Regional Copper-Nickel Study: Soil Decomposition Studies

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INTRODUCTION

Decomposition is an essential process governing the rate of element cycling in terrestrial ecosystems. Soil microflora and micro and mesofauna play important roles in decomposing litter in forests, and heavy metals pollution has been shown to restrict the activities of these organisms (Bond et al. 1976; Tyler 1974, 1975a,b, 1976a). When decomposition rates are reduced, essential nutrients are bound in accumulating litter, and, ultimately, ecosystem production is decreased.

Several factors affect litter decomposition including soil temperature and moisture content, plant species contributing to litter and the degree of decomposition of organic matter. Such factors as pH and chemical composition of litter influence its suitability as a substrate for different organisms. All of these factors have been recently reviewed by Williams and Gray (1974) and will not be discussed in detail in this paper.

The purpose of this report is to present the results of litter decomposition studies conducted by the Regional Copper-Nickel Study. We have also included a review of the literature pertaining to the effects of heavy metals on soil decomposition. To the limited extent possible, results from this study are discussed in light of observations made elsewhere.

METHODS

Terrestrial habitats within the Study Area were characterized on 48 plots that were selected to represent the range of vegetation and soil variability. Six of these plots, which were approximately 1 ha in size, were selected as sites for litter decomposition studies. Plots were selected from among different aged aspen stands that are distributed north and south of the Laurential Divide (Figure 1). Plots north of the Divide are on rather shallow, sandy-loam soils that occur in hilly topography, whereas plots to the south are on deeper, more level loamy soils of the Toimi Drumlin Field. Two recent clearcuts (< 5 years old) were selected, one north of the Divide and the other south. The remaining four stands were selected from well-stocked aspen stands that are 30 or more years old.

Several methods are available for measuring soil decomposition rates, but many require expensive equipment that only simulate field conditions. These microcosm studies provide precise measurements of CO_2 evolution and O_2 consumption. We chose, however, to utilize much simpler techniques. Nylon-mesh bags containing fresh litter and placed on the forest floor have been used extensively to measure litter decay rates (Bocock et al. 1960, Bocock 1964, Heath et al. 1966, Crossley and Witkamp 1964, Gosz et al. 1973; and Grigal and McColl 1977). Because they provide a simple means of accurately determining decay rates, litter decomposition bags were used in this study also.

In late-March 1977, aspen (Populus tremuloides) leaves were collected near the U.S. Forest Service Kawishiwi Field Laboratory. These leaves were placed in large plastic open containers and stored in a cooler (at 4° C). Leaves

were mixed periodically to promote moisture equilibration, and after 3-4 weeks, litter bags were filled with 25 gm (wet weight) of leaves. Litter bags were constructed from 8 inch by 10 inch fiberglass screening (1.5 mm mesh size). Every tenth bag that was filled was set aside. After drying for 24 hours at 80° C, the contents of these bags were weighted, and moisture content was calculated as a percentage of wet weight.

Between May 4 and 6, 1977, litter bags were placed in the field. Twenty bags were placed on each of the six plots in four groups of 5 bags (see Figure 2). At approximately four week intervals, one bag was selected from each of the four groups at each plot. Bags were returned to the laboratory and their contents were placed in brown paper bags that were then placed in an oven and dried at 80° C for at least 24 hours. After drying, the leaves were weighted to $\pm$ .1 gm.

At each plot data were collected in order to describe the overstory and forest floor. The diameter at breast height (dbh) (4.5 ft = 1.5 m) was recorded for all living stems that were > 7 cm dbh on each of five, 15 x 15 m subplots. From these data diversity and cover estimates were obtained. Crown densities were determined at eight locations in each subplot using a densometer held approximately 1 m above the ground. Forest floor samples were obtained with a 5.4 cm diameter impact corer. The weight of each of twelve samples was recorded after drying for 24 hours at 80° C.

Soils of the sample plots were determined by refering to resource inventories published by the Minnesota Land Management Information System and the MINESITE Project, Minnesota Department of Natural Resources. Field checking was used to verify information taken from published maps. Precipitation data were taken from the records of stations located at Hoyt Lakes and Babbitt.

Because bags initially contained almost 10 gm, percent loss can be calculated by subtracting the values in Table 2 from 10 and multiplying the differency by 10. The high value at plot G32 on June 6 is within the 95 percent confidence limit for the original bag weight and probably indicates that little decomposition occurred at this plot between May 4 and June 6.

Data in Table 2 are depicted graphically in Figure 3. Also included in Figure 3 is a bargraph of 23-hour precipitation recorded at Babbitt, Minnesota. Although precipitation data from Hoyt Lakes differ from those at Babbitt when individual days are compared, weekly trends are the same for the two sites. Adequate moisture fell throughout the sampling period although precipitation was below the seasonal average betweeen early July and mid-August.

Data in Figure 3 were fitted to a simple linear model:

$$y = a + bt \qquad (1)$$

where y is bag weight as a percent of original dry weight, t is time in weeks and a and b are constants. The constants(a and b) and correlation coefficients (r) derived for each site are presented in Table 3. An overall regression for all sites yields the equation $y = 1.00 - .0109t$ ($r = .94$). A one-way ANOVA was used to test differences between sampling dates at a given site and between sites for a given date. Results of the temporal comparisons are presented in Table 4. Analysis of spatial variation shows that only the value for G32 on June 6 was significantly different from values for all other plots ($P < .05$). Variability of the June G32 values was not large, suggesting that the results are not due to sampling error.

