# ASSESSING FOREST SOIL DISTURBANCE THROUGH BIOGENIC GAS FLUXES

by

CORNELIUS J. DULOHERY III

B.S.F.R., University of Georgia, 1989

A Thesis Submitted to the Graduate Faculty
of The University of Georgia in Partial Fulfillment
of the
Requirements for the Degree
MASTER OF SCIENCE

ATHENS, GEORGIA
1993

ASSESSING FOREST SOIL DISTURBANCE THROUGH

BIOGENIC GAS FLUXES

by

CORNELIUS J. DULOHERY III

Approved:

Lauren A. Mowi

Major Professor

November 22, 1993

Date

Approved:

Indira L. Patel

Dean of the Graduate School

November 22, 1993

Date

## ACKNOWLEDGMENTS

Many people deserve my thanks. Dr. Larry Morris, my committee chairman, administered confidence and direction whenever needed. Lee Ogden worked long hours under hard field conditions and contributed many lucid insights. Each of my committee members (Dr. Larry Morris, Dr. Peter Dress, Dr. Wade Nutter, and Dr. Richard Lowrance) has contributed substantially to the betterment of my degree program and this thesis.

Without funding from the USDA Forest Service, this research would not have been possible. Dr. Bill McKee of the Forest Service was fully supportive. Special thanks go also to Martha Cambell, who manages the UGA forestry school's service laboratory and Alan Thorn of New Zealand's Forest Research Institute. Thanks are also extended to Mark Tippett of the Virginia Polytechnic Institute for much of original plot location and layout, upon which I relied.

The encouragement and support of my friends and family was invaluable; and I dedicate this thesis to the newest member of my family, Hannah Elizabeth Burnsed.

TABLE OF CONTENTS

## LIST OF TABLES

DISCUSSION

## LIST OF FIGURES

## INTRODUCTION AND LITERATURE REVIEW

After Hurricane Hugo struck the South Carolina coast in the Fall of 1989, vast tracts of damaged timber were salvaged from pine flats of the Francis Marion National Forest. To permit retrieval of rapidly deteriorating wood, prohibitions against wet-season logging were waived, and wetted soils were subjected to widespread disturbance by felling and skidding operations. To guide future harvests, the U.S. Forest Service has expressed an interest in better understanding the nature and implications of extensive soil damage, as well as techniques for ameliorating such damage.

Several studies have been undertaken to determine the effectiveness of tillage and fertilization in ameliorating harvest damage (McKee et al., 1991; Tippet, 1992). These studies focus upon soil physical properties, hydrologic condition, and the success of commercial regeneration. The focus of this study is to determine the sensitivity of biogenic gas fluxes to harvest damage and mitigation treatments in wet soils.

### Mechanized Harvesting System

#### Description of Harvesting System and Machinery

Analysis of soil disturbance begins with an understanding of how it occurs. Three pieces of equipment are typically employed in harvesting pines from Coastal-Plain flatwoods: 1) a feller-buncher, 2) a grapple skidder, and 3) a loader. All feller-bunchers and most skidders used in this region are equipped with rubber tires, rather than tracks, for higher speed and maneuverability.

A feller-buncher grasps a tree at its base, while the tree is either sheared with a hydraulic blade or cut with a rotary saw several inches above the ground. Several trees (depending upon their size) can be collected and held vertically by

the feller-buncher, which then stacks them nearby to be transported to a "log deck" by the skidder. The skidder grasps the bases of several trees at once with a hinged grapple and drags them along the ground, with their crowns scraping over the soil surface. Before reaching the log deck, the tree crowns are backed through a coarse metal grate (gated) to remove branches. The bare stems are then dragged to a log deck located next to a road and loaded onto logging trucks by a small portable crane (the loader).

This harvesting system results in a system of primary and secondary "skid trails" radiating from the log deck into the harvested area. Primary skid trails bear the heaviest traffic (many passes) and usually lead directly to the log deck. Secondary skid trails serve as "feeder lines" between lightly trafficked areas in the stand and primary skid trails and still sustain multiple passes. Other areas are trafficked only once or twice during timber cutting, if at all. The timber-salvage operations following Hurricane Hugo were similar to a normal harvest.

The extensive use of rubber-tired rather than tracked vehicles is an innovation of the past 20-30 years that has probably heightened the potential for soil damage during harvests. Reaves and Cooper (1960) demonstrated that the pressure exerted on the soil surface by rubber-tired vehicles and depth to which it is transmitted is greater than that of tracked vehicles because the weight of a rubber-tired vehicle is concentrated on smaller surface area. However, Burger et al. (1983) found that "despite a three-fold increase in contact pressure, changes in soil density and porosity caused by an unloaded, rubber-tired log skidder did not exceed those caused by a crawler." The similar effects of the two machines were attributed to unidentified machine characteristics. Vibration, for example, is known to exacerbate compaction (Reaves and Cooper, 1960). Chancellor (1977) explained that calculations of the average pressure exerted by tracked vehicles can be misleading, because there is often a peak in pressure 2-3 times the average

behind the center of the vehicle, which intensifies when the tractor is pulling a heavy load.

Rubber-tired vehicles have undoubtedly heightened the potential for soil damage by allowing increased access to wet sites. Campbell et al. (1973) state, "Up to 18 inches of clearance permits them to maneuver freely over such common logging obstacles as stumps, slash, rocks, and deeply rutted trails. Four-wheel drive traction systems, no-spin differentials, and balanced weight distribution enable the machines to operate under very adverse conditions, such as steep slopes and wet soils."

Vehicle impacts can be mitigated technologically. Because the average pressure exerted on the soil surface by a pneumatic tire is approximately equal to the tire inflation pressure (Chancellor, 1977), wide tires with low inflation pressures are often used on compaction-prone wet soils. With proper use, wide- or "flotation-"tired skidders reduce soil impacts. However, they are frequently used to gain access to prohibitively wet sites, resulting in severe rutting and compaction. Such detrimental use of wide-tired skidders occurred on the Francis Marion National Forest during post-Hurricane Hugo timber salvage (Aust et al., 1991; Aust et al., 1993). The increased risk of soil damage associated with rubber-tired vehicles appears to be a function of both the high pressures they exert on the soil surface and the increased access they afford to sensitive wetland sites.

**Aerial Extent of Resultant Skid Trails**

A ground-based, mechanized harvest operation results in skid trails occupying 18 to 54 % of a site (Dryrness, 1965; Hatchell et al., 1970; Campbell et al., 1973; Froelich, 1976; Karr et al., 1987; Aust, 1990). With multiple entries into a stand during a single rotation, or over the course of several rotations, the majority of a site can easily be impacted. Thus, skid trails have the potential to affect the productivity of entire stands or intensively managed plantations.

