 0.78). More conventional parameters such as the C/N ratio yielded a r^2 of only 0.63. With respect to the Van Soest fractions, an even poorer correlation with the fraction soluble in neutral detergent was found (r^2 = 0.57). Water extracts of the residues were also used to yield a labile fraction with which CO_2 evolution correlated with a r^2 of 0.74. These results show that large differences exist in the suitability of quality factors in predicting decomposition and mineralization.

Table 3. Quality parameters from a selection of tropical plants and CO_2 production after 20 weeks of aerobic incubation.

Code[a]	CO_2[b]	%C	%N	%L	%PP	C/N	L/N[c]	L+PP/N[d]	WSF[e]	ACT[f]
	(mg/kg soil)								%	%
LR	760	42.4	4.33	8.1	3.37	9.8	1.87	2.65	35.0	66.0
SR	654	42.7	2.79	10.4	1.41	15.3	3.73	4.23	28.0	50.9
FR	492	40.1	3.26	17.2	2.39	12.3	5.28	6.01	20.0	49.0
DR	282	39.5	1.41	14.9	3.43	28.0	10.6	13.0	18.0	52.6
LP	636	42.4	4.33	8.1	3.37	9.8	1.87	2.65	35.0	66.0
SP	570	42.7	2.79	10.4	1.41	15.3	3.73	4.23	28.0	50.9
FP	376	40.1	3.26	17.2	2.39	12.3	5.28	6.01	20.0	49.0
DP	139	39.5	1.41	14.9	3.43	28.0	10.6	13.0	18.0	52.6
LJP	871	41.9	5.50	7.8	2.60	7.6	1.42	1.89	46.0	75.2
LOP	640	41.5	3.88	11.3	1.66	10.7	2.91	3.34	31.0	65.1
SJP	855	45.5	3.75	8.9	2.28	12.1	2.37	2.98	28.0	69.4
SOP	667	41.0	2.31	14.3	0.98	17.7	6.19	6.61	28.0	55.7
DJP	334	43.9	3.16	16.8	2.12	13.9	5.32	5.99	16.0	52.8
DOP	163	44.2	1.50	18.4	3.23	29.5	12.3	14.4	16.0	49.1

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[a] L=*Leucaena leucocephala*, S=*Senna siamea*, F=*Flemingia macrophylla*, D=*Dactyladenia barteri*; R=Rewetted soil, P=Rewetted and preincubated for 7 days at 25°C, O="Old", J="Young"

[b] Net (Amended soils - control soils) amount of CO_2 recovered after 20 weeks incubation.

[c] L/N = lignin-to-N ratio; [d] L+PP/N = (lignin + polyphenol)-to-N ratio; [e] WSF = water soluble fraction

[f] ACT = "active" fraction, soluble by neutral detergent

Physical separations of soil organic matter : light fractions, soil microbial biomass and microbial products

At the outset of this paragraph it needs to be repeated that the purpose of separating soil organic matter into discrete pools is to obtain fractions that have a biological meaning, in this that the presence or size of a given fraction should have an immediate bearing to direct or indirect soil fertility parameters. In low input farming systems, a strong dependency exists on the presence of litter or "macroorganic" matter as an immediate source of plant nutrients. Within the Tropical Soil Biology and Fertility Programme (1993) therefore, much attention is directed to the separation of this pool. The fraction considered has a low density and by definition floats in liquids with a density between 1.6-2.0 g.cm^{-3}. For convenience, however, water is often used instead. Within a collaborative project with the I.I.T.A. we followed the dynamics of this floating fraction in a microplot experiment amended with residues of *Leucaena leucocephala* and compared the changes in different size fractions of macroorganic matter and in microbial biomass contents. In general, all fractions (> 2 mm, 2-0.25 mm and 0.25 - 0.15 mm) and the microbial biomass responded significantly to the residue application for the top 0-5 cm. In addition, microbial biomass carbon correlated significantly ($r^2 = 0.80$) with the sum of all fractions between 2 and 0.15 mm). At the same time, CO_2 production correlated well with the total C, a correlation entirely to be ascribed to the proportion taken by the "light" fraction. The results confirm the relevancy of both the floating fraction and the microbial biomass as far as their reaction to substrate input is concerned.

With respect to the above fractionation scheme, however relevant to low input soil fertility, we must acknowledge that these fractions seem to refer especially to short term processes such as decomposition in tropical scenarios. Predictions on a longer term must rely on further separation schemes, fractionating soils completely (not only the part which can be flotated in a water jet) into different particle size classes. In the experiment, serving as an example, soils were incubated after amendment with ^{14}C-^{15}N- labelled residues for specified periods, either in the field or in the lab and then separated into particle size fractions (> 250 µm, 250 - 50 µm, 50 - 20 µm, 20 - 2 µm 2 - 0.1 µm and < 0.1 µm) by sieving, settling or centrifugation (Jocteur Monrozier et al., 1991). Total C and N, ^{14}C and ^{15}N can then be determined in these fractions and the microbial components identified. The latter has only been made possible recently by fumigation extraction methodologies, allowing to assess microbial components even after dispersion of the soil in excess water.

Referring to total C, N and microbial biomass, these three have similar distribution patterns, be it after 7 weeks or 6 years of incubation. In both cases the largest proportion of biomass C, organic C and N was found in the 20 - 2 μm class, reported to also have the largest proportion of microaggregates.

The distribution of the labelled or plant derived C and N followed a similar pattern, the more so when time progressed. Apart from an initial larger percentage of plant-derived C in the >250 μm fraction, due to the presence of coarse residue fragments, the labile components leach out from the residue quite easily and are metabolized in sites where microbial cells are distributed. Therefore it seems only logical that newly-formed cells, their metabolic products and remnants are distributed in a similar fashion as the native soil microbial biomass.

Although both illustrated fractionation experiments have been carried out with different soils and different objectives in mind, there seems to be an obvious link between them and also a connection with the residue quality descriptions. However, they also represent the typical state of the art which is one of emphasizing only parts of the entire decomposition process rather than presenting a holistic view taking into account short term perspectives (such as nutrient release in a "fast" tropical environment) as well as long term perspectives such as the accumulation and stabilization soil organic matter.

References

Jocteur Monrozier L, Ladd J N, Fitzpatrick R W, Foster R C and Raupach M (1991) Componennts and microbial biomass content of size fractions in soils of contrasting aggregation. Geoderma 49, 37-62

Kachaka S., B. Vanlauwe and R. Merckx (1993) Decomposition and N-mineralization of prunings of different quality. In : Soil Organic Matter Dynamics and Sustainability of Tropical Agriculture. Eds. K. Mulongoy and R. Merckx, John Wiley & Sons, pp 199-208, pp 66-68.

Merckx R, van Ginkel J H, Sinnaeve J and Cremers A (1986) Plant-induced changes in the rhizosphere of maize and wheat. Part 1. Production and turnover of root-derived materials. Plant and Soil 96, 85-93.

Tropical Soil Biology and Fertility - A Handbook of Methods (1993) 2d edition, Ed. J M Anderson and J S I Ingram, C.A.B. International.

Van Soest P J and Wine R H (1968) Determination of lignin and cellulose in acid-detergent fibre with permanganate. Journal of the Association of Agricultural Chemists 51, 780-785.

Mitt. d. Österr. Bodenkundlichen Gesellschaft, H. 47, S. 69 - 70, 1993

CALCULATING THE AMOUNT OF CARBON RETURNED TO THE SOIL EACH YEAR FROM MEASUREMENT OF SOIL ORGANIC MATTER

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The current Rothamsted Carbon Turnover Model (Jenkinson, et al., 1987) partitions the soil carbon into five discrete pools: decomposable plant material, resistant plant material, biomass, humus and inert organic matter. First order equations describe the transfer of carbon between these pools. The rates constants are modified by the temperature,moisture and CEC of the soil. This simple model has been validated against total and biomass carbon contents of soil from the Rothamsted long-term field experiments; it also gives a good account of how the pulse of 14C produced by the thermonuclear tests of the early 1960's has moved through the soil organic carbon in these experiments.

Carbon models can be used in forward or inverse simulations. A forward simulation predicts the amount of organic matter in the soil from a knowledge of organic matter inputs and rates of decay. For example, theeffects of long-term straw incorporation have been predicted by assuming a straw input of 5t/ha/year (Ocio et al., 1991). An inverse simulation works backwards from the current amount of organic matter in the soil to give an estimate of the amount of carbon that must have been returned to the soil each year to attain this stock. This provides a new way for measuring net primary production (Jenkinson et al., 1992).

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The Rothamsted Carbon Turnover Model is currently being used to address problems associated with carbon dioxide fertilization and climate change. The effects of global warming on the estimated carbon dioxide emissions from the soil were examined by Jenkinson et al., 1991. They estimate that the additional release of carbon dioxide from soil organic matter over the next 60 years will be 61 x 10^{15} gC. This is approximately 19% of the carbon dioxide that will be released by combustion of fossil fuel during the next 60 years, if present use of fuel continues unabated.

Jenkinson,D.S., Hart,P.B.S., Rayner,J.H., Parry,L.C., 1987. Modelling the turnover of organic matter in long-term experiments at Rothamsted. INTECOL Bulletin. 15:1-8.

Jenkinson,D.S., Adams,D.E., Wild,A., 1991. Model estimates of carbon dioxide emmissions from soil in response to global warming. Nature. 351: 304-306.

Jenkinson,D.S., Harkness,D.D., Vance,E.D., Adams,D.E., Harrison,A.F., 1992. Calculating the net primary production and annual input of organic matter to soil from the amount and radiocarbon content of soil organic matter. Soil Biol.Biochem. 24:295-308.

Ocio,J.A., Brookes,P.C., Jenkinson,D.S., 1991. Field incorporation of straw and its effects on soil microbial biomass and soil inorganic-N. Soil Biol.Biochem. 23:171-176.

Mitt. d. Österr. Bodenkundlichen Gesellschaft, H. 47, S. 71 - 78, 1993

EVALUATION OF C:N RATIO AS A PARAMETER OF N MINERALIZATION

R. Vallejo*

Litter C:N ratio

N content of plant tissues depends on:

1) The physiological role of plant organs, with N accumulation in the more active tissues: leaves vs lignified shoots or roots. In the same way, the age of plant tissues also affects N content, with richer young tissues and strong impoverishment in leaves before abcission in many cases.

2) Plant species, in relation to different ecological strategies to cope with plant resource limitations (thick cuticules to dryness, N fixation) or adaptations like anti-herbivory. Soil N availability broadly affects competitivity of different plant species; as a general rule, with the exception of N_2-fixing species, trees growing in low N sites produce litter with relatively high C:N ratio (Vitousek et al., 1982).

3) The N content of a single species is also sensitive to soil N availability. For example, in mature needles of <u>Pinus halepensis</u> from different locations, % N ranged between 0.84 and 1.74, and C:N ratios between 31.4 and 59.4 (Serrano, 1993 pers. comm.). The accumulation of internal plant reserves can vary depending on N availability: phenolics vs. alkaloids as protection against herbivory or sugars vs. amino acids as osmoticants (Bloom et al., 1985). C:N ratio of microorganisms can also fluctuate with growth rate and nutrient availability (McGill et al., 1981).

C:N ratio has been widely used as an expression of resource quality for decomposition. This use assumes that N concentration in most of the cases is roughly related to other litter components which can have a relevant role in defining litter quality, e.g. lignins and other polyphenols, water soluble carbohydrates and other nutrients.

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From these assumptions, Duchaufour (1984) classify litter qualities into: a) improving litter producing mull humus form, with C:N < 25; b) acidifying litter with C:N > 50 and c) intermediate litter with C:N 30-50, giving way to different humus forms depending on environmental factors.