DISCUSSION

Seasonal Variation--Although temporal variations within plots generally fit a linear model, examination of Figure 3 indicates that decomposition slowed on most plots during mid summer. Bag weights actually increased on four of six plots between mid July and mid August. Gosz et al. (1973) observed increases in litter bag weights during the growing season in hardwood stands in New Hampshire, and concluded that increases were probably caused by the accumulation of fine litter and insect frass. Increases noted by Gosz et al. occurred during May and June, however, when large amounts of flower parts, bud scales, etc. fall to the ground. Increases observed in northeastern Minnesota occurred after one would normally expect these plant parts to fall to the ground. An alternate hypothesis proposed by Gosz et al. is that increases during the growing season are due to increases in microbial populations. Because the northeastern Minnesota increases occurred during a period with little rainfall, this hypothesis seems even less satisfactory than the former.

In addition to the May-October period, regression analyses were performed for the periods May-July and August-October. Decomposition rates (b times 100 in the linear model) averaged 1.65 percent per week over the six-month period, whereas rates during May-July averaged 1.82 percent per week. Rates in the autumn (August-October) averaged 2.57 percent per week with a maximum of 4.70 percent at plot G12. Correlation coefficients for the three fall sampling periods were generally lower reflecting decreased decomposition during the 1st sampling period at all plots except G12. The increased decomposition between August 19 and September 20 corresponds closely to increased rainfall during late summer. Factors other than rainfall may also be involved. The rapid weight loss may reflect a washing out from the bags by fine particles

that were broken down earlier. Insufficient rainfall may have prevented washing of materials from the bags between mid July and mid August. Also, the early fall increase in decomposition correlates with the period when leaf senescence begins. The release of nutrients from canopy leaves and their subsequent transport to the forest floor in throughfall may stimulate litter decomposition.

Plots generally followed the same seasonal trends, but some exceptions did occur. The high June value for G32 has already been discussed. Other differences include increased decomposition rates during July-August at plot G10 and during September-October at plot G12. These values may reflect actual differences in decomposition rates due to microclimatic differences among sites, or they may simply represent sample to sample variability. As has been mentioned, however, variability associated with average values was not great, and it seems likely that differences are associated with real differences in decomposition.

Seasonal variation in decomposition rates that were observed in this study were not observed by Grigal and McColl (1977) who sampled aspen decomposition in stands northwest of the Study Area. A careful examination of curves presented by Grigal and McCall suggests, however, that a mid-summer decrease in decomposition rates may have occurred in 1971. Sampling in 1972 and 1973 provided too few data to examine the hypothesis that decomposition slows during mid-summer, but depending upon moisture conditions it seems likely that the pattern observed in the current study recurs annually.

Grigal and McColl (1977) report decomposition rates that are lower than those reported here. Rates were .39 percent per week for 1971 and 1972-series litter bags and .94 percent per week for 1973-series bags. The inclusion of two

over-winter periods for the first two series may account for decreased weekly rates, although the data of Grigal and McColl suggest that as much as 25 percent of the original material is lost during the first six-month overwintering period. Gosz et al. (1973) report that the leaves of some species are subjected to significant leaching losses during the first month following leaf fall, and it is possible that much of the early loss reported by Grigal and McColl (1977) represents leaching. Graphs of their data suggest the losses from one-year old leaves may be only 10% between October 1 and June 1.

In addition to the weekly weight-loss parameters discussed earlier, several parameters have been proposed for expressing rates of decay. Gosz et al. (1973) discuss the use of a constant (k), which has been proposed by Olson (1963) for situations where loss occurs with no addition of material (e.g. litter bags). Values can be derived from equation 2:

$$k = \frac{X}{X_0} = e^{-kt} \qquad (2)$$

where X is the weight of litter remaining at time t, X_0 is the original weight, and e is the base of natural logarithms. K is referred to as the "decomposition constant". According to Gosz et al., the amount of time required for one-half of the litter in a sample to decompose is .693/k years, and 95% of the litter will decompose in 3/k years.

The above parameters for our data have been computed and are presented in Table 5. The data show that it takes less than one year for half of the aspen litter at plot G12 to decompose. Half lives for litter at other plots are all one year or more. These data are comparable to those of Grigal and McColl (1977) with the exception of their 1972-series samples collected at the end of two years. D.F. Grigal (conversation, 1978) has indicated that 1972 values may be anomalous, however. The rapid decomposition rates for aster illustrate the fact that not all species react the same. This is

also illustrated by Hubbard Brook data (Gosz et al. 1973).

The time required to lost 95% of the litter ranges from a minimum of three years to a maximum of 5.7 years for RCNS plots. Values computed for data of Grigal and McColl (1977) are slightly higher than the RCNS values. Differences may reflect differences in quality of litter and/or different climatic regimes during the sampling period.

Variations in forest-floor weights--Forest-floor weights for the six sample plots are included in Table 1. Although values represent the average of 12 samples, intra-plot variability is high.

Variability among plots also is high. Plots ranked by forest floor weight are G33> G09> G32> G07> G10> G12. Weights at plots G33, G09 and G32 are two times higher than typical values reported elsewhere in Minnesota (Kittredge 1948, Tappeiner and Alm 1975). The value for plot G12 is, by contrast, less than half that reported elsewhere.

The forest-floor weights reported here may, however, represent the actual range of values within aspen stands in the Study Area. The poorly developed forest floor at plot G12 contains only litter. No humus or fermentation layer is present, and depths average less than 1 cm. Soils at plot G33 on the other hand, contain a well developed humus layer, and the forest floor is nearly 5 cm thick.

Two factors may account for the observed variability. Figure 4 shows the relationship between forest-floor weight and total overstory basal area. For plots north of the Laurentian Divide there appears to be a clear relationship between the two parameters. The data suggest that as basal area increases, canopy development is greater and more litter is produced. If

decomposition rates are similar among stands, those stands that produce more leaf litter are likely to have a better developed forest floor. Age of the overstory is also an important consideration, for older stands have more time to accumulate litter. Although it cannot be assumed that no forest floor exists at the time of stand initiation, most of the aspen stands sampled probably originated following disturbances that reduced the forest floor to some degree. In contrast to the northern stands, those south of the Laurentian Divide show no relationship between forest-floor development and overstory development. Little forest floor exists on either site, and the older of the two (G12) has the least well developed forest floor of all six sites. One might conclude, therefore, that decomposition proceeds more rapidly in the southern stands. Figure 3 shows that this may be true. Certainly decomposition was greatest at plot G12. At plot G10 decomposition occurred more rapidly than at some northern sites but was not as fast as at plot G33.