On nine logging areas of the South Carolina and Virginia coastal plains, Hatchell et. al. (1970) found that primary skid trails occupied 12.4 (3.2 to 22.8) % of the total forested area; secondary skid trails occupied 19.9 (8.8 to 42.3) %; and log decks occupied 1.5 (0.3 to 4.6) %. The wetter sites were marked by above average disturbance, which was attributed to loggers abandoning some trails after they had become excessively rutted. Karr et. al. (1987) found little difference in aerial extent of soil damage among three thinning systems, and no difference in total area of disturbance between wet and dry logging conditions. However, rutting was more extensive on wet sites, where 13-16 % of the area was occupied by ruts 1-3 inches deep and 8-24 % was occupied by ruts 3 inches or deeper.

On a four-hectare site in the Francis Marion National Forest salvaged after Hurricane Hugo, Aust et al. (1993) classified 14 % as slightly disturbed (subjected to 1-2 skidder passes with litter still in place) and 34 % as rutted up to 15.2 cm in primary and secondary skid trails. This percentage matched closely the average value for primary and secondary skid trails on Coastal Plain logging sites (35 %) reported by Hatchell et al. (1970), and was within the range for rutting of wet sites reported by Karr et al. (1987).

The estimated extent of disturbance can be limited by the classification scheme used. Martin (1988) noted that most earlier studies looked only for well-defined skid trails and predated the advent of mechanical fellers, which impact extensive areas while cutting trees. In an unusually detailed disturbance classification, he estimated that 48 to 81 % of soils on whole-tree harvested New England sites had received some compaction and that 71 to 93 % had been disturbed in some way. An unknown portion of the disturbance that he measured may have been of little long-term significance.

## Soil Disturbance Associated with Skid Trails

Three forms of soil disturbance are associated with skid trails: 1) compaction, 2) rutting, and 3) displacement of topsoil. Each has been implicated in reduced soil productivity.

### Compaction

Compaction is a reduction in the percent of total soil volume occupied by pores, or conversely, an increase in bulk density. Compaction can reduce above- and below-ground tree growth, limit water and air movement, and fundamentally alter soil microbial activity.

Roots of a wide range of plant species are incapable of penetrating soils above critical bulk densities, which range from 1.46 to 1.9 g $cm^{-3}$ (Veihmeyer and Hendrickson, 1948; Daddow and Warrington, 1983). Critical bulk densities are higher in coarse-textured soils. In a greenhouse experiment, Foil and Ralston (1967) reported a negative linear correlation between both root weight and penetration in loblolly pines and soil bulk density, which ranged from 0.8 to 1.4 g $cm^{-3}$. When root penetration is limited, so is a plant's ability to exploit water and nutrients in the soil (Chancellor, 1977). In the Foil and Ralston (1967) study, above-ground growth of the seedlings in soils initially subjected to 3.5 and 10.5 kg $cm^{-2}$ of surface pressure was greatly reduced versus those grown in soils to which no pressure was applied.

Compaction reduces air-filled porosity by reducing the macropore space, from which water freely drains (Grable and Siemer, 1968); i.e.., when a greater fraction of pore volume is in micropores, more water is retained. Disproportionate reductions in macroporosity due to skid-trail damage are common (Dickerson, 1976; Karr et al., 1987; Tippet, 1992; Aust, 1993). Below 10-15 % air-filled porosity, most soils and unconsolidated sands are virtually impermeable to gases (Blake and Page, 1948; Wyckoff and Botset, 1936).

At approximately the same level (5-15 %), plant growth and yield of a variety of annual agricultural crops suffer appreciable reduction (Vomocil and Flocker, 1961; Grable and Siemer, 1968). In a review, Vomocil and Flocker (1961) concluded that a portion of air space in a moist soil is not available as diffusion pathways, and that 10 % air-filled porosity is a dangerously low level for plant growth. Diffusion may be inhibited in wet soils because air spaces are either discontiguous or separated by pores of capillary size, which are effectively impassable. For the same reasons, 10-15 % air-filled porosity may be a critical level for microbial activity, below which metabolism and nutrient cycling are radically altered.

## Rutting

Rutting is among the most radical forms of soil disturbance associated with skid trails, yet literature concerning its effects is scant. Burger (1990) offers the following definition: "Rutting is the surface displacement of soil as it moves laterally and upward from under the tire or track of a machine. Puddling always accompanies rutting and compaction will also when soil moisture is around field capacity." When the soil is saturated, rutting can be severe, but compaction minimized because of the incompressibility of water.

Burger (1990) defines puddling as the shearing or disintegration of soil aggregates, in which soil structure is lost, macropore space collapses, and drainage is greatly reduced. Rutting sometimes impedes lateral flow of water below the soil surface due to the presence of a puddled, and thus impermeable, rut wall that extends vertically into the soil (Aust et al., 1993). In such cases, drainage of an area well beyond the skid trail can be reduced (Hatchell et al., 1970; Aust et al., 1993). Ruts often pond and are invaded by hydrophytic plants. Goncharov and Shein (1991) found that rut formation resulted in a redistribution

of soil moisture, in which evaporation and soil drying in the rut zone resulted in a pressure gradient that caused water movement toward ruts.

Given its known physical effects, rutting is likely to be detrimental to the reestablishment and growth of a site's endemic vegetation and/or commercially desirable tree species. In Foil and Ralston's (1967) greenhouse experiment, pine-seedling establishment in soils that were puddled and compacted was consistently worse than in soils that were compacted alone or the undisturbed control group. Puddling was achieved by kneading the soil at a high moisture content. In the field, poor reestablishment and growth due to rutting are often manifest but have not been well documented. The widely perceived detrimental effects of rutting have led to the promulgation of voluntary best management practices (BMP's) in several southeastern states that limit the length and depth of ruts.

**Displacement of Topsoil**

Displacement of topsoil, as a component of skid-trail damage, has not been well studied, but has been hypothesized to be an important contributor to the deleterious effects of skid trails on tree growth (McKee and Haselton, 1989; McKee, 1990). The soil displacement that occurs during skid-trail formation is similar to that resulting from intensive mechanical site preparation.

Site preparation and soil displacement have little effect on mineral-soil nutrient reserves of Coastal-Plain sites, but can severely diminish forest floor nutrients (Morris and Pritchett, 1982). In a poorly drained Spodosol of the northern Florida Coastal Plain, about 10 % of the site's total nutrient reserves were displaced into windrows during site preparation (Morris et al., 1983). Morris (1981) found that the N, P, K, and Mg contents of the forest floor were 19, 1, 8, and 2 kg $ha^{-1}$, respectively, in intensively prepared plantations, while forest-floor nutrient contents in undisturbed and minimally disturbed areas were at

least 10 times greater. Morris (1981) speculated that the pool of mineralizable N in the forest floor was disproportionately large. The hypothesis that nutrients in the litter layer are more readily mineralizable than those deeper in the soil profile is intuitively appealing, but has not been clearly demonstrated.