Initial litter N content has been found in many works to be positively correlated with litter mass loss rates (Fogel & Cromack, 1977; Aber & Melillo, 1982; White et al., 1988). However, some results show that this index does not always explain differences in decompostion rates. For example, Berg & Ekbohm (1991) incubated different litter types in Sweden and found that at initial stages of litter decomposition mass loss was positively related to the initial N content, but after 3 years of incubation, the situation was reversed and finally the poorer substrate qualities decomposed faster.

More generally accepted is that the initial litter C:N ratio affects the N dynamics along decomposition. For example, Vigil & Kissel (1991) explained N mineralization from crop residues by the C:N ratio, either from field and laboratory incubations. N mineralization from soil particle fractions was found inversely correlated to the C:N ratio in many works (see in Christiensen, 1992). Following the interpretations in Smith (1982), C availability primarily controls microorganisms growth while the combination of C and N controls the balance of N immobilization and mineralization; however, microbial growth can be limited by N (or P) in very poor substrates. For McGill et al. (1981), N dynamics operates through the C:N ratio of the microorganisms, which fluctuates between limits depending on the initial substrate C:N ratio.

C:N ratio dynamics in the decomposition process

Organic carbon and organic nitrogen are stabilized together and are mineralized by microorganisms in a process driven by the search of energy (McGill & Cole, 1981). The litter C:N ratio decreases along the decomposition process owing to a greater release of C than N. The substrate C:N ratio approaches monotonically to an asymptotic value depending on the initial properties of the substrate, e.g. N content and substrate quality, and on the microbial properties (Bosatta & Agren, 1985).

Relative N accumulation can be due either by microbial immobilization or by incorporation into humic substances. Lignin level in litter can strongly influence the N immobilization capacity (Aber & Melillo, 1982).

Following the theoretical analysis from Agren & Bosatta (1987), the dynamics of substrate decomposition can be defined in terms of change of quality which is interchangeable with time units; from the value of substrate quality at a given stage of decomposition it is possible to calculate the C fraction remaining from the initial input of fresh litter. From this approach, the question is to what extent the C:N ratio can be used as a organic matter quality descriptor from fresh litter to humic substances.

C:N distribution in the soil profile

There is a general trend of narrowing C:N ratio as the fractions particle size decreases in the forest floor layers (Ohta & Kumada, 1977). This pattern is reflected in the overall C:N of the forest floor layers, which decreases the average separates size and the C:N ratio from the L layer to the A horizon. This pattern of C:N ratio distribution in the soil profile is consistent in soils with negligible organic matter leaching rates and is suggested by Kumada (1987) as the most effective index to determine the degradation of litter.

From the analysis of a set of forest soil profiles in mediterranean conditions (Vallejo, 1983; Vallejo & Hereter, 1986), the C:N distribution in the forest floor-soil profile shows a characteristic shape but the parameters of the curve depend on litter quality, climatic conditions and soil type. For each set of these conditions, the single soil profiles differ among them in small changes in litter inputs, microclimatic conditions or soil properties which affect more or less the rates of litter decomposition but following the same process of litter transformations; therefore, each profile sample can represent one step of litter decomposition, expressed by its quality (or C:N ratio), and a set large enough of profiles fits to a single curve that reproduces the time sequence of transformations for litter.

These curves show three relevant features: 1) In the early stages of litter decomposition, L layer, a rapid decrease of the C:N rate is produced corresponding to net N immobilization (Vallejo Rosich, 1990). 2) In the F, H and A layers, the C:N ratio remains approximately constant in spite of changes in particle size distribution, faunal activity and mineral particles mixing (OC content decreasing from 40 % to 7 %); the maintenance of constant C:N ratio means that C and N are released at the same rates; in the upper mineral A horizon, there is a mixing of organic matter of similar C:N ratio thas in the F and H layers; the range of C:N ratios found for these layers is 17 to 25. This constancy of C:N ratio has also been observed by Ohta & Kumada (1977) through most of the size fractions in the F and H layers, with C:N 25-30. Maximum levels of N release have been postulated for these layers. The turning point between net N immobilization and mineralization has been defined both empirically and theoretically at critical C:N ratios which have been postulated to depend on initial substrate C:N ratio and decomposition rates (Bosatta & Staaf, 1982). 3) In the lower soil layers, C:N ratio changes abruptly to lower values which asymptotically decrease to steady values downwards in the soil profile; these changes are parallel to a sharp increase in the humic substances content (Hereter & Vallejo, 1987) ; therefore these organic compounds accumulate N in humic substances. In the mineral soil profile, organic carbon and nitrogen are linearly correlated depending on soil type, no matter the initial litter quality (Vallejo, 1983).

A similar sequence of C:N ratio changes in the soil profile has been described by Ohta & Kumada (1977, 1978a, 1978b). For these latter soils, net N mineralization showed a minimum with very high and very low C:N ratios, and maximum values with intermediate ratios, e.g. C:N = 40 to 20 in the F layer, along the forest floor - A horizon profile. These observations agree with the general trends found in the literature of decomposing forest floor litter (Berg & Staaf, 1981), with the explanations on N immobilization/mineralization dynamics from microbiological models (Smith, 1982) and with the simulations produced by a semi-empirical model (Bosatta & Staaf, 1982) based upon the initial N content in litter and its decomposition rate.

Limitations

One advantage of C:N ratio is that being a routine analysis, it is plentiful available. One limitation is that interpretations assume that the index actually reflects the ratio organic carbon:organic nitrogen. Most of total N analysis (kjeldahl) account for organic plus ammonium N and in most of the cases it is assumed that ammonium N is insignificant in relation to organic N. However, this situation is not the case in soils or horizons poor in organic matter, having high NH_4 fixation capacity and/or that have been heavily fertilized. This problem is frequent in subsurface horizons of arid regions (Gallardo & Moyano, 1987).

With respect to the fresh substrate quality, the overall C:N ratio of a given litter reflects the combinations of different molecules with different composition. Ideally, similar values of this ratio could correspond to very different combinations of compounds giving different behaviour with respect to decomposition rates and N mineralization. In field studies, litterfall is mostly composed by a relatively reduced number of major organic compounds which reduce the real possible combinations that explain a given C:N ratio. However, no unique interpretation can be drawn from the initial litter C:N ratio with respect to N dynamics; for example, Berg & McClaugherty (1989) showed that during litter decomposition the onset of net N release corresponded to different C:N ratios in different litter types.

If the total soil profile is taken into consideration, with the different stages of soil organic matter decomposition, a unique interpretation of the C:N ratio of a particular organic fraction with respect to decomposability or N dynamics is not possible.

Humic substances are very recalcitrant in spite of its narrow C:N ratio whereas microbial bodies with similar rartios are easy to decompose. This is an extension of the above mentioned basic shortcoming of this parameter. In fact, the parameter that actually will express the quality of a given substrate with respect to decomposition and N dynamics is the ratio between available energy and N (Jansson & Persson, 1982) or the C:N ratio of mineralizable soil organic matter (Hadas et al., 1992). The former authors stress that different types of organisms may have different Energy:N demands and that this characterization at the ecosystem level needs further attention. In conclusion, two main gaps are identified that suggest the need of further research effort: 1) to improve the characterization of litter or soil organic matter energy availability by a handy parameter, instead of total organic carbon, and 2) to stablish the Energy/N demand for decomposer communities characteristics either of different type of ecosystems and of the succession of litter decomposition in a given ecosystem.

Perspectives

Some main environmental issues have been related to changes in the C:N ratio of plants and soil organic matter that may introduce changes in N dynamics. The foreseen atmospheric CO2 increase has been shown to dramatically raise the leaf C:N ratio in chamber culture (Couteaux et al., 1991). In the Mediterranean area, forest wild fires cause net N losses which can affect post-fire plant community succession and its N budget (Ferran & Vallejo, 1992). In industrial areas of Central and Western Europe, atmospheric N load can cause the opposite trends (Van Breemen, 1990). The present perspectives of extensive old or marginal fields afforestation for the next decades, in areas of South Europe, is going to modify soil organic matter and N reserves and dynamics. One way to keep with these issues is a better and more general interpretation of data which are plentiful available, like the C: N ratio of litter and soil organic matter.

References

Agren, G.I. & Bosatta, E., 1987. Theoretical analysis of the long-term dynamics of carbon and nitrogen in soils. Ecology, 68,5, 1181-1189.

Aber, J.D. & Melillo, J.M., 1982. Nitrogen immobilization in decaying hardwood leaf litter as a function of initial nitrogen and lignin content. Can. J. Botany, 60, 2263-2269.

Berg, B. & McClaugherty, C., 1989. Nitrogen and phosphorus release from decomposing litter in relation to the disappearance of lignin. Can. J. Bot., 67, 1148-1156.

Berg, B. & Ekbohm, G., 1991. Litter mass-loss rates and decomposition patterns in some needle and leaf litter types. Long-term decomposition in a Scots pine forest. VII. Can J. Bot. 69, 1449-1456.

Berg, B. & Staaf, H., 1981. Leaching, accumulation and release of nitrogen in decomposing forest litter. In: Terrestrial Nitrogen Cycles. Clark, F.E. & Rosswall, T. eds. Ecol.Bull. (Stockholm) 33, 163-178.

Bloom, A.J., Chapin, F.S. III & Mooney, H.A., 1985. Resource limitation in plants - an economic analogy. Ann. Rev. Ecol. System., 16, 363-392.

Bosatta, E. & Agren, G.I., 1985. Theoretical analysis of decomposition of heterogeneous substrates. Soil Biol. Biochem., 17, 5, 601-610.

Bosatta, E. & Staaf, H., 1982. The control of nitrogen turn-over in forest litter. Oikos, 39, 143-151.

Christensen, B.T., 1992. Physical fractionation of soil and organic matter in primary particle size and density separates. Adv. Soil Sci., 20, 1-90.

Couteaux, M.M., Mousseau, M., Celerier, M.L. & Bottner, P., 1991. Increased atmospheric CO2 and litter quality: decomposition of sweet chestnut leaf litter with animal food webs of different complexities. Oikos, 61, 54-64.

Duchaufour, P., 1984. Edafolog!a. 1. Edafog nesis y Clasificaci"n. Masson, Barcelona.

Ferran, A. & Vallejo, V.R., 1992. Litter dynamics in post-fire successional forests of Quercus ilex. Vegetatio, 99-100, 239-246.

Fogel, R. & Cromack, K. Jr., 1977. Effect of habitat and substrate quality on Douglas-Fir litter decomposition in Western Oregon. Can. J. Botany, 55, 1632-1640.

Gallardo, J.F. & Moyano, A., 1987. Nitr"geno fijado en suelos de la Cuenca del Duero. Jornades sobre bases ecol giques per la gesti" ambiental. Diputaci" de Barcelona, Montesquiu.

Hadas, A., Molina, J.A.E., Feigenbaum, S. & Clapp, C.E., 1992. Factors affecting nitrogen imobilization in soil as estimated by simulation models. Soil Sci. Soc. Am. J.56, 1481-1486.

Hereter, A. & Vallejo, V.R., 1987. Evaluaci"n de la fertilidad de los suelos forestales: ejemplo del encinar de la Castanya (Montseny). Jornades sobre bases ecol giques per la gesti" ambiental. Diputaci" de Bacelona, Montesquiu.

Jansson, S.L. & Persson, J., 1982. Mineralization and immobilization of soil N. In: Nitrogen in agricultural soils. Stevenson, F.J. ed. Agronomy, 22, 229-252, ASA, Madison.

Kumada, K., 1987. Chemstry of soil organic matter. Elsevier, Tokyo.