An alternate hypothesis to explain the poorly developed forest floors of stands in the southern part of the Study Area has been posed by D.H. Prettyman, Forest Soil Scientist, Superior National Forest. He suggests that the Palo-Markham-Aurora fire, which burned the area in 1936, may have consumed all of the forest floor and that sufficient time has not elapsed for redevelopment. Prettyman has observed that tree productivity in this area is less than could be expected and suggests that the 1936 fire may be responsible. Prettyman hypothesizes that the fire burned so intensely that the forest floor was destroyed and the nutrients that had formerly been held near the surface were volatilized (in the case of nitrogen) or leached from the soil.

Although Prettyman's hypothesis may apply to some stands in the area, it does not explain the variability we observed. Plot G10 lies within the area burned by the 1936 fire, but plot G12 is east of the fire's easternmost boundary. Plot G12 has, however, low aspen productivity (site index = 46 versus 62 at plot G33), and it is possible that the low tree productivity and high litter decomposition rates are related. Heterotrophic decomposing bacteria immobilize large amounts of nitrogen. Bacteria utilize carbon as an energy source, and, because litter generally contains less nitrogen than carbon (C:N ratios of 30:1 or more are typical), heterotrophs will immobilize mort of the nitrogen in litter until C:N ratios of the litter approach those of microbial organic matter (~10:1). Quantities of nitrogen are made available for higher plant growth only after microbes have expended their energy supply. Thus, in a situation that favors rapid decomposition, nitrogen may limit the development of higher plants.

Such a situation may exist at plot G12. As has already been noted, soils of the Toimi Drumlin Field are fine textured and the topography is flat. These factors are combined with the development of a fragipan (indurated horizon) in most of the soils. As a result drainage is poor and water is retained in surface horizons to a greater extent than in the coarser-textured soils of rolling topography that occur in the northern portion of the Study Area. Data from this study emphasize the importance of adequate moisture to microbial development (note the mid-summer decrease in decomposition rates on Figure 3). Although soil-moisture data were not obtained, subjective observations indicated that high soil-moisture conditions exist in the soils of the southern portion of the Study Area. Ponded water was observed in sm all depressions at plot G12 during the spring and fall, and an ordination of plant communities based on synecological coordinates (Sather 1978) shows

that plot G12 is the most moist of the six study plots. It is possible that denitrification may result in direct losses of nitrogen when sites like that at plot G12 are exceedingly moist. Aeration in the soils of plot G12 is probably improved by what appears to be a very active earthworm population. The activities of Oligochaetes may also contribute to high rates of decomposition by breaking down small pieces of leaf litter. Worms could not, however, have been directly involved in the high rates observed with the litter bags, for the mesh size of the bags was sufficiently small to exclude soil mesofauna.

The effects of heavy metals on decomposition--The toxic effect of heavy metals on soil microorganisms has been recognized for quite some time. Most early studies dealt with the use of heavy-metal-containing fungicides, however. Metal salts are among the most important fungicides and investigations into the relative toxicities of different metals date from the 19th century. Horsfall (1956) surveyed existing literature and concluded that fungitoxicity decreased in the following order: Ag> Hg> Cu> Cr> Ni> Pb> Co> Zn> Ca. Somers (1961) examined fungitoxicity with regard to electronegativity for 24 metal cations and her results support the conclusions of Horsfall. Somers observed that the logarithm of metal ion concentration at ED_{50} shows a direct exponential relationship with electronegativity. Somers states that initial uptake of cations is at or near the cell wall and that the "ultimate fungicidal" action may well be caused by secondary reactions following the formation of un-ionized complexes." Proposed mechanisms include the breakdown of membrane permeability and the diffusion of metal ion complexes into cell interiors.

Studies of the effects of heavy metal pollutants on microorganisms in natural ecosystems are more recent. Among the most extensive are those

conducted in Sweden by Rühling and Tyler (1973) who initially examined the effects on litter decomposition of emissions from two metal processing plants in central and southeastern Sweden. Rühling and Tyler found significant negative correlations between total heavy metal concentrations (Zn + Cu + Cd + Ni) and CO_2 evolution rates for partially disintegrated spruce needle litter. They note that heavy metals are particularly important in inhibiting germination of fungal spores. Fungi, because of the relative tolerance to acid conditions, are important in the decomposition of conifer litter, and Rühling and Tyler conclude that even moderate concentrations (30-50 ppm total, dry weight basis) of heavy metals will depress general decomposition in acid forest sites, at least when adequate moisture supplies are present.

Additional studies by Tyler (1975a) showed that there is a synergistic relationship between heavy metals and water availability, for water deficiency was first shown to be a factor limiting decomposition in samples that had high metal contents. Tyler (1975a, b) reports that nitrogen mineralization, measured as the net increase of ammonium, nitrite and nitrate during incubation of undisturbed samples, is significantly affected by heavy metals, especially Cu. As copper concentrations increase in excess of 20 ppm, nitrogen mineralization decreases rapidly. Levels of Cu between 15-30 ppm appear to increase nitrogen mineralization, but Tyler indicates that these findings need to be substantiated by additional studies. He points out, however, that Cu is often deficient in organic soils. Relationships observed for Cu are less clear for Zn, Cd, of Pb, but nitrogen mineralization is measurably disturbed when concentrations Cu and Zn exceed background levels by a factor of three.