In addition to total nutrient inventories, nutrient concentrations in the upper horizons can be diminished. Miller and Sirois (1986) observed displacement of topsoil and nutrients on 8-14 % of cable yarded and skidded sites that resulted in concentrations of P, K, Ca, and Mg considered limiting to the growth of loblolly pines. However, Burger (1979) and Morris (1981) found that displacement of litter, woody debris, and soil by windrowing on northern Florida pine sites did not reduce extractable P, K, Ca, and Mg.

In the Morris (1981) study, soil temperature was higher and extremes of soil moisture more severe on the intensively managed sites, suggesting the importance of an intact forest floor in moderating microsite conditions. Similar moderation of soil temperature and moisture by the forest floor has been observed by other investigators (e.g., Donnelly and Shane, 1986 and Skinner et al., 1989).

On a relatively infertile and P-deficient New Zealand soil, the four-year volume growth of *Pinus radiata* seedlings was reduced 25 % relative to controls where litter had been removed by hand; 65 % where litter had been removed by machine (and compaction had occurred); 70 % where topsoil had been removed and the subsoil compacted by two passes of a loader; and 80 % where topsoil had been removed and the subsoil compacted with eight passes of a loader (Skinner et al., 1989). Losses in productivity were attributed to "combinations of nutrient loss through topsoil removal, changes in soil resistance, and less favorable soil temperature regimes, owing to the absence of litter."

## Net Effects of Skid trails on Tree Growth

The complex of effects associated with skid trails results consistently in the inhibition of tree growth. Seedling survival is less often affected; and with minimal disturbance, natural seedling recruitment can actually be enhanced.

Skid trails are seldom detrimental to the early survival of trees, except on fine-textured soils. Hatchell et al. (1970) measured height growth of pines and stocking from natural regeneration on recently formed skid trails in the South-Carolina and Virginia Coastal Plain. After one year, pine stocking from natural recruitment on secondary skid trails was either equal to or greater than the controls. This effect was attributed to scarification and exposure of the mineral soil, which rendered microsites favorable for germination. Stocking was reduced on primary skid trails with fine-textured soils and severely depressed on log decks. Pomeroy (1949) also saw an increase in germination of pine seedlings in scarified areas, but high mortality on fine-textured soils that were puddled. In a study on the Francis Marion National Forest (Hatchell, 1981), survival of planted seedlings during their first four years was unaffected by compaction from logging.

Unlike survival, the growth of trees is frequently reduced in heavily trafficked areas. In the Hatchell et al. (1970) study, one-year height growth averaged 48 % lower on primary skid trails than control areas. On secondary skid trails, only a site with a fine-textured soil exhibited reduced height growth (21 %). Per-hectare biomass of seedlings growing in the middle of skid trails (between the ruts) was 1/3 to 1/2 that of the controls. Biomass of seedlings in the ruts themselves was even lower. In Hatchell's subsequent (1981) study, height growth of seedlings was reduced 18 % in soils compacted during logging.

Thirty two years after logging in the mountains of western Oregon, per-hectare volume of Douglas-fir (*Pseudotsuga menziesii)* was 73 % lower on former

skid trails than in undisturbed areas and 17 % lower in 3-meter strips adjacent to the skid trails, resulting in a volume reduction of 11.8 % for the entire area (Wert and Thomas, 1981). On an Oxisol in the Soloman Islands, the basal area of *Gmelina-aborea* and *Terminalia-brassii* seedlings planted in areas heavily trafficked by a tractor crawler was about half that of seedlings planted in minimally disturbed areas (Cheatle, 1989). Moehring and Rawls (1970) showed that heavy wet-weather traffic (on three to four sides of a tree) during thinning reduced growth for at least five years. Lighter traffic in wet weather (confined to one or two sides of a tree) and dry-weather logging did not affect tree growth.

Weather can play a serendipitous role in mitigating the effects of soil disturbance. On a clayey Piedmont Ultisol, Campbell et al. (1973) saw no difference in the one-year survival or growth of seedlings planted in areas disturbed by tree-length skidding and seedlings planted in undisturbed control areas—despite significantly higher bulk density and lower porosity on the skid trails. They conjectured that the planting method, which loosened the soil around the roots, and abundant rainfall during the first growing season minimized compaction effects.

Most studies have focused upon early survival and growth, but Perry (1964) addressed long-term impacts by examining tree growth on abandoned Piedmont log roads and adjacent undisturbed areas 26 years after logging. Tree volume was 53 % lower on the abandoned log roads, which should be regarded as representative of extreme disturbance.

**Persistence of Physical Effects**

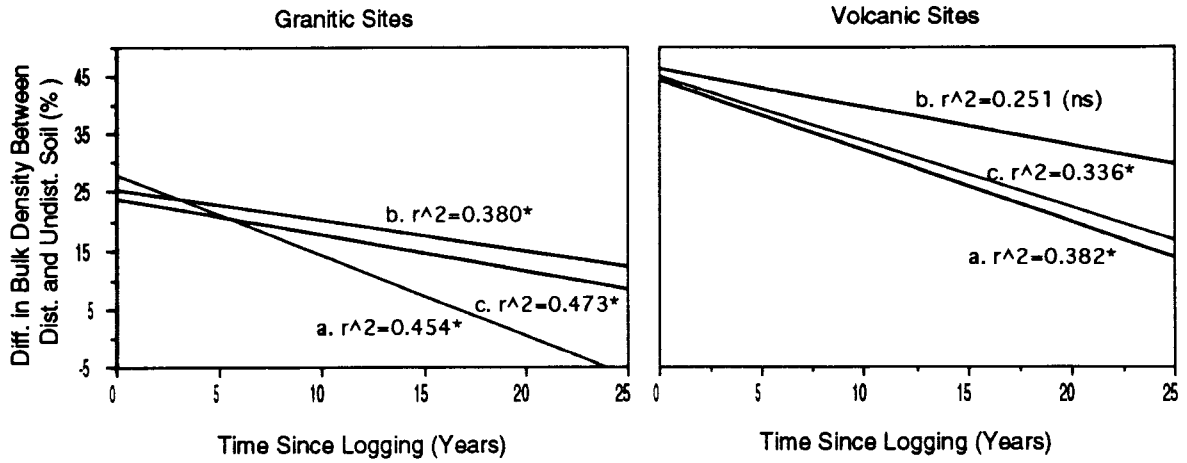
Soil compaction associated with skid trails can persist for 40 or more years after their formation. Review of the literature suggests that bulk density recovers more slowly if the compaction is initially severe, the soil is fine textured, and/or the compaction occurs deep in the profile.

In several studies, bulk density was measured on skid trails of different ages to develop regressions of compaction versus time (Figure 1). Estimates for recovery at the surface soils in the southeastern Coastal-Plain ranged from 8 years in the area between ruts where logs are dragged (Dickerson, 1976) to about 18 years on log decks (Hatchell and Ralston, 1971). In these cases, skid-trail damage persisted well into—and certainly throughout the establishment phase of— the subsequent rotation. Most Coastal-Plain studies have neglected subsurface compaction, which may be even more persistent.