McGill, W.B. & Cole, C.V., 1981. Comparative aspects of cycling of organic C, N, S, and P through soil organic matter. Geoderma, 26, 267-286.

McGill, W.B., Hunt, H.W., Woodmansee, R.G. & Reuss, J.O., 1981. Phoenix, a model of the dynamics of carbon and nitrogen in grassland soils. In: Terrestrial Nitrogen Cycles. Clark, F.E. & Rosswall, T. eds. Ecol. Bull. (Stockholm)33, 49-115.

Ohta, S. & Kumada, K., 1977. Studies on the humus forms of forest soils. III. Elementary composition of fractionated horizons of forest soils. Soil Sci. Plant Nutr., 23, 3, 355-364.

Ohta, S. & Kumada, K., 1978a. Studieson the humus forms of forest soils. V. Elementary composition of soil organic matter of brown forest soils. Soil Sci. Plant Nutr., 24, 1, 27-40.

Ohta, S. & Kumada, K., 1978b. Studies on the humus forms of forest soils. VI. Mineralization of nitrogen in brown forest soils. Soil Sci. Plant Nutr., 24, 1, 41-54.

Smith, O.L., 1982. Soil microbiology: a model of decomposition and nutrient cycling. CRC Press, Boca Raton.

Vallejo, V.R., 1983. Los suelos forestales de la Depresi"n Central Catalana. Tesis doctoral, Universidad Barcelona.

Vallejo, V.R. & Hereter, A., 1986. Analysis of the C/N ratio distribution in mediterranean forest soils. Transact. XIII Congress ISSS. 650-651, Hamburg.

Vallejo, V.R & Rosich, D., 1990. Nitrogen evolution in mediterranean forest soils. Transact. XIV Congress ISSS, 438-439, Kyoto.

Van Breemen, N., 1990. Impact of anthropogenic atmospheric pollution on soils. Transactions XIV Congress ISSS, vol. V. 258-262.

Vigil, M.F. & Kissel, D.E., 1991. Equations for estimating the amount of nitrogen mineralized from crop residues. Soil. Sci. Soc. Am. J. 55, 757-761.

Vitousek, P.M., Gosz, J.R., Grier, C.C., Melillo, J.M. & Reiners, W.A., 1982.Ecol. Monographs, 52,2, 155-177.

White, C.S., Moore, D.I., Horner, J.D. & Gosz, J.R., 1988. Nitrogen mineralization-immobilization response to field N or C perturbations: An evaluation of a theoretical model. Soil Biol. Biochem., 20, 101-105.

Mitt. d. Österr. Bodenkundlichen Gesellschaft, H. 47, S. 79 - 81, 1993

EFFECTS OF ANTHROPOGENIC N-DEPOSITION ON SOIL FAUNA-MICROBE INTERACTIONS AND THE IMPACT ON DECOMPOSITION PATHWAYS

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A great deal of research has been dedicated in the last decade to revealing hypothetical causes for forest decline, such as soil acidification, aluminium toxicity, ozone-induced nutrient deficiencies, magnesium deficiency. None of these can be considered a general cause for forest decline (1). In the past few years, excess nitrogen deposition has gained ground as an important factor. Although nitrogen availability is often a limiting factor in plant growth, deposition in excess of requirement may increase susceptibility to climatic and biotic stress and accelerate soil leaching (2).

These effects have been observed in pine stands in the Netherlands. There, nitrogen deposition can locally exceed 50 kg N ha^{-1} gr^{-1}. Further, the geographical pattern of forest decline fits in very well in this country with areas with high nitrogen deposition values (3).

In this study, it is shown that, in areas with high nitrogen deposition, not only fresh needles on trees but also the different organic soil layers have elevated nitrogen concentrations.

The impact of such elevated nitrogen concentrations on nitrogen dynamics have been studied by microcosms with separate soil horizons (: litter, F- and H-layers) using organic material from "clean" coniferous forests (*Pinus sylvestris* L.) with low nitrogen deposition (about 15 kg ha^{-1} y^{-1}) and polluted forests with high nitrogen deposition (40 kg ha^{-1} y^{-1}).

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Previous studies on nitrogen dynamics in coniferous forests have shown the reliability of similar microcosm studies compared with field data (4).

Nitrogen mobilization appears to be horizon-specific: in the litter layer of clean forests, there is a relatively small ammonium mobilization and no nitrate mobilization, probably due to the absence of nitrifying organisms. In the F-layer, a high nitrate mobilization is correlated with high numbers of autotrophic nitrifying bacteria (5).

Nitrogen mobilization from polluted litter shows a gradually increasing ammonium mobilization, indicating that the atmospheric nitrogen input is incorporated in the organic material and cannot be considered an easily leachable component of the needles. Nitrate mobilization, too shows an increasing trend. In spite of the negative effect of nitrogen input on nitrifying bacteria (5), the increasing ammonium availability is probably the cause of this increasing nitrate mobilization. However, the fact that total microbial activity, measured as microbial respiration, is not different in clean and polluted organic layers, points more to a shift in microbial composition than to a general negative effect on micro-organisms.

Soil fauna, such as collembola, are involved directly with ammonium and indirectly with nitrate mobilization. The increased ammonium leaching derives from excretion products, both faeces and urine. Calculated contributions to nitrogen mobilization based on individual nitrogen budget analyses agree with the data from these microcosm studies. Suppressing fungal and bacterial populations through their grazing activities the animals prevent subsequent microbial immobilization (6).

In polluted litter, soil animals decrease nitrogen mobilization. This is probably caused by their stimulation of the growth of special microbial groups, such as various basidiomycetes, which are sensible for high nitrogen concentrations and play an important part in the ligninolysis.

There are negative effects of high nitrogen levels on soil animals, too, revealing itself in retarded reproduction rate, which may have important implications on their population dynamics and thus on nitrogen dynamics.

References:

1. Blank, L.W., Roberts, T.M. & Skeffington, R.A. *Nature* 336, 27-30 (1988).
2. Nihlgård, B. *Ambio* 14, 2-8 (1985).
3. Verhoef, H.A. & Dorel, F.G. in *Air Pollution and Ecosystems*, pp. 847-851 (1988).
4. Verhoef, H.A. & Goede, R.G.M. de, in *Ecological interactions in soil*, pp. 367-376 (1985).
5. Verhoef, H.A., Dorel, F.G. & Zoomer, H.R. *Biology and Fertility of Soils* 8, 255 - 259 (1989).
6. Verhoef, H.A. & Brussaard, L. *Biogeochemistry* 11, pp. 175-211 (1990).

Mitt. d. Österr. Bodenkundlichen Gesellschaft, H. 47, S. 83 - 95, 1993

THE RELATIONSHIP BETWEEN DISSOLVED ORGANIC CARBON AND BASAL METABOLISM IN SOIL

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ABSTRACT

Respiration (basal metabolism) and dissolved organic carbon (DOC) are critical parameters in the studies of humification, but are both highly operationally defined. The former is used, depending on experimental conditions, as a measure of microbiological activity, biomass, mineralizable organic matter, and bioavailability. The latter can be either the truly mobile organic material or an aqueous extract, which in turn can reflect different pools, depending on the extraction temperature and the condition of the soil before extraction. Therefore, it is impossible to report a simple interaction or relationship between these 2 parameters.

One aspect does appear to be consistent both in the literature and in the research briefly reported here. The amount of respired carbon is always greater than the amount in the dissolved state. Therefore, whenever a relationship exists between respired and dissolved carbon, dissolved carbon can be considered to be an indicator of the bioavailability of the organic material in the relatively much larger non-mobile organic matter pool. The fluxes between the immobile and potentially mobile organic pools in soils need to be experimentally investigated and quantified, if we wish to fullfill the goals of this Concerted Action.

Furthermore, an important parameter in humification studies where DOC plays a role, is the degree to which DOC itself can be mineralized.

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Such direct measurements are rarely reported in the literature and need to be defined and standardized. Research has been initiated in this direction to see to what extent respiration can be used for this purpose.

DISSOLVED ORGANIC CARBON

This soil fraction is essentially physically defined as being soluble in water or weak aqueous ionic solutions. In practice, it is strongly influenced by the means, with which it is obtained. This is illustrated with a conceptual model in Fig. 1. In soil both the mobile organic carbon or matter (MOM) and water extractable organic carbon or matter (WEOM) pools are composed of DOC. The interaction between WEOM and MOM is largely a function of soil structure and water flow. Soils having a structure, which results in less preferential flow, will tend to have relatively more MOM. Water flow is a function of precipitation and rhizosphere demand and is critical, since without it there can be no MOM. Basically, WEOM is potential MOM. The sources of WEOM are litter, fertilizer, exudates, and the immobile organic matter, including the biomass. The extent that this material enters the WEOM pool is also experimentally determined. WEOM obtained from air dried soil will about 5 times greater than that from a moist soil, while oven drying will increase WEOM up to 100 times. The use of boiling water will cause an increase of about 10 fold.

Although the WEOM generally makes up less than 0.5% of the total organic carbon (TOC) in arable soil (eg. Burford and Bremner, 1975; McGill et al., 1986; Zsolnay and Steindl, 1991; Cook and Allan, 1992) it is, by definition, the most mobile abiotic organic pool. It can play a major role in the process of podzolization. Furthermore, if this material is labile, it can "fuel" biochemical reactions in areas not proximate to organic energy sources, such as litter, organic debris, and the rhizosphere (Martin, 1975; Haller and Stolp, 1985; Johansson, 1992). One such possible reaction of environmental importance is denitrification (eg. Burford and Bremner, 1975; Rolston et al., 1984; Katz et al., 1985; Seech and Beauchamp, 1988).

Generally, for the analysis of DOC wet dichromate oxidation is used (Walkley, 1947; Mebius, 1960). Quantification can then either be with titration or, more automatically, with infra-red absorption. An alternative method is the measuring of the absorption of the solution in the ultra-violet. This latter approach is only valid if the absorptivity of the DOC is constant. This is sometimes, but by no means always

(Zsolnay and Steindl, 1991), the case. It is however a very simple method and does give a rough comparative measure of DOC concentrations.

BASAL METABOLISM

This parameter is understood to be the amount of CO_2 released over a given time period. Normally, it is the result of metabolism and is often used as a measure of the amount of organic material, which has been mineralized. Since it can be expressed in moles or g carbon, it is well suited for the quantification of organic geochemical fluxes. Furthermore, it is ideally a measure of the organic substrate available for the biota, and thus a suitable parameter for studies of bioavailability. Unfortunately, in situ measurements in soil are not easy to make, and the sampled volume is difficult to define. Weier et al. (1991) found approximately 6 µg CO_2-C/ml of soil, while Linn and Doran (1984) reported about 16 mg/l of air- filled pore space. Another approach is to sample only at the surface with chambers. Freijer and Bouten (1991) did a comparison study and found dynamic chamber headspace sampling to be better than static sampling. Their data show an average amount of about 10 g CO_2- C/(d*m^2) with maximum values of 20g. Folorunso and Rolston (1985) found roughly on an average 7 g/(d*m^2) with maximum values being over 32g.

Unfortunately, with surface sampling one can not determine the extent of mineralization and degree of bioavailability at different soil depths. Also impacting conditions such as rhizosphere activity, temperature, and water content can not be controlled. Therefore, the soils samples are usually taken from the field and incubated in vitro in the laboratory. Mechanical manipulation, in the same way as plowing can, however, artificially increase respiration, by either increasing the number of sites available to the microorganisms and their extracellular enzymes or by improving the ability of oxygen to penetrate into the soil matrix. Studies done here have shown that this increase can be up to 50% depending on soil type. This effect can be minimized by the using of intact cores, which, however, complicates the experiment considerably.