Phosphatases are intra- or extra-cellular hydrolytic enzymes that catalyze the splitting of organic phosphorus compounds. The effects of copper and

zinc pollution on soil phosphatase activity has also been studied by Tyler (1975a, 1976a), who found that these processes decreased as metal concentrations (especially of Cu) increased. Significant decreases were noted at copper concentrations of 30-200 ppm, although effects were ameliorated in soils of higher pH.

B-glucasidas and urease are hydrolytic enzymes of general occurrence in soil microorganisms. The extent of the activity of these enzymes reflects the ability of soil organisms to metabolize cellulose and other polysaccharids in the case of the former and urea in the case of the latter. Tyler (1975a) found the activities of these enzymes to be less sensitive to heavy metals than phosphatases, but Tyler (1975a) did find a significant positive relationship between lead content and B-glucosidase activity. Neither Cu nor Zn significantly reduced the activity of this enzyme, even in high concentrations. Urease activity was, however, reduced by very high metal concentrations. Cadmium appeared most effective in reducing urease activity. In another study in southern Sweden, Tyler (1976b) observed that soil phosphatase activity was depressed by concentrations of vanadium that exceed 100 mg/kg dry weight of soil. Vanadium is a potential pollutant in areas where large amounts of fuel oil are burned.

In conjunction with his decomposition studies, Tyler (1975a) observed decreased tree growth in areas subjected to heavy metals pollution. Depressed growth was observed in areas where unusually large amounts of litter were accumulating on the forest floor. Tyler concludes that the biological activity of the soil decreases with increasing heavy metal content and that this may occur before litter production and apparent vitality of coniferous trees declines. In another paper, Tyler (1972) suggests that decreased litter decomposition may adversely affect tree productivity before the trees

themselves are subjected to phytotoxic levels of air pollutants. As noted in the introduction, decomposition plays a vital role in the cycling of essential elements, and when decomposition slows, nutrients are bound in unavailable forms in the forest floor. Tree growth then declines due to a lack of nutrients rather than the direct effects of air pollutants. Trees experiencing subtle decreases in vigor may also be more susceptible to injury from biotic or abiotic stresses (see Zeyen 1978).

In another field study, Buchauer (197) has studied the effects of zinc-smelter emissions on forest-soil microflora near Palmerton, Pennsylvania in the United States. Buchauer measured zinc concentrations as high as 13.5% by weight in the O2 horizons of soils within 2 km of the smelter site. She notes that 90% of the zinc deposited on the soil surface is retained in the O2 and upper A horizons. Most heavy metals are bound by organic matter and surface accumulations would be expected for metals other than zinc. Using dilution plate counts, Buchauer determined that populations of bacteria and fungi were much lower in O2 and A1 horizons near the smelter but were at normal levels in the A3 horizon. Selection of zinc tolerant varieties of microbes may have occurred at Bucharer's sites, for she observed no decrease in cellulose decomposition despite reduced microbial populations.

William Friedman, a University of Toronto graduate student, is presently completing a Ph. D. dissertation that includes a study of leaf litter decomposition in the Sudbury, Ontario area. This area has been impacted by emissions (including SO_2, Cu and Ni) from a smelting complex that has been in operation for more than 70 years. Friedman used litter bags similar to those employed in this study and examined the decomposition of leaves of Betula papyrifera, Populus tremuloides and Pinus strobus at both impacted and control sites.

His unpublished data are presented in Table 6. Values for k, .693/k and 3/k are recalculated from the original data using equation (2). An examination of the chemical data shows that values for Cu, Ni and, to a lesser extent, $SO_4^=$, fall sharply between 8.0 and 29.7 km from the smelter complex. Values for Mn and Ca generally increase with increasing distance, whereas Mg values are variable. Soil decomposition data show that decay rates increase for all species between 8.0 and 29.7 km. Ninety-five percent loss times for aspen leaves near the smelter are at least 1.5 times those reported in this study and by Grigal and McColl (1977). At a distance from the smelter, however, values are close to those of Grigal and McColl and only slightly higher than reported here. Values for Cu and Ni in the forest floor at distances of 29.7 and 34.3 km, although a fraction of levels near the smelter, are ten times higher than levels determined for soils of northeastern Minnesota (RCNS, 1978). These comparisons suggest that even the remote sites of Sudbury have been impacted by pollutants. The somewhat lower decay rates at Sudbury compared to northeastern Minnesota may reflect the effects of low levels of heavy metals pollution on decomposition.

In addition to field investigations, several studies have used bioassay techniques to examine the effects of heavy metals on soil microorganisms. Liang and Tabatabi (1977) studied the effects of trace elements on nitrogen mineralization in soils of Iowa. The four soils that were selected ranged in pH from 5.8-7.8. Organic matter content of soils were 2.58-5.45 percent and a range of textures were represented (23-45% clay, 39-45% silt, and 1-38% sand). Liang and Tabatabi tested nineteen elements using 5 μ mole/g of soil. They found that Ag(1) and Hg (II) were most effective in nitrogen-mineralization inhibition, whereas As (III), Se(IV), As(V) and W(VI) were least effective. Other elements that inhibited mineralization were Cu(II),

Cd(II), Pb(II), Mn(II). Fe(II), Zn(II), Sn(II), Cr(III), Fe(III), Al(III), B(III), V(IV) and Mo(VI). Elements that inhibited nitrification were Ag(I), Hg(II), Cd(II), Ni(II), Cr(III), Fe(III), Al(III), B(III) and Se(IV). As a result of decreased nitrification NH_4^+ accumulated in some of the soils. Liang and Tabatabi found that the effect of trace elements varies with soil physical and chemical properties. Generally, the most acid, coarser-textured soils showed the greatest inhibition.