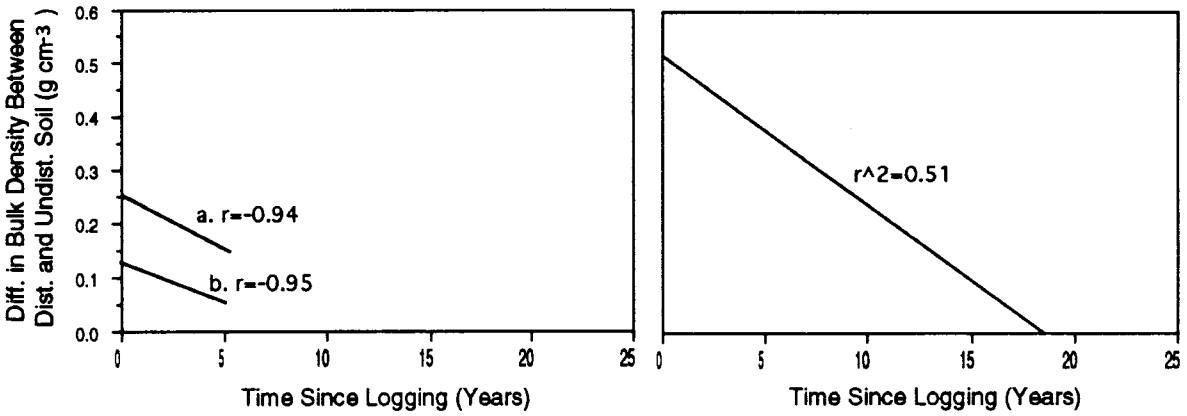
Froelich et al. (1985) measured bulk density at 3 depths on former skid trails in west central Idaho. Granitic and volcanic soils were included in the study. On average, surface bulk density of former skid trails on granitic soils was equal to that of undisturbed soils after 23 years, but was still elevated in volcanic soils. Extrapolation of the linear equation for volcanic soils suggests that they would recover fully in approximately 38 years. (Intercept values were not reported by Froelich et al. (1985) but were calculated for this review.) Regression equations for the volcanic and granitic surface horizons, however, were not significantly different.

In both granitic and volcanic soils, compaction deeper in the profile (up to 30.5 cm) was more persistent. The slopes of the regression lines were less steep at depth (indicating slower recovery), and subsurface horizons had not recovered after 23 years. Zero-difference time intercepts for subsurface compaction in the Idaho study ranged from 40 to 70 years. Froelich et al. (1985) concluded that subsoils recover more slowly than those at the surface, which was consistent with earlier findings of Thorud and Frissell (1976) and Wert and Thomas (1981).

In a clayey Piedmont soil (Cecil series) of North Carolina, infiltration in log roads abandoned 26 years before was five times slower than in nearby undisturbed soils (Perry, 1964). Perry offered a conservative estimate for full




Recovery of soils in former skid trails in West Central Idaho at depths of a) 5.1 cm, b) 15.2 cm, and c) 30.5 cm. All regressions based on 15 observations. * Indicates significance at 0.05 level. (ns) indicates no significance. (Adapted from Froelich et al., 1985.)




Recovery of soils on former skid trails in the Coastal Plain in a) wheel ruts and b) the area between ruts disturbed by log dragging. Both regressions based on six observations. Significance level, if any, not stated. (Adapted from Dickerson, 1976)

Recovery of soils on log decks in the Virginia Coastal Plain. Regression based on 15 observations and significant at an unstated level. (Adapted from Hatchell and Ralston, 1971)

Figure 1. Bulk density recovery on former skid trails, with regression lines from several studies. Lines shown only for the approximate period represented by the measurements.

recovery of abandoned woods roads of 40 years. Hatchell and Ralston (1971) concurred that this estimate was reasonable for recovery of soils subjected to extreme disturbance.

Sandy soils with light to moderate traffic may recover much more quickly. Mace (1971) observed full recovery in the bulk density of a relatively dry, sandy soil in northern Minnesota one year after tree-length skidding with a rubber-tired skidder. Recovery was attributed to soil freezing and a small degree of initial compaction. Other soils in this study did not recover appreciably in the same period.

## Ameliorating Skid Trail Damage

Studies indicating that soil physical properties and tree growth remain adversely affected well into the subsequent rotation have prompted efforts to develop methods of ameliorating or repairing skid trails after they are formed. Fertilization and tillage, or a combination of the two, have been the most common and effective recommendations.

Hatchell (1981) evaluated several treatment combinations to ameliorate skid-trail damage on the Francis Marion National Forest and concluded that the highest pine yields were obtained through a combination of tillage and fertilization. In his study, biomass of four-year-old pine seedlings was 31 % less on soils compacted by tree-length skidding than on uncompacted soil. Bedding increased biomass yield 240 % over plots that received no mechanical treatment[1], regardless of the soil's previous condition. But residual compaction effects were apparent in the beds installed over skid trails, where growth remained about 33% less than in beds installed on uncompacted soil. Plots that were fertilized in addition to bedding appeared to overcome this residual compaction effect,

---

[1] Herbicide was applied only to plots that received no mechanical treatment; so the meaning of comparisons with mechanically treated plots was unclear, and no true untreated control group was included in the study.

yielding similar biomass on compacted and uncompacted soils. Unfortunately, limitations of the experimental design prevented statistical analysis of the combined effects of bedding and fertilization. A disking treatment alone did not affect survival, growth, or biomass production.

Fertilization produced mixed results in Hatchell's (1981) study, increasing growth of individual seedlings but decreasing overall survival (by stimulating competition). In uncompacted soils, fertilization had little effect on per-acre biomass production. But in compacted soils, despite decreased survival, trees were consistently responsive to fertilization in terms of individual seedling growth and per-acre biomass yield.

A study by Gent et al. (1983) cast doubt on the effectiveness of bedding alone as a mechanical remedy for skid-trail damage. He found that while bedding created a new, elevated soil surface with a bulk density approximately equal to that of an undisturbed soil, compaction of the underlying (original) surface was not corrected. Indeed, it was exacerbated by the mechanical passes required for bedding. In all cases, underlying compaction was in the range generally considered limiting for root growth. Many studies report only on the first few years of growth, but growth may not be affected until seedlings become large enough for the effects of subsurface compaction and a limited root volume to become apparent. Gent et al. (1983) recommended, but was unable to evaluate, disking of skid trails to loosen the soil before bedding.

In New Zealand, a combination of ripping to a depth of 45-60 cm and phosphorus fertilization increased height growth 144 % and 3-year survival 23 % higher on log decks (Berg, 1975). Similar treatments on "hard clay" ridges also produced favorable results and were eventually adopted on a management scale. Deep tillage to repair skid-trail damage in forest soils has not been

experimentally evaluated in the United States but may be effective in countering subsurface compaction.