Work is also done with substrate induced respiration (SIR) to investigate respiration potentials. The values obtained with this approach are naturally far greater than those which exist in situ. Fig. 2 conceptually summarizes the above.

INTERACTION

Since it is impossible to measure basal metabolism and DOC in a soil simultaneously, these 2 parameters must be determined either sequentially or in aliquouts. Furthermore, all the data in the literature are based on the WEOC rather than the MOC pool. Very few field studies have been published. Linn and Doran (1984), Folorunso and Rolston (1985), and Jäggi et al. (1991 VDLUFA Congress) all reported no relationship between basal metabolism in the field and WEOC.

Incubations in the laboratory can be static or dynamic. The former have the advantage that relatively little space is required, moisture content can be precisely regulated, and certain phenomena such as "initial bursts" can be well studied. Dynamic or flow-through incubations allow subtle temporal changes to be investigated, once the system has come to equilibrium. Also, there is no unnatural accumulation of CO_2 during the incubation. All the results presented here have been done under static conditions. One of the most cited results are from Burford and Bremner (1975) and are illustrated in Fig. 3. They did a 7 d incubation to determine total mineralizable organic carbon and found a strong correlation between the 2 parameters. Of interest is the fact the mineralizable organic carbon exceeded WEOC almost by a factor of 2. In a similar study done here (Fig. 4), the fit was less successful. Again, the mineralizable pool is greater then WEOC even after a 1 d incubation. Davidson et al. (1987) incubated forest soils for 7 d. Statistically, their results are difficult to interpret, since a few outliers have undue weight (Fig. 5), but a general relationship appears to be present, and mineralizable organic carbon exceeds WEOC by an order of magnitude. On the other hand, Beauchamp et al. (1980) as well as Wolters and Joergensen (1991) found no relationship between basal metabolism and WEOC in laboratory incubations.

A special category of WEOC is the hot water extractable organic carbon (HWEOC). Results are given from Davidson et al. (1987, Fig. 5) and from Körschens et al. (1990, Fig. 6). The presence of extreme values makes the accuracy of the coefficient of determination questionable, but the fit appears to be better than that obtained with WEOC. Similarly, WEOC obtained from air dried soil correlates better with basal metabolism than WEOC obtained from moist soils (Figs. 5 and 7).

A study was done here to see how WEOC concentrations change over a longer incubation period. Two different soils were incubated over an 18 wk period, and periodically the basal metabolism determined as an index of mineralization. Furthermore, on the sampling dates WEOC (room temperature, moist soil) was determined. Fig. 8 shows the mineralization of the TOC, as estimated from respiration over time. Lines for 1st order kinetics were fitted to the data. The reaction rate was 0.51×10^{-3} d^{-1} for the sandy and 0.12×10^{-3} d^{-1} for the muck soil. This would indicate that the TOC was mineralized in the sandy soil at 5.6 µg/(g*d) at the beginning and 5.3 µg/(g*d) at the end of the incubation. For the muck, the values were 14.3 and 14.1. The WEOC concentrations are shown in Fig. 9. The overall mean concentrations were 8.8 µg/g (oven dry weight basis) and 19.9 µg/g for the sand and muck resp. A test for randomness of runs confirms the visual impression that no significant ($P > 0.05$) trends are to be seen over the course of the incubation. Cook and Allan (1992) reported similar results for the WEOC obtained from old fields and a savanna.

The results (Fig. 8) indicated that mineralization was occurring during the incubation. In the case of the sandy soil, a 1st order kinetic model could be fitted very well while a simple linear mineralization model proved to be less satisfactory. The higher (4.3 times) rate for the sandy soil than for the muck could be attributed to qualitatively different TOC, but physical differences, such as the quantity of microsites, certainly must have played a major role (Burns, 1991). The daily amount of TOC respired was large (on an average 5.5 for the sandy and 14.2 µg/(g*d) for the muck soil) when compared to the amount of WEOC present at any one time. If one uses the not unreasonable mineralization rate of 8.5×10^{-3} d^{-1} for WEOC, less than 2% of the CO_2-C respired daily would have been from this pool. To explain this, the existence of a relatively large pool of immobile but biodegradable substrate must be assumed. This is conceivable (cf. Burns, 1991), but reported research dealing with this area is very limited. In terms of the conceptual model shown in Fig. 10, it means that $F_4 >> F_3$, primarily because TOC >> WEOC.

Since it can be assumed that a part of the WEOC was labile and since there was no significant change in the WEOC concentration over time, a flux (F_1) must have existed from the non-mobile soil matrix to the WEOC pool. Quite likely, there was also a flux (F_2), which removed organic material from the WEOM pool. The overall conclusion is that under no-flow conditions, such as those present during this incubation, the WEOM was quantitatively at steady state even over a longer time period. Furthermore, unless one assumes very high mineralization rate constants for this fraction (on the order of 1000×10^{-3} d^{-1}), there must also have been present a relatively large pool of labile, non-mobile organic material. The often conflicting results in the literature in regards to the relationships between basal metabolism and DOC are most likely the result of our inadaquate understanding of these pools and fluxes. A future research goal is to quantify the fluxes shown in Fig. 10 and to determine which environmental conditions effect their relative magnitudes. Also there is a paucity

(Zsolnay and Steindl, 1991; Qualls and Haines, 1992) of published results dealing with the direct mineralization of WEOM.

REFERENCES

Burford, J.R. and J.M. Bremner (1975). Relationships between denitrification capacities of soils and total water soluble and readily decomposable soil organic matter. Soil Biol. Biochem. 7: 389-394.

Cook, B.D. and D.L. Allan (1992). Dissolved organic carbon in old field soils Total amounts as a measure of available resources for soil mineralization. Soil Biol. Biochem. 24: 585-594.

Folorunso, O.A. and D.E. Rolston (1985). Spatial and spectral relationships between field-measured denitrification gas fluxes and soil properties. Soil Sci. Soc. Am. J. 49: 1087-1093.

Freijer, J.I. and W. Bouten (1991). A comparison of field methods for measuring soil carbon dioxide evolution Experiments and simulation. Plant Soil 135: 133-142.

Haller, Th. and H. Stolp (1985). Quantitative estimation of root exudation of maize plants. Plant Soil 86: 207-216.

Johansson, G. (1992). Release of organic-C from growing roots of meadow fescue (Festuca pratensis L). Soil Biol. Biochem. 24: 427-433.

Körschens, M., E. Schulz, and R. Behm (1990). Heißwasserlöslicher C und N im Boden als Kriterium für das N-Nachlieferungsvermögen. Zentralbl. Mikrobiol. 145: 305-311.

Linn, D.M. and J.W. Doran (1984). Effect of water-filled pore space on carbon dioxide and nitrous oxide production in tilled and nontilled soils. Soil Sci. Am. J. 48: 1267-1272.

Martin, J.K. (1975). 14C-labelled material leached from the rhizosphere of plants supplied continuously with 14CO2. Soil Biol. Biochem. 7: 395-399.

Mebius, L.J. (1960). A rapid method for the determination of organic carbon in soil. Analyt. Chim. Acta 22: 120-124.

Qualls, R.G. and B.L. Haines (1992). Biodegradability of dissolved organic matter in forest throughfall, soil solution, and stream water. Soil Sci.

Soc. Amer. J. 56: 578-586.

Seech, A.G. and E.G. Beauchamp (1988). Denitrification in soil aggregates of different sizes. Soil Sci. Soc. Am. J. 52: 1616-1621.

Walkley, A. (1947). A critical examination of a rapid method for determining organic carbon in soils: Effects of variation in digestion conditions and inorganic soil constituents. Soil Sci. 63: 251- 263.

Wolters, V. and R.G. Joergensen (1991). Microbial carbon turnover in beech forest soils at different stages of acidification. Soil Biol. Biochem. 23: 897-902.

Zsolnay, A. and H. Steindl (1991). Geovariability and biodegradability of water-extractable organic material in an agricultural Soil. Soil Biol. Biochem.

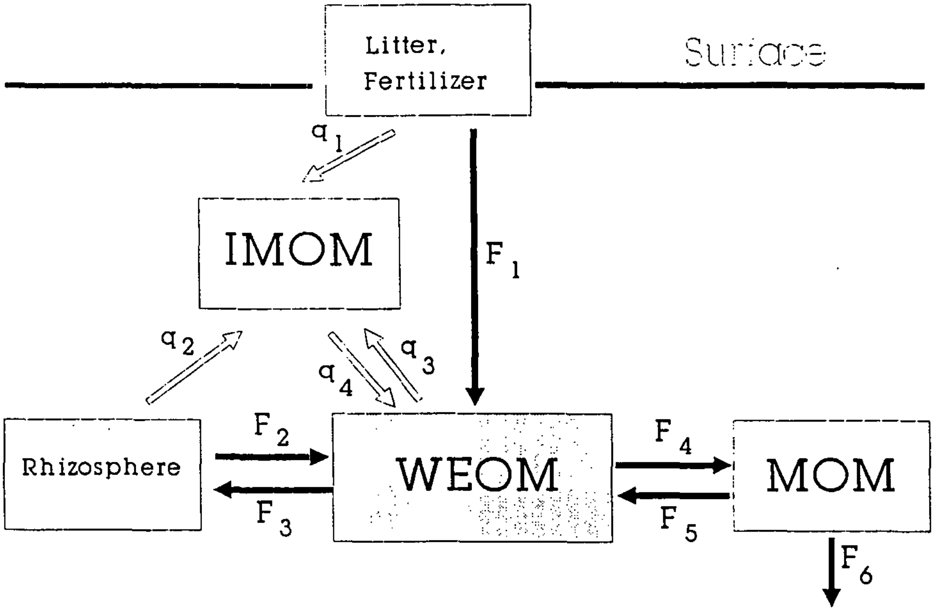


Fig. 1 Conceptual model of different organic carbon pools. MOM is mobile organic matter, WEOM water extractable organic matter, IMOM immobile organic matter. The F_i refer to material fluxes while the q_i are alterations of state.