In England, Bhuiya and Cornfield (1972) studied the effects of 1000 ppm of Cu, Ni, Pb and Zn on CO_2 release from a slightly acid (pH 6.0) sandy (82.5%) soil. Soil samples were incubated with and without added straw (0.5%) for 12 weeks at 30^o C. The results of this study indicate that the level of added metals depressed CO_2 release from soils without straw. By contrast, soils with straw showed decreased CO_2 evolution for Pb, Cu and Ni, but not for Zn. Mineralization of straw-C was decreased considerably by Cu and Ni and increased by Pb and Zn. Tests showed that neither heavy metals effects on nitrogen mineralization nor pH changes associated with metals additions were responsible for alterations in CO_2 evolution. Bhuiya and Cornfield concluded that changes were directly related to metals additions. They state that the toxic effects of high element concentrations on the activity of heterotrophic organisms are probably due to the metals' ability to compete with essential elements (Mn, Fe, Mg) for active sites (-SH, $-NH_2$, +NH) of enzymes. Addition of organic matter (straw) had no effect on available zinc levels and decreased available lead levels. but decreased the toxic effects of both metals. Bhuiya and Cornfield reason that less easily decomposable straw fractions (lignin and modified protein-lignin associations) may fix the metals in forms less available to microorganisms. Addition of straw increased the levels of "available" Cu and Ni and these increases were

reflected in depressed CO_2 evolution from the soil as a whole. Decreases were due to decreased mineralization of both native organic matter and added straw.

Table 7 is taken from Bhuiya and Cornfield (1972) and shows the effects of the addition of 1000 ppm of each metal and of straw on metal availability. The data show that much greater proportions of Cu and Ni compared to Pb and Zn were initially bound by the soil. The control nickel value in Table 7 is slightly lower and the copper value much lower (1.8 vs. 20-30 ppm) compared to soils of northeastern Minnesota, whereas lead and zinc values are slightly higher in the English soil.

Bond et al. (1976) used microcosms to study the effects of cadmium on soil respiration. They found that oxygen consumption was stimulated by additions of .01 ppm Cd and, initially, by additions of 10 ppm Cd. The production of both O_2 and CO_2 was reduced by 40% during latter stages of the 10 ppm addition experiment. The authors found no changes in organism density caused by cadmium addition and conclude that Cd initially affects respiration, possibly by uncoupling respiratory phosphorylation. They suggest that experiments longer than four weeks may be necessary to detect population density changes. In a second study, Lighthart et al. (1977) tested the effects of "high naturally found concentrations" of several metals on soil respiration. They report reduced respiration for all metals tested and concluded that similar responses in natural ecosystems could "reduce primary and secondary production of the dependent populations."

CONCLUSIONS

Studies of litter decompososition in the northeastern Minnesota indicate that decay rates for aspen leaves vary. Soil moisture contents are high throughout much of the growing season in some fine-textured soils of level topography, and forest-floor development on these sites is very poor. High decomposition rates were observed at one such site, and it seems likely that rapid litter decomposition is at least partially responsible for the lack of humus buildup in the forest floor. Studies of forest floor and overstory vegetation on coarser-texture soils indicate that there is a direct relationship between forest-floor development and total tree basal area. Decomposition rates on these soils are typical of those reported elsewhere in the region (Grigal and McColl 1977).

The basal area-forest floor weight relationship does not hold for aspen stands on fine textured soils. Although aspen stands on these sites exhibit low productivity and have site indices of < 50, the forest-floor weights are much lower than could be expected. Poor growth of aspen may result from a deficiency in nitrogen, for an active population of heterotrophic decomposers could be expected to immobilize large quantities of this element. Other factors, such as high water tables, could also account for decreased aspen productivity, and further studies are necessary to test the nitrogen-deficiency hypothesis.

Forest-floor materials tend to accumulate heavy metals in areas where external sources (e.g. smelter emissions) exist. Soil materials have an ability to immobilize heavy metals to a certain extent, but above certain levels, which vary depending upon soil physical and chemical characteristics, decomposition rates are reduced. If loading of heavy metals in the Study Area

were to increase, the response of soil decomposition rates to higher concentrations of metals would vary among soils. Small increases in heavy metals might increase decomposition rates, whereas large increases would almost certainly decrease the rate of decomposition. Litter decomposition is important in that it is responsible for the mineralization of macronutrients that are necessary for the growth of higher plants. Impacts of metal-induced decreases in decomposition on ecosystem productivity would most likely first occur where decay rates are low. It is possible that those stands that currently appear to have abnormally high decomposition rates might actually benefit from metal-induced reductions in decay rates. This suggestion is based upon an untested hypothesis, however, and further studies, including microcosm bioassays, need to be conducted.