In a study now underway, McKee et al. (1991) are attempting to clarify the results of earlier research and assess the recommendation of disking before bedding. Several studies, including the present, have been conducted in conjunction with McKee et al. (1991).

## Biogenic Gas Fluxes

### Rationale for Use in Assessing Soil Condition

Biogenic gas fluxes are composite indicators of soil condition that provide a relatively new way of viewing and analyzing soil disturbance. Skid-trail damage has been assessed customarily by physical measures such as bulk density, porosity, and hydraulic conductivity. Survival and growth of planted seedlings are more inclusive composite indicators of soil condition, as they are governed by a host of factors altered during skid-trail formation. The seedling-performance approach yields important information concerning commercial productivity and soil quality but requires years of monitoring. Like seedling growth, soil biological activity is influenced by a broad range of factors, and should be a robust composite indicator of soil condition. Taken together, the monitoring of physical parameters, seedling performance, and soil biological activity constitute a well-rounded approach to soil damage assessment.

Soil biological activity, through the products of respiration, may reveal effects not apparent by other means. For example, fertilization has been proposed as a means of mitigating soil damage (e.g., Hatchell, 1981; McKee et al., 1991). In seedling trials—especially in the early stages of growth—fertilization could compensate for loss of soil quality through increased growth, but without actually restoring "damaged" soil parameters. Soil respiration, however, may be less sensitive to fertilization because microbial carbon mineralization (usually the

dominant source of $CO_2$) is often limited by factors such as available carbon and aeration. Estimates of soil respiration and seedling performance, thus, complement one another by integrating over vastly different time scales and different, though overlapping, sets of factors.

In wet soils, indicators of biological activity other than $CO_2$ evolution may be useful. Oxygen diffuses about 300,000 times slower through a water-saturated soil than an air-dry soil (Stepniewski and Glinski, 1988). Therefore, after an area is saturated, oxygen is quickly depleted from the soil water by aerobic respiration. Organisms must then use compounds other than $O_2$ as terminal electron acceptors in respiration. Nitrous oxide and methane are products of anaerobic microbial respiration at low redox potentials that commonly occur in wet soils (Table 1). The measurement of carbon-dioxide, nitrous-oxide, and methane fluxes, together, provides an assessment over the full range of redox potentials that occur in wetland soils.

Table 1. Oxidized and reduced forms of several elements and approximate redox potentials for transformation

| *Element* | *Oxidized Form* | *Reduced Form* | *Critical Redox Potential for Transformation (mv)* |
|---|---|---|---|
| Nitrogen | $NO_3^-$ (nitrate) | $N_2O$, $N_2$, or $NH_4^+$ | 220 |
| Manganese | $Mn^{+4}$ (manganic) | $Mn^{++}$ (manganous) | 200 |
| Iron | $Fe^{+++}$ (ferric) | $Fe^{++}$ (ferrous) | 120 |
| Sulfur | $SO_4^=$ (sulfate) | $S^=$ (sulfide) | -75 to -150 |
| Carbon | $CO_2$ (carbon dioxide) | $CH_4$ (methane) | -250 to -350 |

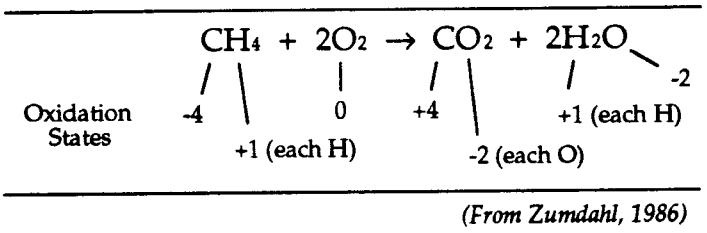
(From Mitsch & Gosselink, 1986)

## Redox Potential and Electron Flow

Oxidation-reduction (or redox) potential is inversely proportional to electron availability in the soil solution (Mitsch and Gosselink, 1986; Paul and Clark, 1989). At low redox potentials, many compounds in the soil have gained electrons (been reduced) and electrons are considered excessively available. This reduced state results primarily from the biological decomposition of organic matter in the absence of oxygen. Redox potential can be measured by inserting an inert platinum electrode into the soil and measuring its electric potential (in millivolts) relative to a reference hydrogen electrode (Mitsch and Gosselink, 1986).

During decomposition, electrons flow from organic molecules to electron acceptors (preferentially oxygen, because of its affinity for electrons). A simple example of this process is methane combustion (Reaction 1), in which eight electrons shift from the carbon in $CH_4$ to the oxygen in $CO_2$ and $H_2O$. The carbons are oxidized (lose electrons) and the oxygens reduced (gain electrons). Energy from such electron flow can be captured by an organism in respiration or simply liberated, as in combustion.

Reaction 1.




*(From Zumdahl, 1986)*

In an aerated soil, the supply of oxygen into the soil and flux of $CO_2$ out occurs through gaseous diffusion and is relatively fast. However gaseous diffusion is reduced dramatically by flooding, and oxygen is depleted because it cannot be replenished quickly enough. In the absence of oxygen, electron flow

shifts to more recalcitrant acceptors such as nitrate and ferric iron (Table 1). If all substances in the soil capable of being reduced are reduced, decomposition would cease; there would be no outlet for electron flow. In reality this rarely—if ever—occurs, because a variety of electron acceptors are slowly replenished through processes such as aqueous diffusion, oxygen leakage from plant roots, and oxidation of substances at aerated microsites in the soil.

Because of the slow replenishment of electron acceptors in wetlands, organic matter decomposition is almost always inhibited, and organic accumulations are greater than in upland ecosystems.

## Sources of Gas Fluxes

**Carbon Dioxide.** Some investigators regard $CO_2$ evolution from the surface as representative of virtually all metabolic activity in the soil (e.g., Lundegardh, 1927; Behera et al., 1990). In aerated soils, where anaerobic respiration is minimal, this generalization may hold. Other routes of metabolite export from uplands such as $CO_2$ dissolved in ground water are thought to be negligible (Schlesinger, 1977).

However, under anaerobic conditions a variety of respiratory pathways result in the production of other metabolites such as low-molecular-weight organic compounds (through fermentation), nitrogenous gases, hydrogen sulfide, and methane (Mitsch and Gosselink, 1986). $CO_2$ also is produced through anaerobic processes, such as those represented in Reactions 2, 3, and 4. In fact, the sulfur-reduction / fermentation pathway, represented in Reactions 3 and 4, accounted for at least half of annual $CO_2$ evolution from the soil of a New England salt marsh (Howes et al., 1984).