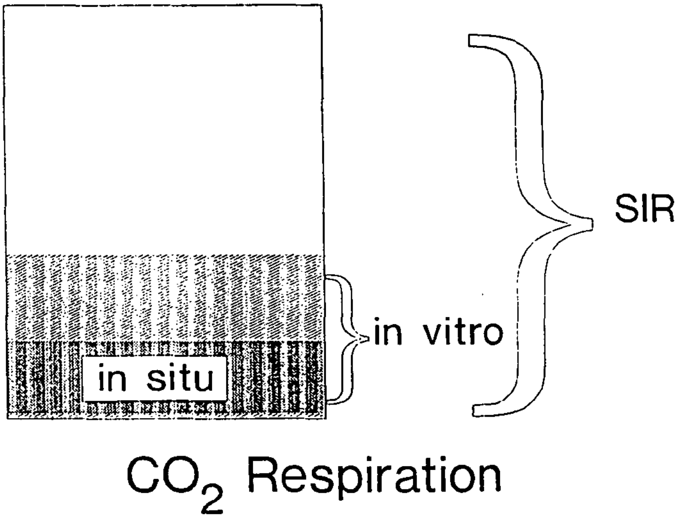
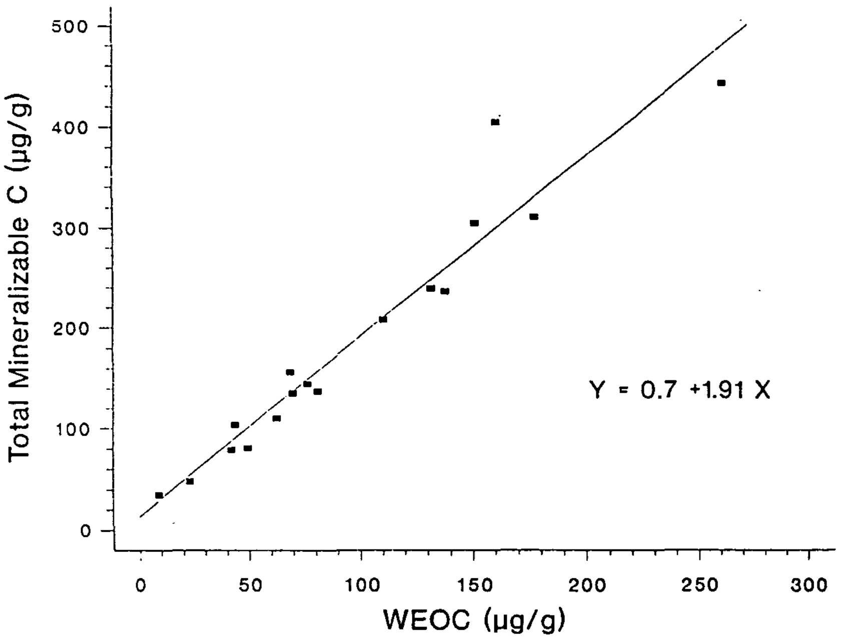
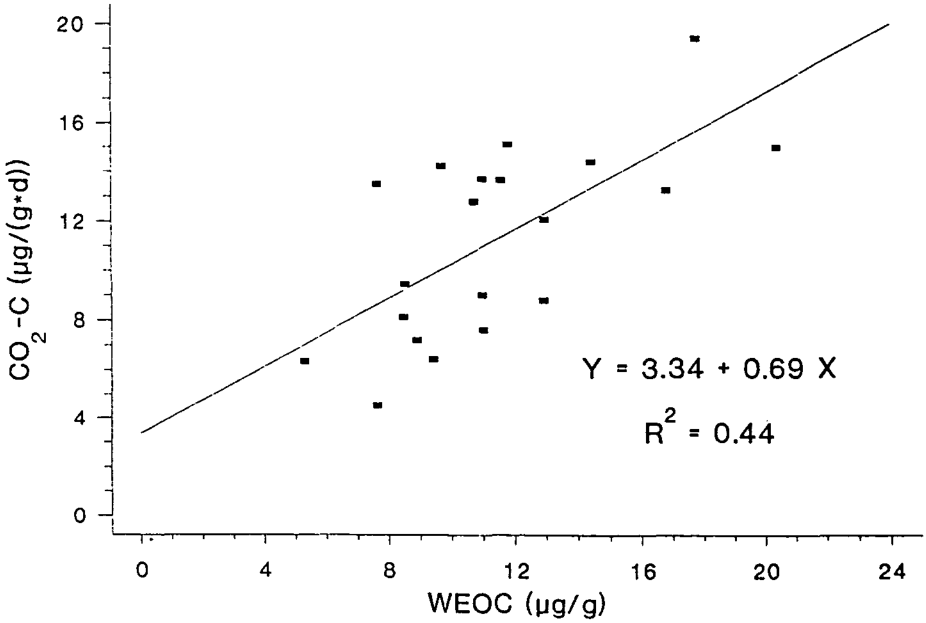


Fig. 2 Conceptual model of the different metabolism pools as a function of experimental approach.



(Buford and Bremner, 1975)

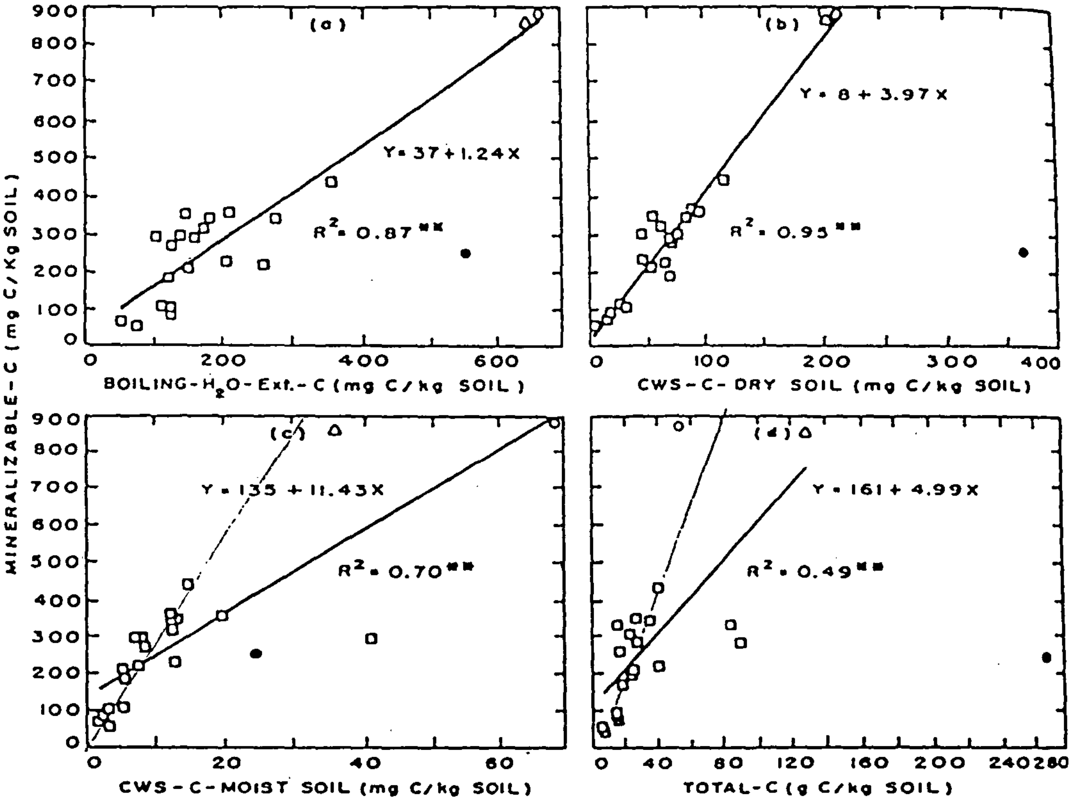
Fig. 3 Total mineralizable organic carbon (7d incubation) as a function of water extractable organic carbon (room temperature, moist soil) per g soil (oven dry weight).



Field Fresh

Fig. 4 Same as Fig. 3. The incubation was, however, only for 1 d.

1987 DAVIDSON, GALLOWAY, AND STRAND



Relationships between measures of C and mineralizable C. Symbols are the same as in Fig. 1. The Bayboro sample was excluded for regression analyses (N = 21). The measures of C are: (a) Boiling water extractable C; (b) Cold water soluble C of air-dried soil; (c) Cold water soluble C of field-moist soil; and (d) Total C.

Fig. 5 Illustration from Davidson et al. (1987).

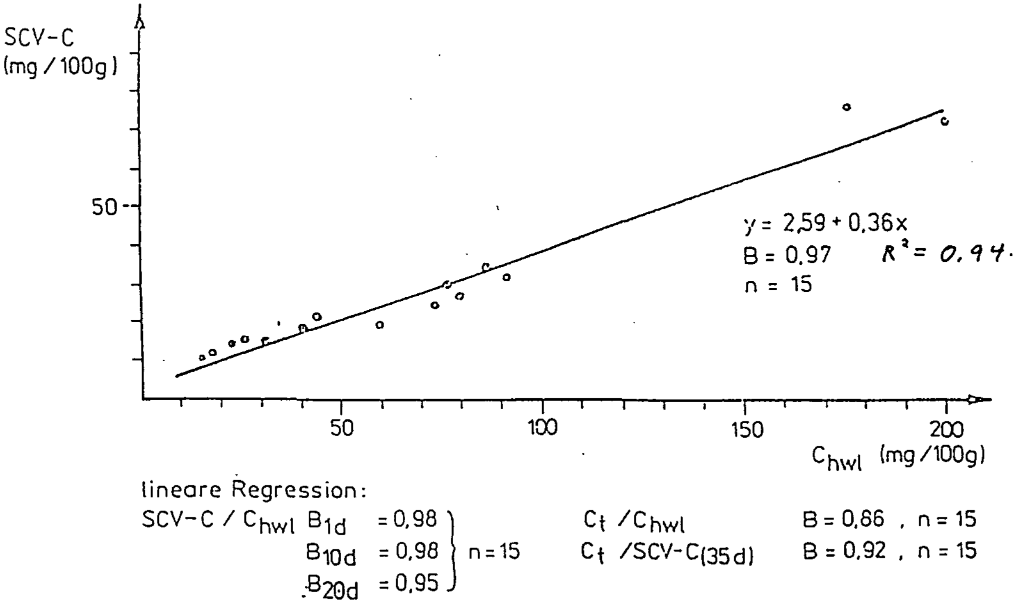


Fig. 6 Illustration from Körschens et al. (1990). The Y-axis gives the amount of carbon released as CO_2 over a 35 d incubation. The X- axis is the amount organic carbon extracted from moist soil with boiling water. B is the correlation coefficient.

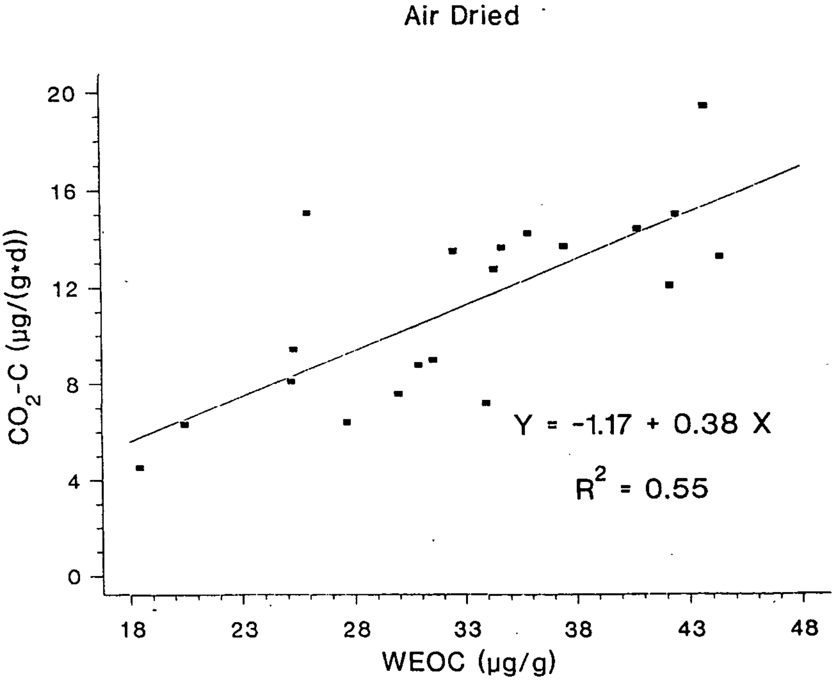


Fig. 7 Same as Fig. 4 except that the soil was air dried before extraction.