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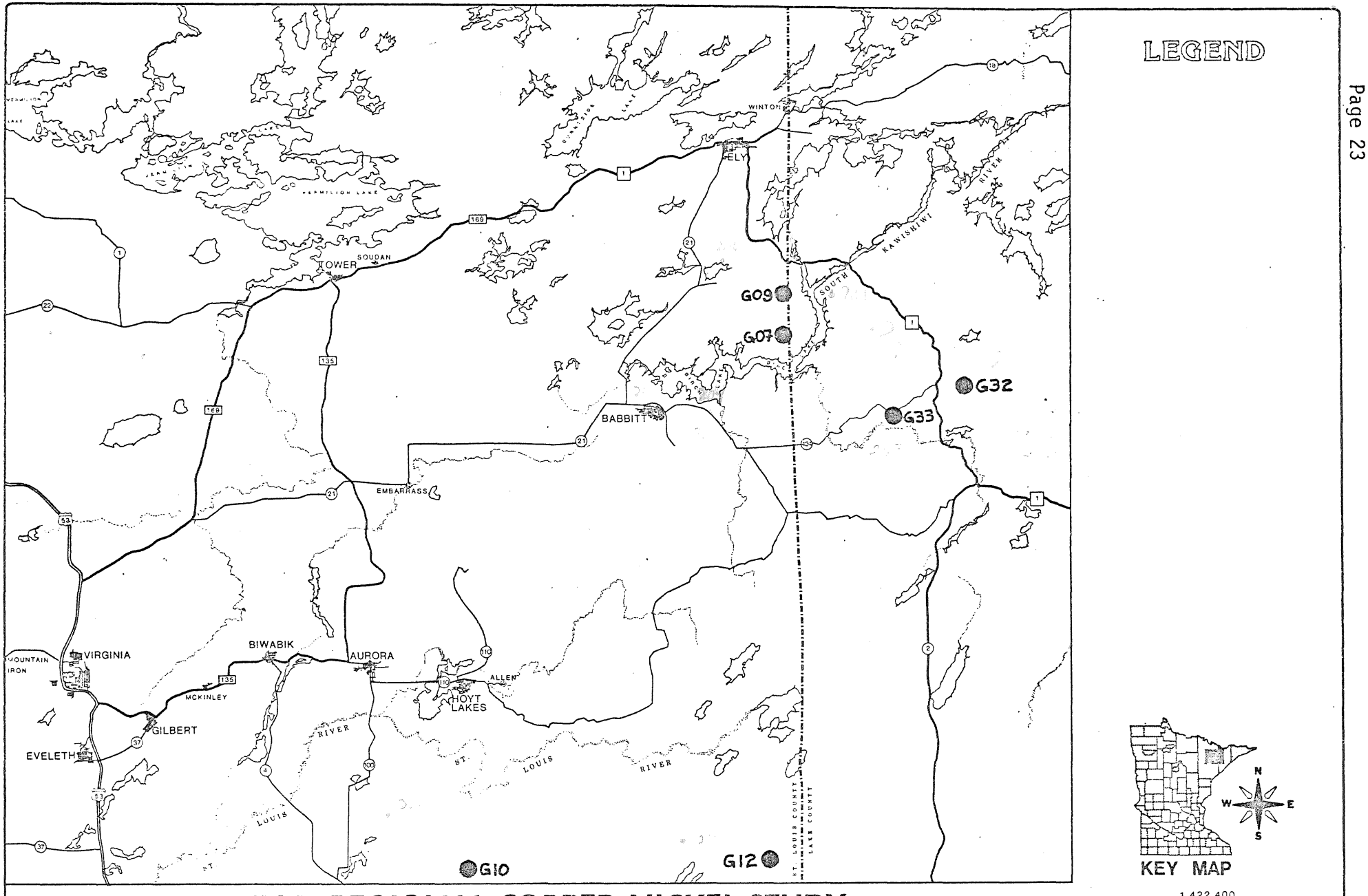


MEQB REGIONAL COPPER-NICKEL STUDY

Figure 1. Map showing the location of study sites.

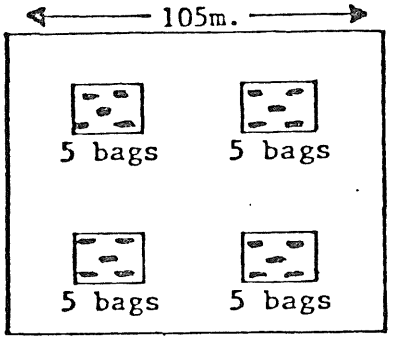
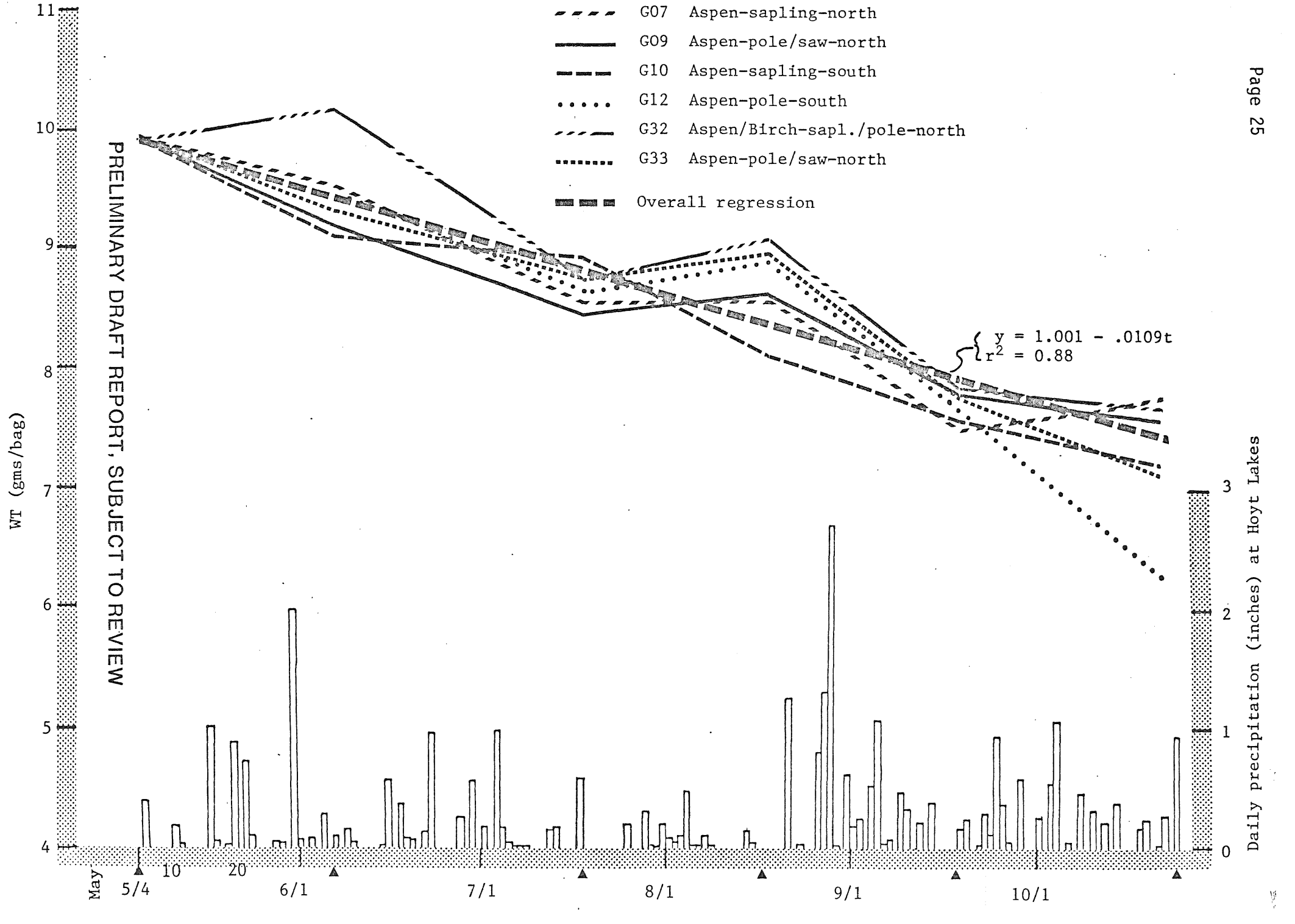