Reaction 2. $$C_6H_{12}O_6 + 4NO_3^- \rightarrow 6CO_2 + H_2O + 2N_2$$

*(From Mitsch and Gosselink, 1986)*

Reaction 3.

$$2CH_3CHOHCOO^- + SO_4^= + 3H^+ \rightarrow 2CH_3COO^- + 2CO_2 + 2H_2O + HS^-$$

(lactate)

*(From Valiela, 1984)*

Reaction 4.

$$CH_3COO^- + SO_4^= \rightarrow 2CO_2 + 2H_2O + HS^-$$

(acetate)

*(From Valiela, 1984)*

The summed production of metabolites other than $CO_2$ in wetlands may constitute a significant share of total metabolism, though no such direct measurement or calculation was found in the literature. However, equations provided by Valiela (1984) suggest that—along with $CO_2$—a significant quantity of hydrogen sulfide is produced during sulfur-reduction / fermentation. And in some ecosystems, for example a Michigan Swamp (Baker-Blocker et al., 1977) and a tidal freshwater marsh of Louisiana (Smith et al., 1982), the rate of methane evolution is only one order of magnitude lower than typical rates for $CO_2$. Still, none of the literature appears to challenge the assumption that $CO_2$ evolution from wetland soils represents most of total soil metabolism.

In addition to heterotrophic carbon mineralization, $CO_2$ evolution originates from respiration of living plant roots. No accurate method exists to partition this fraction from heterotrophic respiration, but a variety of indirect methods has been employed to approximate their ranges. In most studies, root respiration of forest soils falls within the range of 20 to 60 % of total soil respiration (Table 2), varying by ecosystem, season, and method of measurement. Heterotrophic respiration is thus in the range of 40-80 %. Microorganisms and/or fungi appear to be the overwhelmingly dominant source of heterotrophic respiration (Figure 2). The contribution of invertebrates such as arthropods, arachnids, and

nematodes has been estimated at 5 to 10 % of heterotrophic respiration (Figure 2) and, thus, an even smaller fraction of total soil respiration.

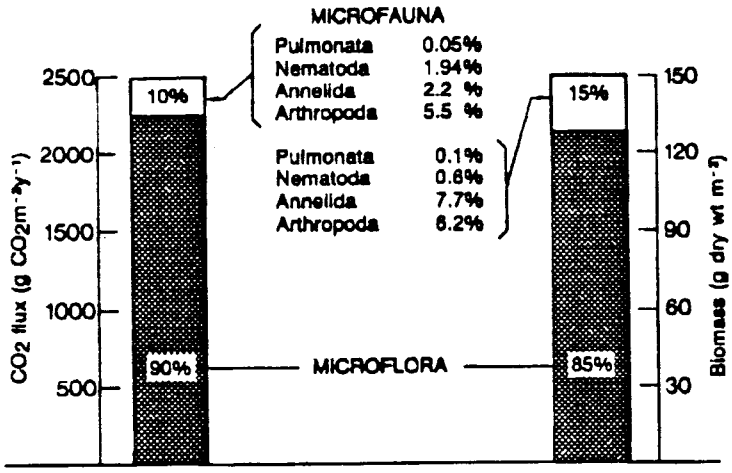
Table 2. Root contribution to total soil respiration reported by different workers

| Source | Habitat | % Respiration |
|---|---|---|
| Boois (1974) | Oak Forest | 40 |
| Edwards and Sollins (1973) | Forest | 22-36 |
| Kira (1978) | Warm temperate oak forest | 50 |
| Lieth and Ouellette (1962) | Abies forest | 30 |
| Nakane and Kira (1978) | Warm temperate oak forest | 50 |
| Nakane (1980) | Primeval moist forest | 50 |
| Nakane et al. (1983) | Red pine forest | 50 |
| Odum and Jordon (1970) | Tropical Rain Forest | 22 |
| Witkamp and Frank (1969) | Pine | 50 |
| Behera et al. (1990) | Deciduous tropical forest | 50.5 |

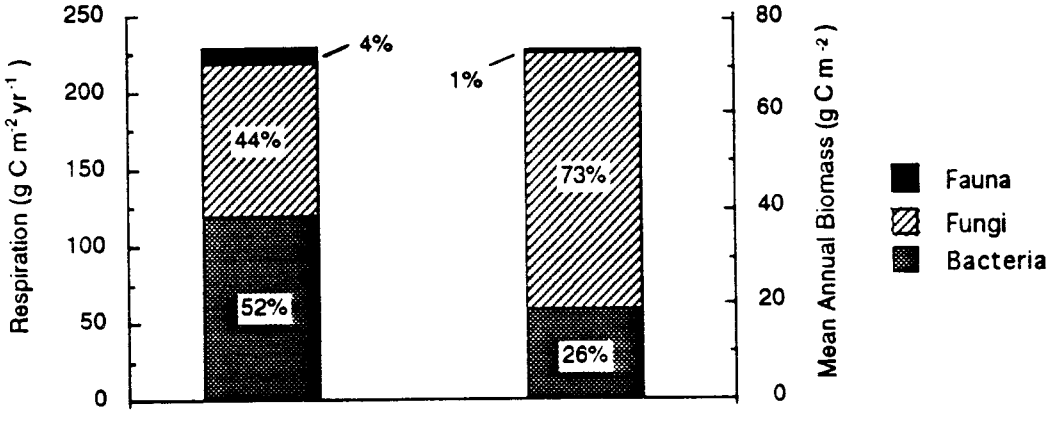
*(From Behera et al., 1990)*

In a northern Florida slash pine plantation, Ewel et al. (1987) estimated the contributions of root respiration to be 51 % in a 9-year-old plantation and 62 % in a 29-year-old plantation. Root respiration was calculated as the difference between total soil respiration and the sum of other contributions (decomposition of litter, fine-roots, and fine particles of organic matter). They obtained a similar result by trenching around small plots to severe root systems and inhibit root respiration. Above-ground regrowth was suppressed. Root respiration was taken to be the difference between $CO_2$ evolution from the trenched plots (minus an estimate of the contribution by decaying roots) and nearby undisturbed areas.

An estimate of the root respiration in an oak forest by Coleman (1973) is notably low at 6 to 11 % of total soil respiration. However, Coleman's methodology was more remote from field conditions than most. Small (2.8 x 5




A. Respiration and biomass the microflora and faunal decomposers in a deciduous forest litter. (From Reichle, 1976)




B. Respiration and biomass of bacteria, fungi, and fauna in a scots pine forest of Sweden. (Adapted from Persson et al., 1980)

Figure 2. Estimated contributions to heterotrophic respiration in a hardwood forest litter (A) and the soil of a pine forest (B).

cm) soil cores were removed from the field. The interrelated components (roots, litter, and soil) were physically disassociated from one and another, placed in jars, and the $CO_2$ evolution from each measured by the alkali absorption method. Respiration of the summed compartments ranged from 89 to 143 % that of the total from intact cores.