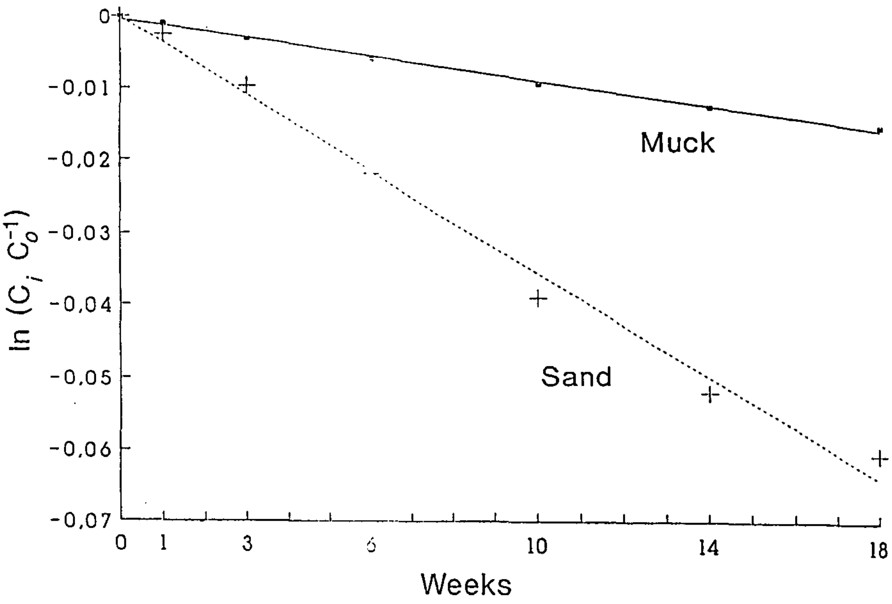


Fig. 8 First order fit of loss of organic carbon from soil as a function of incubation time. C_i is the amount present at sampling, C_o the amount initially present.

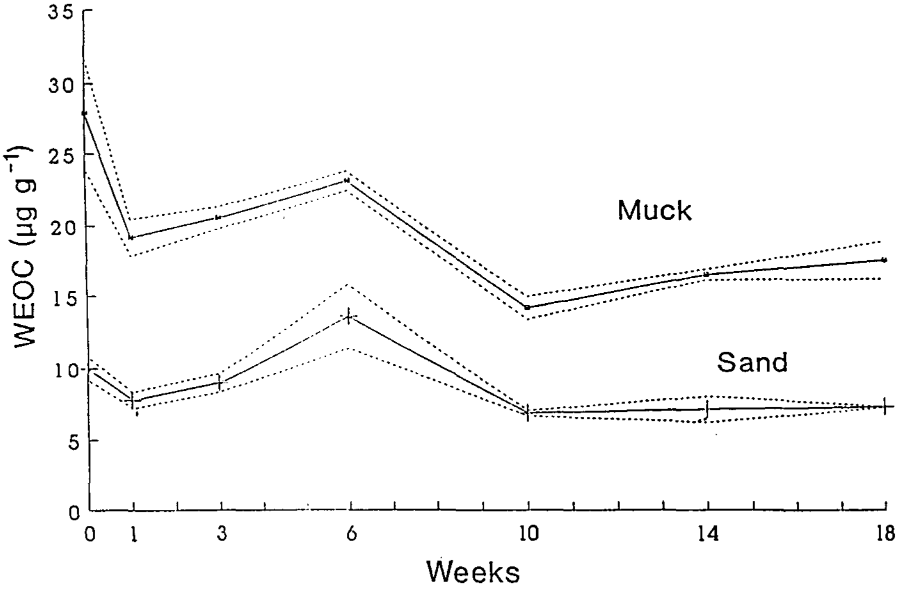


Fig. 9 Concentration of water extractable organic carbon (room temperature, moist soil) in incubated soil over time. The dashed lines represent the 90% confidence limits of the means (n=3).

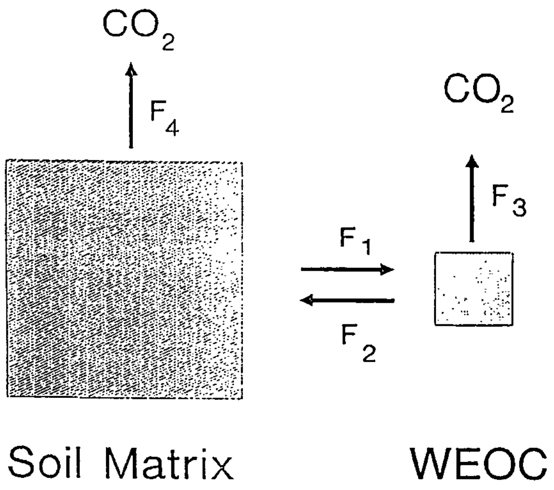


Fig. 10 Conceptual model showing possible fluxes (F_i) of organic carbon between soil matrix, water extractable organic carbon, and CO_2.

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SULPHUR DEPOSITION DISTRIBUTION AND SULPHUR BALANCE IN SICHUAN BASIN, CHINA

P. Chen and L. Li

ABSTRACT

Sichuan Basis in one of the main acid rain regions in China. Based on monitoring data, sulphur deposition in different areas of Sichuan Basin was calculated. The result shows that dry sulphur deposition predominates in urban areas, while wet sulphur deposition predominates in rural areas. The ammount of wet sulphur deposition during the warm season (May to October) was larger than during the cold season (November to April). Because of the vast area, tha ammount of sulphur deposition in rural locations accounted for 95% of total, or even more. The result from the analysis of sulphur balanceshows that 87% of artificial sulphur emission deposited into the basin, and only 13% sulphur was transported out of the basin.

Verteilung und Bilanzierung der Schwefelgehalte in Niederschlägen des Beckens von Sichuan, China

ZUSAMMENFASSUNG

Das Becken von Sichuan ist eine jener Regionen Chinas, welche am meisten vom sauren Regen betroffen ist. Sulfatgehalte und pH-Werte wurden an verschiedenen Standorten, getrennt in nasse und trockene Depositionen, laufend erhoben und daraus eine Bilanz berechnet. Während in städtischen Gebieten der Schwefelgehalt im trockenen Niederschlag überwog, überwog in den ländlichen Gebieten der Schwefelgehalt im Regen. In der warmen Jahreszeit (Mai bis Oktober) waren die Schwefelgehalte im Regen höher als in der kalten Jahreszeit (November bis April). Wegen der großen Fläche enthielten die Niederschläge in den ländlichen Gebieten aber mehr als 95% des Gesamtschwefels. Eine Bilanz aus Schwefelein- und -austrag bezogen auf das Becken zeigte, daß sich 87% der anthropogen im Becken emittierten Schwefelmenge in den Niederschlägen des Beckens wiederfanden und nur 13% davon aus dem Becken heraustransportiert wurden.

INTRODUCTION

Wet depostion (mainly rain or snow, with a pH level less than 5.6 is defined as acid rain. The pH- level is a comprehendive index reflecting the balance of anions and cations in precipitation. The transportation, conversion and deposition of acid- causing materials result in precipitation acidification. China's acid rain regions, especially Sichuan Basin and Guizhou Plateau, belong to typical regions of sulfate acid rain, where a great deal of SO_2 emitted from human activities. Obviously it is better to hold a discussion mainly on sources of sulphur deposition than pH only.

SOURCES AND DISTRIBUTION CHARACTERS OF SULPHUR DEPOSITION IN SICHUAN BASIN

I. Main Sources of Sulphur Deposition

Depositions can be divided into two categories: dry deposition and wet deposition. In Sichuan Basin, wet deposition mainly results from the process in which atmospheric sulfides are washed out by precipitation, while dry deposition results from the sedimentation of gaseous SO_2 and sulfide- containing atmospheric particulates, mainly.

1. Calculation equations of sulphur deposition

For wet deposition

$$D_w = \Sigma\, C_{wi} \cdot A_i / \quad A_i \cdot A_y$$

where D_w - wet deposition (g/m.yr)
C_{wi} - sulphur concentration of precipitation in event i (g/l)
A_i - precipitation in event i (mm/m^2)
A_y - annual mean precipitation (mm)

For dry deposition

$$D_d = (C_{SO2} \cdot V_d \cdot T)/2 + (C_{TSP} \cdot V_d \cdot T.f)$$

where D_d - dry deposition (g/m^2.yr)
C_{SO2} - mean concentration of SO_2 in ground atmosphere (g/m^3)
V_d - velocity of deposition, taking a constant 0.01 m/s
T - duration of deposition (s)
C_{TSP} - TSP concentration in ground atmosphere (g/m^3)
f - soluble sulphur in TSP, taking 0.02 on the basis of measurements

2. Results of Calculation

Based on monitoring data, the results of calculation for main cities and their suburbs (excluding Chongqing) are shown in Table 1 and 2. It is found that sulphur deposition was larger in urban areas than in their suburbs.

Table 1. Sulphur Depositions of Main Cities in Sichuan Province

City	Wet deposition		Dry deposition				Total deposition
	g/m^2.yr	%	g/m^2.yr		total	%	g/m^2.yr
			SO	TSP			
Chengdu	5.53	42	6.05	1.45	7.50	58	13.03
Leshan	5.63	28	12.10	2.21	14.31	72	19.94
Zigong	-	-	12.10	1.28	13.38	-	-
Yibin	19.44	27	50.11	2.38	52.49	73	71.93
Daxian	5.94	33	6.91	4.38	11.29	67	16.83
Nanchong	7.60	22	24.19	2.28	26.47	78	34.07
Luzhou	3.95	18	14.69	2.83	17.52	82	21.47
Mianyang	2.61	11	19.01	1.38	20.39	89	23.00
Wanxian	10.11	48	8.64	2.14	10.78	52	20.89
Fuling	8.72	16	39.74	4.70	44.44	84	53.16
Ya'an	3.85	64	1.73	0.48	2.21	36	6.06
Guangyuan	13.86	57	9.50	1.07	10.57	43	24.43

3. Analysis of Main sources

It is easy to see from the tables that urban sulphur deposition inside the basin derives from dry deposition, as a result of atmospheric sulphur pollution in the cities. In most of the cities, the dry deposition accounted for 60% or more of the total deposition, whereas wet deposition was less than 40%.

Sulphur deposition in suburbs mainly comes from wet deposition as a result of the wash-out by rain. In this place, the wet deposition accounted for 60% or more of the total. Owing to airborne pollutant transportation from large and median cities, Jiajiang had its wet sulphur deposition more than 80% of the total.

In short, dry deposition of sulphur is the main source for urban areas, whereas in rural and more peripheric locations, major amounts of sulphur are wet- deposited.

Table 2. Sulphur Depositions of Main Suburbs in Sichuan Province

City	Wet deposition		Dry deposition				Total deposition
	$g/m^2.yr$	%	$g/m^2.yr$		total	%	$g/m^2.yr$
			SO	TSP			
Chengdu	3.06	66	0.86	0.76	1.62	34	4.70
Leshan	5.30	69	1.72	0.62	2.34	31	7.64
Zigong	2.61	43	2.58	0.86	3.44	57	6.05
Yibin	11.28	40	15.55	1.17	16.72	60	28.00
Daxian	3.77	45	2.59	1.93	4.52	55	8.29
Nanchong	5.53	68	1.72	0.83	2.55	32	8.08
Luzhou	3.81	70	0.86	0.76	1.62	30	5.43
Mianyang	1.97	45	1.72	0.70	2.42	55	4.43
Wanxian	6.93	-	-	-	-	-	-
Fuling	2.57	26	6.05	1.24	7.29	74	9.86
Ya'an	5.49	82	0.86	0.35	1.21	18	6.70
Guangyuan	13.40	85	1.72	0.70	2.42	15	15.82
Jiajiang	5.20	83	0.60	0.45	1.05	17	6.25
Jiuzhaigou*	0.17	19	0.52	0.20	0.72	81	0.89

* A contrast region where the atmosphere is unpolluted. It is one of the national nature conservation areas in the northe of Sichuan.