Figure 2. Distribution of litter bags on sample plots.

Figure Precipitation data for Babbitt (bar graph) and litter bag weight and percent weight loss vs. time (line graph).



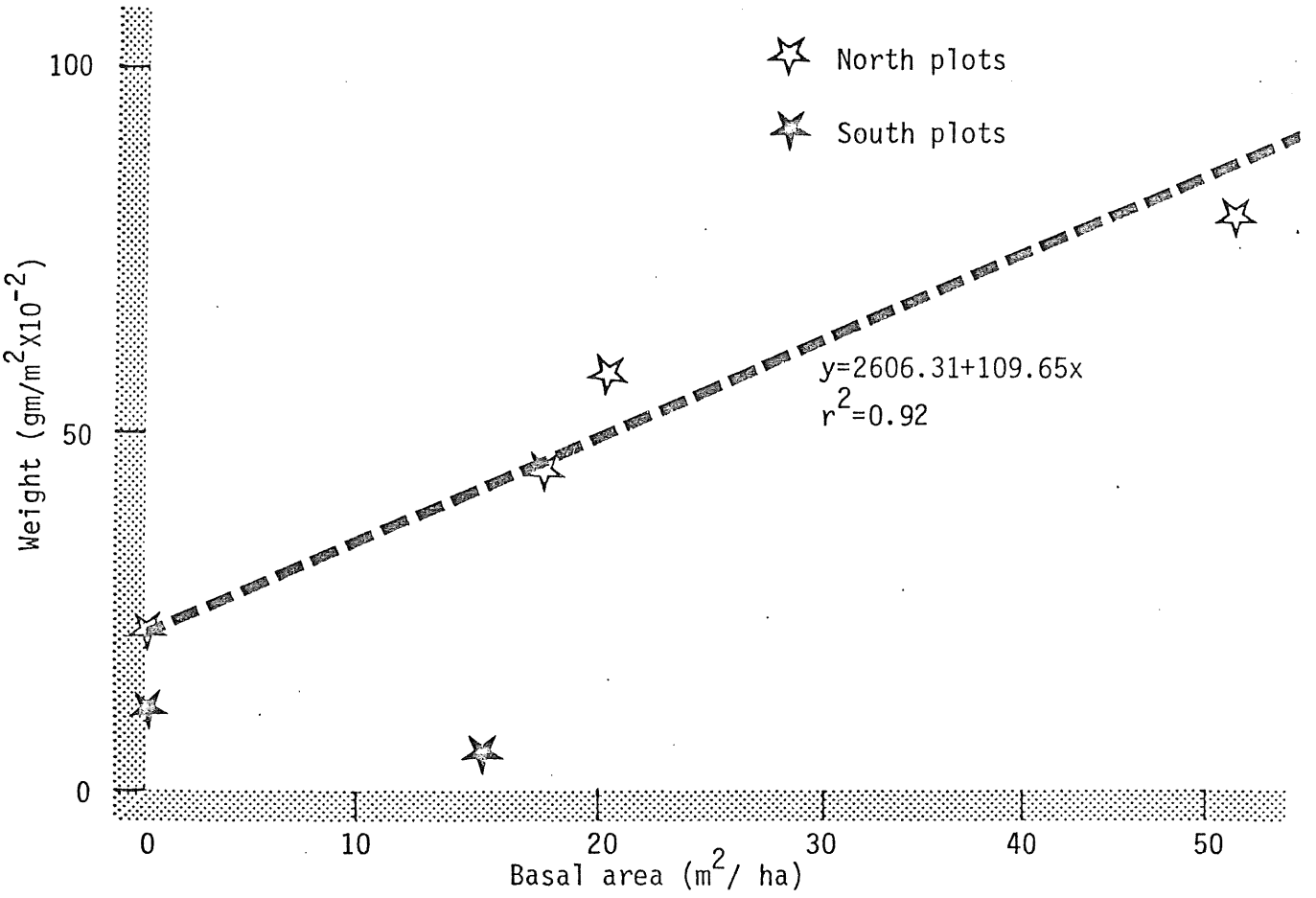


Figure 4. Graph of forest floor weights vs. total overstory basal area. Regression line is for north plots only.