Coleman's root-respiration estimates are probably low. Studies by Davidson and Milthorpe (1966) and Frossard (1976) have demonstrated an almost immediate reduction in the root respiration of annual plant species to 50% or less following clipping of foliage and severance of the labile-photosynthate supply to the roots. The reduction in root respiration due to clipping is similar in magnitude to that resulting from prolonged darkness (Frossard, 1976). A consistent maintenance level of root respiration can be maintained for days without photosynthesis (Davidson and Milthorpe, 1966; Frossard, 1976). Thus, while physical separation of the respiratory compartments in the laboratory is useful as an exploratory tool, extrapolation to the field level is questionable.

All methods of partitioning $CO_2$ fluxes have involved radical manipulation of the soil and/or a number of tenuous assumptions. The weight of literature suggests that root respiration in forests is commonly 30 to 50 % of total annual soil respiration and that the remaining heterotrophic contribution is dominated by microorganisms and fungi.

**Methane.** Conrad (1989) reviewed the factors controlling methane production in terrestrial ecosystems, and his conclusions included the following: 1) biogenic methane production is accomplished only by specialized methanogenic bacteria in the strict absence of oxygen (redox potential < -200 millivolts); and 2) methanogens are capable of metabolizing only a limited number of very simple organic compounds (Table 3). Reaction 5 is a generalized

equation for methane production, which is actually accomplished in stages by a suite of organisms (Conrad, 1989).

Reaction 5. $C_6H_{12}O_6 \rightarrow 3CO_2 + 3CH_4$

This reaction occurs only under anaerobic conditions. If methane passes through an aerated zone in the soil or water, it is readily oxidized by methanototrophic bacteria in a process similar to Reaction 1. In fact, methane gas must frequently pass through an aerated layer before reaching the atmosphere. A subsurface $CH_4$ flux can be partially, or even completely, attenuated by passage through an aerobic layer (Conrad, 1989; King et al., 1990). Oxygen emissions from plant roots and algae also attenuate methane fluxes by promoting oxidation (King, 1990; Wang et al, 1993). King (1990) observed accelerated methane production in laboratory incubations with the onset of darkness and consequent cessation of oxygen production by photosynthesis. Thus, biological oxidation of methane is a primary constraint on its evolution from the soil. Such oxidation is largely avoided when methane is released in bubbles, which rise quickly to the surface in a process known as ebulliation. Bubbles are formed when the rate of $CH_4$ production in the anoxic zone exceeds the capacity for $CH_4$ to be to diffused away in solution.

In many well-aerated soils, methane is not produced at all, but is continuously consumed from the atmosphere by biological oxidation. Some investigators believe this process could be an important global sink for methane (Keller et al., 1983; Steudler et al., 1989; Keller et al., 1990; Mosier et al., 1991).

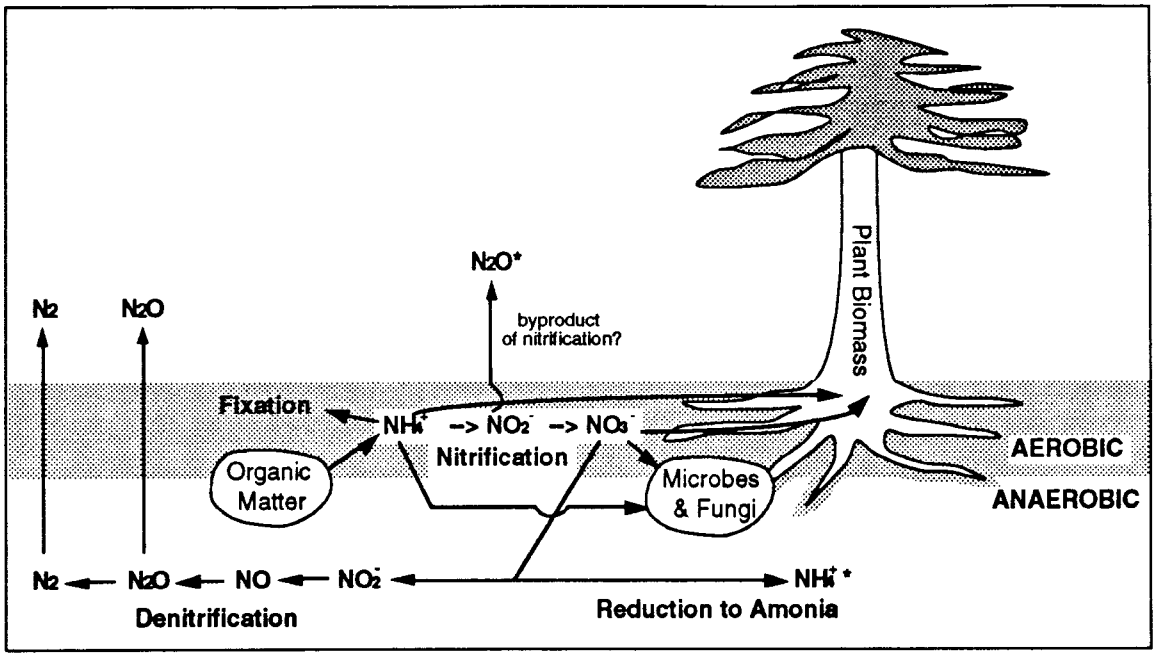
Table 3. Substrates of methanogenic bacteria

| Process | | $\Delta G^{\circ\prime}$ (kj / mol $CH_4$) | Abundance [c] |
|---|---|---|---|
| *Complete Degradation* | | | |
| $4\ CO + 2\ H_2O$ | $\rightarrow\ 3\ CO_2 + CH_4$ | -186[a] | many species |
| $4\ H_2 + CO_2$ | $\rightarrow\ 2\ H_2O + CH_4$ | -136 | almost every species |
| $4\ HCOOH$ | $\rightarrow\ 3\ CO_2 + 2H_2O + CH_4$ | -130 | most species |
| $4\ CH_3OH$ | $\rightarrow\ CO_2 + H_2O + 3\ CH_4$ | -105[b] | *Methanosarcina, Mehanococcus* |
| $4\ (CH_3)_3\ NH^+ + 6\ H_2O$ | $\rightarrow\ 3\ CO_2 + 4\ NH_4^+ + 9\ CH_4$ | -73[b] | |
| $2\ (CH_3)_2S + 2\ H_2O$ | $\rightarrow\ CO_2 + 2\ H_2S + 3\ CH_4$ | -49[b] | recently discovered |
| $CH_3COOH$ | $\rightarrow\ CO_2 + CH_4$ | -31 | *Methanosarcina, Mehanothrix* |
| *Incomplete Degradation* | | | |
| $2\ CH_3CH_2OH + CO_2$ | $\rightarrow\ 2\ CH_3COOH + CH_4$ | -116[a] | |
| $4\ CH_3CHOHCH_3 + CO_2$ | $\rightarrow\ 4\ CH_3COCH_3 + H_2O + CH_4$ | -36[a] | recently discovered (Widdel, 1986) |

[a] Ecological importance unknown
[b] Noncompetitive substrates (Oremland, 1988)
[c] e.g., see Vogels et al. (1988)

*From Conrad (1989)*

**Nitrous Oxide.** In the laboratory, many biological pathways of nitrous oxide production have been identified. In fact, Firestone and Davidson (1989) state in a review that most biological processes involving the oxidation or reduction of N though the +1 or +2 state can probably produce trace amounts $N_2O$ and NO. However, only denitrification and nitrification have been widely accepted as significant direct contributors to $N_2O$ evolution from soil (Sahrawat and Keeney, 1986; Firestone and Davidson, 1989)—though the role of nitrification remains controversial. Knowledge of other potential contributors (e.g., fungal metabolism) is insufficient to judge their extent in nature (Tiedje, 1988). Under anaerobic conditions, the advantages of denitrification to organisms are clear, and denitrification has been firmly established as a large contributor to $N_2O$ production in many situations. Other processes can limit $N_2O$ evolution by competing for carbon and nitrogen in the soil (Figure 3.).