II. Time - Space Distribution of Sulphur Deposition

1. Time Distribution

Dry sulphur deposition mainly comes from atmospheric SO_2 pollution. Data acquired from long-term monitoring show that SO_2 pollution level inside the is higher in winter and lower in summer. This rule in in accordance with the rule of dry sulphur deposition. Within this context, however, we emphasize the discussion of wet sulphur deposition distribution. Since the major portion of wet deposition takes the form of SO_4^{2-} existing in precipitation, we begin the discussion with SO_4^{2-}.

In Fig. 1, the curve shows the change of SO_4^{2-} concentration in precipitation in Chengdu throughout a year. The SO_4^{2-} concentration was higher in winter than in summer, The SO_4^{2-} deposition depended not only on the SO_4^{2-}concentration, but also on the amount of precipitation. Fig. 2 shows the trends of wet deposition of SO_4^{2-} in

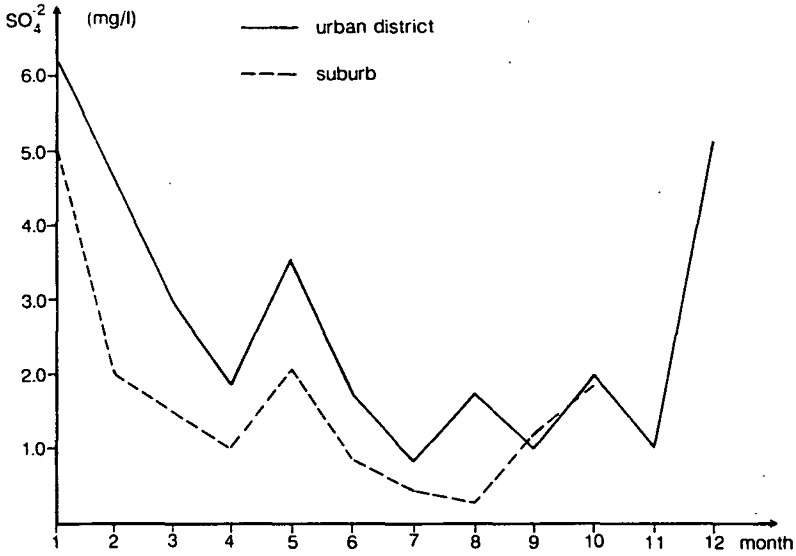

Fig. 1 SO_4^{-2} concentration charge in precipitation in Chengdu

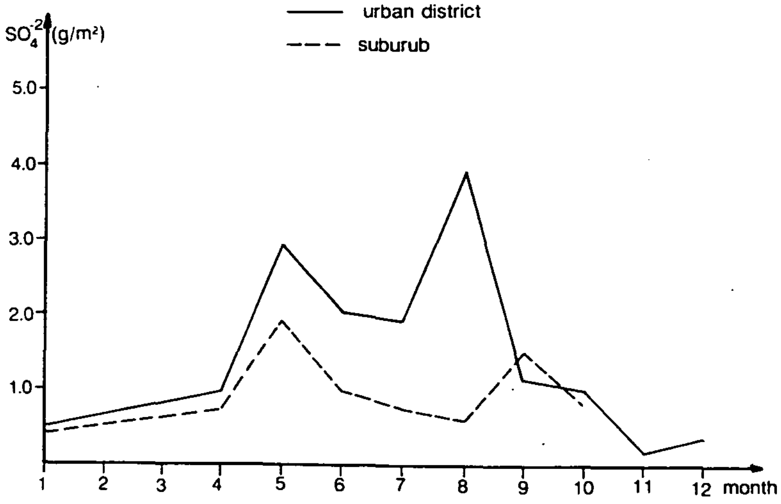

Fig. 2 Trends of wet deposition of SO_4^{-2} in Chendu

Chengdu. It was higher in summer than in other seasons, and lowest in winter. From the two curves, it can be concluded that the total SO_4^{2-} deposition was higher in urban area than in the respective suburbs. The changing rule of wet deposition of SO_4^{2-} in some main cities in Sichuan is listed in Table 3. The rule is in accordance with the rule of Chengdu.

Summing up, wet sulphur deposition inside the basin occurred frequently in the seasons of summer and spring, and the deposition amount in urban area was larger than that in its suburb.

Table 3. The Change of wet deposition of SO_4^{2-} in some cities in Sichuan Basin

		Spring		Summer		Autumn		Winter		Annual
City		Amount	%	Amount	%	Amount	%	Amount	%	Amount
Daxian	urban	5.78	35%	2.94	18%	4.20	25%	3.71	22%	16.63
	suburb	5.78	52%	1.59	14%	2.05	18%	1.79	16%	11.30
Nanchong	urban	6.98	31%	8.21	36%	2.37	10%	5.25	23%	22.79
	suburb	3.89	23%	7.02	42%	1.45	9%	4.24	26%	16.60
Wanxian	urban	10.86	36%	10.18	34%	1.81	6%	7.49	24%	30.34
	suburb	8.10	39%	6.81	33%	1.51	7%	4.36	21%	20.78
Guangyuan	urban	8.56	21%	14.47	35%	16.73	40%	1.81	4%	41.57
	suburb	11.79	30%	18.97	47%	8.56	21%	0.87	2%	40.19
Mean	urban	8.04	29%	8.95	32%	6.27	23%	4.57	16%	27.83
value	suburb	7.41	33%	8.60	39%	3.39	15%	2.82	13%	22.22

2. Space Distribution

Fig 3. shows the sulphur deposition distribution of among the main cities as well as between the respective urban districts and suburbs inside the basin. In the northwest part of the basin, the sulphur deposition in the cities is usually higher than in the southeast part, because of topographic reasons.

SULPHUR BALANCE IN SICHUAN BASIN

Since burning of high sulphur-containing coal as fuel, Sichuan Basin has become one of the highest SO_2 emission regions in China. The amount of SO_2 emission (including Chongqing) accounts for one tenth of the total in China. This is one of the radical reasons of higher sulphur deposition in the basin.

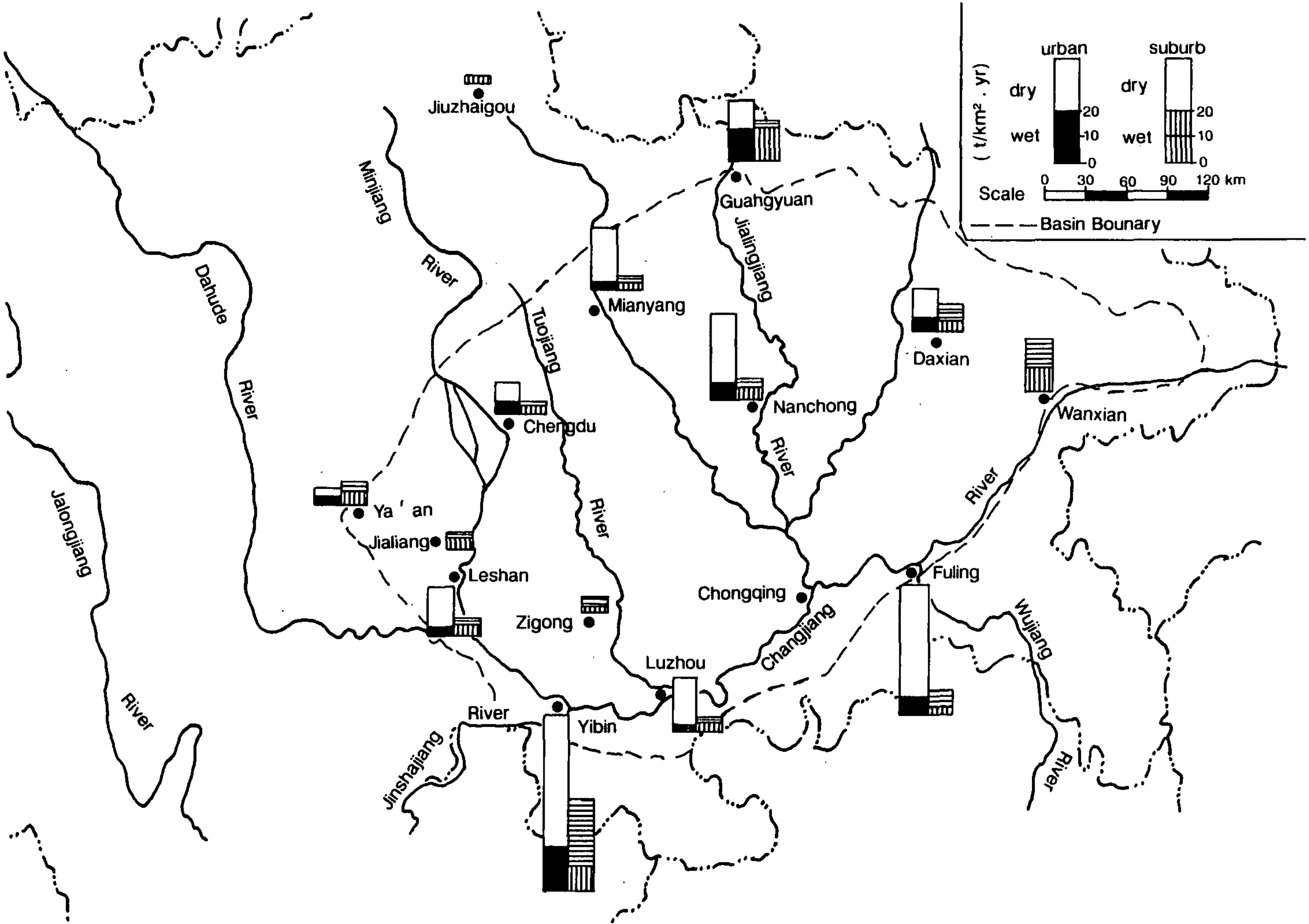

Fig. 3 The Distribution of Sulphur Deposition in Sichuan Basin

The balance relation between SO_2 emission and sulphur deposition is the important basis for working out the countermeasures of controlling acid deposition inside the basin.

I Sources of Sulphur

There are two types of sulphur sources. The first is artificial emission, mainly as SO_2, as well as from soot and dust, all from coal burning in the processes of cooking and industrial production. Soot and dust contain soluble sulfates. The second is external influence. This can be estimated by monitoring in the surrounding mountains of the basin.

1. Artificial Emission

It has been estimated that 781 400 tons of SO_2 gas is emitted annually into the air, over about 150 000 km^2 area of Sichuan Basin, equivalent to 390 700 tons of sulphur. Additionally, the total soot and dust amount is about 115 100 tons per year, equivalent to 22300 tons per year (2% soluble sulphur contents). Summing up these two items, the annual artificial sulphur emission is 413 000 tons per year, or 2.75 tons per km^2.

2. Background Deposition Level

Artificial emission is the radical factor effecting sulphur deposition inside the basin. Meanwhile effects caused by nature and long- distance transportation result in the background level of sulphur deposition. We choose Jiuzhaigou as the background area. Based on our monitoring, the annual mean sulphur deposition level is 0.89 tons per km^2. Considering the affecting of tourism, 0.445 t/km^2 is taken as the background level. Hence, the annual mean background level of sulphur deposition over the basin is 66750 tons. This amount is equivalent to 16% of artificial emission.

II Sulphur deposition

1. Area Division of Sulphur Deposition

According to its features, four types of area can be found (see table 4).