Table 1. Vegetation and Soil Characteristics for Sample Plots

Plot	Location*	Soil Texture	---Forest Floor---		----------Overstory----------		
			Depth (cm)	Weight (g/m^2)	Density (stems/ha)	Cover (m^2/ha)	Canopy Density (% cover)
G07	N	sandy-loam	1.2	2020	0	0	38
G09	N	sandy-loam	5.9	5830	738	20.0	79
G10	S	loam	1.2	1190	0	0	66
G12	S	loam	0.9	581	844	14.4	78
G32	N	sandy-loam	3.0	4470	1480	17.7	87
G33	N	sandy-loam	4.4	7910	1110	51.8	92

* with respect to the Laurentian Divide

Table 2. Average bag weight at the end of each collection period.

	Date	May 4	June 6	July 18	Aug 19	Sept 20	Oct 24
	Days	0	33	75	106	138	172
Site	Week	0	4.71	10.71	15.14	19.71	24.57
G-07		9.98	9.54	8.57	8.58	7.50	7.77
		1.000[a]	.956	.859	.860	.752	.779
G-09		9.98	9.20	8.47	8.61	7.78	7.56
		1.000	.922	.849	.863	.780	.758
G-10		9.98	9.11	8.89	8.11	7.56	7.23
		1.000	.913	.891	.813	.758	.725
G-12		9.98	9.45	8.62	8.91	7.67	6.26
		1.000	.947	.864	.893	.769	.628
G-32		9.98	10.19	8.75	9.08	7.83	7.66
		1.000	1.02	.877	.910	.785	.768
G-33		9.98	9.32	8.66	8.96	7.74	7.20
		1.000	.934	.868	.898	.776	.722

a the second row for each site gives the ratio of $X:X_o$ where X= the weight of leaves at the time of sampling and X_o = the initial bag weight (9.98 g).

Table 3. Results of simple linear regression analyses of percent weight loss versus time.

	a	b	r
G-07	.9925	-.0100	-.955
G-09	.9798	-.0094	-.969
G-10	.9883	-.0110	-.987
G-12	1.0206	-.0137	-.940
G-32	1.0267	-.0107	-.936
G-33	.9988	-.0106	-.956
OVERALL	1.001	-.0109	-.937

Table 4. Results of ANOVA for temporal comparisons.

	May	June	July	August	September	October
May		4*	6	5	6	6
June			4	2	6	6
July				1	4	5
August					5	4
September						1
October						

* No. of sign. differences in mean bag weights.
A maximum of 6 sites were available for each comparison.

Table 5. Litter decay rates calculated for data from this study and recalculated from data of Grigal and McColl (1977). Also included are data from Gosz et al. (1973).

Plot	$\frac{X^1}{X_o}$	K	TIME PARAMETERS (YRS) half-time $\frac{(0.693)}{K}$	95% loss time $\frac{3}{(K)}$
G08	0.779	0.53	1.3	5.7
G09	0.758	0.59	1.2	5.1
G10	0.724	0.68	1.0	4.4
G12	0.627	0.99	0.7	3.0
G32	0.768	0.56	1.2	5.4
G33	0.721	0.69	1.0	4.3
Grigal & McColl (data recalculated)				
Aspen 71 (1)[2]	0.590^2	0.53	1.3	5.7
(2)	0.352^3	0.52	1.3	5.8
Aspen 72 (1)	0.647^2	0.44	1.6	6.8
(2)	0.585^3	0.27	2.6	11.1
Aspen 73 (1)	0.507^2	0.68	1.0	4.4
Aster 72 (1)	0.194^2	1.64	0.4	1.8
Aster 73 (1)	0.136^2	2.00	0.4	1.5
Gosz et al.				
Yellow Birch	---	0.85^2	0.8	3.5
Sugar Maple	---	0.51	1.4	5.9
Beech	---	0.37	1.9	8.1

1 t = .471

2 t = 1

3 t = 2

Table 6. Litter decomposition and forest floor chemical analysis data for the Sudbury, Ontario area (unpublished data from W. Friedman).

Site	Distance[1] (km)	Betula papyrifera $\frac{x^2}{x_o}$	K	$\frac{1}{2}$ time (years)	95% loss (years)	Populus tremuloides $\frac{x}{x_o}$	K	$\frac{1}{2}$ time (years)	95% loss (years)	Pinus strobus $\frac{x}{x_o}$	K	$\frac{1}{2}$ time (years)	95% loss (years)
L1	3.5	.37	.75	.92	4.00	.61	.37	1.87	8.11	.60	.38	1.82	7.89
L2	5.8	.43	.63	1.10	4.76	.64	.33	2.10	9.09	.62	.36	1.93	8.33
BP	8.0	.42	.65	1.07	4.62	.64	.33	2.10	9.09	.60	.38	1.82	7.89
L7	29.7	.33	.83	.85	3.66	.54	.46	1.51	6.52	.55	.45	1.54	6.67
L8	34.3	.28	.95	.73	3.16	.53	.48	1.44	6.25	.50	.52	1.33	5.77

Table 6. continued

FOREST FLOOR ELEMENTAL CONTENT[3]
ppm

SITE	Cu	Ni	Mn	Mg	Ca	$SO_4^=$
L1	2700±800[4]	2100±1300	170± 55	2200±400	2100±1300	210±20
L2	2200±600	1800± 802	200± 70	1200±600	1800± 800	100±30
BP	1650±250	1100± 200	280± 40	770±220	2000± 300	125±30
L7	400±400	300± 270	650±300	1370±300	12000±5000	70±25
L8	200± 80	160± 40	700±270	560±100	6200±1300	35±15

1 from smelter complex

2 bags containing fresh leaves set out June 22, 1976, collected October 22, 1977.

3 ammonium acetate extract

4 $\bar{x} \pm$ s.d. (n = 6)

Table 7. The effects of the addition of 100 ppm of heavy metals and of straw on EDTA-extractable metals (data from Bhuiya and Cornfield 1972).

Sample	Cu	Ni	Pb	Zn
		ppm		
Control	1.8	1.6	18.8	10.0
Soil & Metal	227	253	709	668
Soil & Metal & Straw	373	549	638	673