Table 4. Four Types of Area Inside the Basin and their Dimension

Type	Chengdu	Median cities	Towns	Countryside
Dimension (km^2)	400	645	1500	147455

2. Calculation

Total deposition amount: $D = D_i$

$$D_i = d_i . S_i$$

where D - total deposition amount

D_i - regional deposition amount of type i

d_i - regional deposition amount per area of type i

S_i - dimension of type i

Deposition amount of median cities: $D_i = \sum d_{ij} . S_{ij}$

where j - median citiy j

3. Determination of Sulphur Deposition in the Countryside

SO_4^{2-} in precipitation as well as SO_2 and TSP in the atmosphere were measured in Guangyuan (in the north of the basin), the West Sichuan Plain (in the west of the basin), Leshan, Emei and Ranshou (in the southwest of the basin), and Nanchong (in the centre of the basin). Their mean levels are: SO_4^{2-}: 5 mg/l

SO_2 : 0.007 mg/m^3

TSP : 0.148 mg/m^3

Based on the features of precipitation in the basin, the annual mean precipitation in the countryside in the basin is 1000 mm. The sulphur deposition in the countryside is:

wet deposition:	1.67 $t/km^2.yr$	60%
dry deposition:	1.11 $t/km^2.yr$	40%
total deposition:	2.78 $t/km^2.yr$	100%

The above results show that wet deposition took the major portion (60%) of the total.

4. Sulphur Deposition

Table 5 shows the dry and wet depositions in different areas in the basin. Further on, the ration of dry to wet deposition in different areas in the basin is shown in table 6.

Table 5. Sulphur Deposition Amount in Sichuan 3asin, t/yr

	Urban District			Suburb			Total		
	Dry	Wet	Total	Dry	Wet	Total	Dry	Wet	Total
Chengdu	750	553	1303	486	924	1410	1236	1477	2743
Median cit.	3063	1061	4124	2242	2977	5219	5305	4038	9343
Towns							2280	3027	5307
Countryside							163675	246250	409925
Total							172496	254792	427288

Table 6. Ratios of Dry Sulphur <dposition to Wet Sulphur Deposition in Sichuan Basin

	Dry Deposition		Wet Deposition		Total	
	tons	%	tons	%	tons	%
Cities	8821	2.1%	8542	2.0	17363	4.1%
Countryside	163675	38.3%	246250	57.6%	409925	95.9%
Total	172496	40.4%	254792	59.6%	427388	100 %

It can be seen from Table 6 that the sulphur deposition in the cities and their suburbs accounted for only 4.1% of the total, because of their smaller area, even if they get a greater amount of sulphur deposition per area. In the area of cities including their suburbs, the sulphur deposition was approximately equal to wet sulphur deposition. Because of the large area, the sulphur depositon in the countryside accounted for 95.9% of the total.

From the above discussion it can be concluded, that the vast countryside of Sichuan Basin is the major acceptor of sulphur deposition, while wet sulphur deposition is the major form of sulphur deposition in the countryside.

III Sulphur Balance

At present, the artificial sulphur emission in Sichuan Basin is 413000 t/yr, whereas the total amount of dry and wet sulphur deposition is 427288 t/yr. After the subtraction of the background level, the real (net) sulphur deposition in the basin is 360538 t/yr. That means, sulphur deposition accounts for 87% of the emission. Most of the artificial sulphur emission deposits onto the basin by dry and wet ways, because of its special topography and climate. The rest of 52463 tons of sulphur mostly deposits onto the high surrounding mountainous areas, where it is humid and rainy with much vegetation. Therefore few sulphur emission is transported to other provinces by airflow. This is quite different from sulphur transportation in Europe.

CONCLUSION

Through the above discussions, we get

1. In Sichuan Basin, sulphur deposition (mainly sulphur deposition) comes from urban areas of the cities. Wet sulphur deposition is the major portion in suburbs and vast rural areas.
2. The amount of wet sulphur deposition per area in Sichuan Basin in warm season (May to October) is larger than that in cold season (November to April), and in urban areas it is larger than in suburbs.
3. In the vast rural areas of Sichuan Basin, the annual sulphur deposition is 409925 tons (95.9% of the total). Among them, wet sulphur deposition is 246250 tons (60%), and dry sulphur deposition is 163675 tons (40%). Wet sulphur deposition is the major portion of sulphur deposition in rural areas.
4. Because the special topography and climate are unfavourable for transportation, most of the artificial sulphur emission (87% of the total) deposits onto the basin, and 13% of artificial sulphur emission mostly deposits onto the surrounding mountains. Only few artificial sulphur emission is transported to other provinces.

REFERENCES

[1] Roffman A, The Journal of Environmental Science 23.2 (1980)

[2] Zum Problem "Saure Niederschläge", Bericht über eine international Tagung in Lindau/Bodensee, 1983

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Bodeninformationssystem des Landes Nordrhein-Westfalen (BIS NRW)

Vorstudie über Schwermetallgehalte in Waldböden des Kreises Recklinghausen

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1. Einleitung

Das Bodeninformationssystem des Landes Nordrhein-Westfalen (BIS NRW) liefert die Informationsgrundlagen für die Maßnahmen im Bodenschutz (THIELE & ROENICK 1992). Hierzu werden die landesweit verteilt vorliegenden Datensammlungen für entsprechende Auswertungen in Anwendungen des BIS NRW verfügbar gemacht. Die Daten verbleiben weiterhin in der Verantwortung der datenerhebenden bzw. datenführenden Stellen. Sie werden nach dem Auszugsprinzip zur Verfügung gestellt. Der Nutzer kann sich im Kernsystem des BIS NRW über die im Lande verfügbaren Daten und über die in der Sammlung der Anwendungen vorgehaltenen Auswertungen informieren und bei Bedarf die Ergebnisse von Anwendungen in sein eigenes DV-System transferieren.

2. Prototyp des BIS NRW

Das BIS NRW wird schrittweise eingerichtet. Der erste Schritt umfaßt die Einrichtung des Prototyps (THIELE, NEITE & GOLLAN 1994). Mit dem Prototyp werden unter dem Thema "stoffliche Bodenbelastung" die wesentlichen Funktionen des BIS NRW eingerichtet. Neben dem Prototyp wird das Fachinformationssystem Stoffliche Bodenbelastung (FIS StoBo) als integraler Bestandteil des BIS NRW aufgebaut (NEITE; THIELE; GOLLAN 1994).

Im einzelnen werden die folgenden Systemkomponenten realisiert:

- Das Kernsystem mit den Metadaten, den Suchhilfen und den Systemhilfen
- Die Sammlung der Anwendungen mit ca. 30 vorgefertigte, "standardisierte" Auswertungen

Für die Anwendungen werden im Prototyp die folgenden Datensammlungen erschlossen und Auszüge daraus bereitgestellt:

- Fachinformationssystem Stoffliche Bodenbelastung (FIS StoBo)
- Datensammlung Staubniederschlag
- digitale Bodenkarte 1:50.000 (BK 50 d)
- Amtliches Topographisch-Kartographisches Informationssystem (ATKIS)

Für die Darstellung von Ergebnissen werden die nachstehend aufgeführten Karten eingesetzt:

- topographische Karte 1:50.000 (TK 50) im Rasterformat
- Karte der Verwaltungsgrenzen 1:200.000 im Vektorformat
- Karte der naturräumlichen Gliederung im Vektorformat

3. Vorstudie "Schwermetallgehalte in Waldböden des Kreises Recklinghausen"

Zur Vorbereitung auf die Einrichtung des Bodeninformationssystems wurde eine Vorstudie (KAZDA, NEITE & THIELE 1993) am Beispiel der Schwermetallgehalte in Waldböden des Kreises Recklinghausen durchgeführt. Mit Hilfe des DVWK-Modells wurde auf Basis der Daten aus der

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digitalen Bodenkarte 1: 50 000 eine Abschätzung der potentiellen Grundwassergefährdung vorgenommen (Deutcher Verband für Wasserwirtschaft und Kulturbau 1988).

In der Vorstudie wurde anhand vorhandener Daten geprüft, welche Probleme bei der Beschaffung und Verarbeitung der Daten auftreten. Die Daten über Schwermetallgehalte wurden von der Landesanstalt für Immissionsschutz und vom Kreis Recklinghausen zur Verfügung gestellt. Die Gebiete der Waldnutzung wurden aus der topographischen Karte 1:50 000 im Rasterformat entnommen. Das Geologische Landesamt stelllte die digitalen Geometrien der Bodenkarte im Vektorformat zur Verfügung. Die für das DVWK-Modell notwendigen Daten mußten in der Vorstudie den jeweiligen Objektnamen und der Legende der analogen Karte entnommen werden.

Die Anwendung des DVWK-Modells wurde in zwei Varianten durchgeführt:

1. Die Berechnungen wurden flächenbezogen mit generalisierenden Annahmen für den pH-Wert und den Humusgehalt (Medianwerte aus den Meßwerten) durchgeführt.
2. Die potentielle Grundwassergefährdung wurde für die einzelnen Meßpunkte bestimmt, wobei die bodenkundlichen Parameter der digitalen Bodenkarte entnommen wurden

Beide Varianten führten zu vergleichbaren Ergebnissen. Der Vergleich der Schwermetallgehalte in Böden mit der potentielle Grundwassergefährdung zeigt zudem, daß Probenahmeorte mit hoher Schwermetallanreicherung überwiegend in Gebieten mit niedrig bis mittel einzustufender potentieller Grundwassergefährdung liegen.

Für zukünftige Auswertungen ist anzumerken, daß eine überarbeitete Fassung des DVWK-Modells zur Abschätzung der potentiellen Grundwassergefährdung angewendet werden sollte (vgl. BLUME & BRÜMMER 1991).

4. Ausblick

Nach Fertigstellung wird der Prototyp in Abstimmung mit den datenführenden Stellen und Nutzern getestet. Anschließend geht der Prototyp in die Basisversion über. In der Basisversion wird das BIS NRW durch Erschließung weiterer Themen und Datensammlungen und durch Einrichtung zusätztlicher bzw. Erneuerung vorhandener Funktionen ausgebaut. Zur Zeit werden Projekte zur Erstellung von digitalen Bodenbelastungskarten und zur Nutzung des BIS NRW in der Bauleitplanung und in raumordnerische Verfahren vorbereitet.

5. Literatur

BLUME, H.-P., BRÜMMER, G. (1991): Prediction of heavy metal behavior in soil by means of simple field test. Ecotoxicology and Environmental Safety 22, 164- 174.

Deutscher Verband für Wasserwirtschaft und Kulturbau (1988): Filtereigenschaften des Bodens gegenüber Schadstoffen. Tei 1: Beurteilung der Fähigkeit von Böden, zugeführte Schwermetalle zu immobilisieren. Verlag Paul Parey, Hamburg und Berlin, Heft 212. 8S.

KAZDA, M., NEITE, H., THIELE, V. (1993): Flächenhafte und punktuelle Anwendung des DVWK-Modells zur Einschätzung der Grundwassergefährdung durch Schwermetalle. Mitteilungen der Deutschen Bodenkundlichen Gesellschaft, in Vorbereitung

NEITE, H., THIELE, V., GOLLAN, B. (1994): Fachinformationssystem Stoffliche Bodenbelastung (FIS StoBo) - Zusammenfassung der Ergebnisse, Abschlußbericht. Hrsg.: Bodenschutzzentrum des Landes Nordrhein-Westfalen, in Vorbereitung

THIELE, V., ROENICK, C. (1992): Vor- und Hauptuntersuchung zur Einrichtung des Bodeninformationssystems des Landes Nordrhein-Westfalen (BIS NRW) - Zusammenfassung der Ergebnisse, Abschlußbericht. Hrsg.: Bodenschutzzentrum des Landes Nordrhein-Westfalen, Oberhausen. 23 S.

THIELE, V., NEITE, H., GOLLAN, B. (1994): Bodeninformationssystem des Landes Nordrhein-Westfalen (BIS NRW) - Einrichtung des Prototyps. Zusammenfassung der Ergebnisse, Abschlußbericht. Hrsg.: Bodenschutzzentrum des Landes Nordrhein-Westfalen, in Vorbereitung

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