*(Modified from Gambrell and Patrick, 1978)* ** Potentially significant but extent in nature uncertain*

Figure 3. N-cycling processes related to the production of nitrous oxide

Nitrification is the biological oxidation of ammonium to nitrate, which occurs in two steps (Reactions 6 and 7), mediated by two different groups of organisms. Paul and Clark (1989) write that nitrifiers are "obligate aerobes that derive their carbon solely from $CO_2$ or carbonates, and their energy from the oxidation of $NH_4^+$ or $NO_2^-$."

Reaction 6. $$2NH_4^+ + 3O_2 \rightarrow 2NO_2^- + 2H_2O + 4H^+ + \text{energy}$$

*(From Mitsch and Gosselink, 1986)*

Reaction 7. $$2NO_2^- + O_2 \rightarrow 2NO_3^- + \text{energy}$$

*(From Mitsch and Gosselink, 1986)*

Nitrification undoubtedly promotes $N_2O$ production in the soil by providing $NO_3^-$ for denitrifiers, but $N_2O$ production by nitrifiers has also been observed (Goreau et al., 1980; Anderson and Levine, 1986). Most authors have been unable to explain $N_2O$ production by nitrifiers, but have assumed that it was a direct result of nitrification (e.g., Sahrawat and Keeney, 1986; Firestone and Davidson, 1989). Evidence of $N_2O$ production by nitrifiers in soil has been provided by studies in which the addition of ammonia to aerated soils increased $N_2O$ production, but the addition of nitrate did not (Bremner and Blackmer, 1978; Breitenbeck et al., 1980; Bremner et al., 1981).

Poth and Focht (1985) seriously challenged the belief that nitrification is a source of $N_2O$. They tested two mutually exclusive hypotheses concerning $N_2O$ production by the common nitrifier, *Nitrosomonas europaea*:

1) **Nitrification- unstable intermediate** (the more popular theory): "Nitrous oxide is produced during nitrification by various reactions of intermediates formed during ammonium oxidation and is enhanced by oxygen stress, which promotes the formation and retention of intermediates."
2) **Denitrification.** "*N. europaea* reduces nitrite to nitrous oxide directly (denitrification) under conditions of oxygen stress while it is actively oxidizing ammonium."

The latter process is not nitrification because neither $NO_2^-$ or $NO_3^-$ is produced. With the aid of $^{15}N$ tracers, Poth and Focht (1985) showed that *N. europaea* produced $N_2O$ only under oxygen-limiting conditions and $N_2O$ resulted from reduction of $NO_2^-$ (not $NO_3^-$). In effect, *N. europaea* became a denitrifier when oxygen was limiting.

Poth and Focht (1985) noted that this conclusion was entirely consistent with the accumulated evidence that had previously been used to support the nitrification-unstable intermediate theory: namely, that 1) nitrification inhibitors also inhibit $N_2O$ production, possibly by denying $NO_2$- or $NO_3$- for denitrification, 2) nitrous oxide production is promoted by low oxygen in cultures and the field, and 3) $N_2O$ production in uplands is stimulated by the addition of ammonium rather than nitrate, perhaps because ammonium and nitrite are required for denitrification by *N. europaea.*

Still, $N_2O$ production by bacteria that are predominantly nitrifiers could be significant in uplands. Firestone and Davidson (1989) write, "High rates of $N_2O$ production are more commonly associated with denitrification rather than nitrification. Nevertheless, denitrification rates are spatially and temporally highly variable across ecosystem types, whereas nitrification is a relatively constant process in many ecosystems. Hence, a small $N_2O/NO_3^-$ production ratio [in nitrification] may be globally significant." This would be true regardless of whether $N_2O$ was produced directly through nitrification or by the denitrification pathway favored by Poth and Focht (1985).

Denitrification is the respiratory reduction of $NO_3^-$ or $NO_2^-$ to the gaseous products NO, $N_2O$, and/or $N_2$. Nitrous oxide evolution from the soil indicates a special set of circumstances in which $N_2O$ is produced, its reduction to $N_2$ is avoided, and is it able to diffuse to the surface or otherwise escape. Thus, $N_2O$ evolution represents only a fraction of one anaerobic respiratory pathway in the soil. Payne (1985) regards nitric oxide only as an intermediate, never a terminal product of denitrification. However, rapid NO production, attributed to denitrification, has occurred in flow-through soil cores (Johansson and Galbally, 1984, Drury et al., 1992). Firestone and Davidson (1989) contend that flow-through systems are quite artificial, but leave open the possibility that significant

amounts of NO are produced during denitrification and dispersed in the soil. Apparently, NO is readily reduced and little escapes the surface.

$N_2O$ and $N_2$ appear to be the dominant terminal products of denitrification in soils. Some denitrifiers terminate respiration at $N_2O$ (lack the $N_2O$ reductase), while others are capable of full denitrification from $NO_3^-$ to $N_2$ (Payne, 1985; Zumft and Kronek, 1990). A variety of organisms possess intermediate capabilities (Payne, 1985). The ratio of $N_2O$-N to $N_2$ evolved is highly variable and sensitive to several soil factors (Table 4).

Table 4. Factors affecting the proportion of $N_2O$ and $N_2$ produced during denitrification

| Factor | Will increase $N_2O/N_2$ |
|---|---|
| $[NO_3^-]$ or $[NO_2^-]$ | Increasing oxidant |
| $[O_2]$ | Increasing $O_2$ |
| Carbon | Decreasing C availability |
| pH | Decreasing pH |
| $[H_2S]$ | Increasing sulfide |
| Temperature | Decreasing T |
| Enzyme status | Low $N_2O$ reductase activity |

*(From Firestone and Davidson, 1989)*

## Sensitivity to Disturbances

The previous review has demonstrated that 1) soil disturbance can alter temperature, moisture, aeration, structure, and nutrient distribution in soils; 2) rates of biogenic gas flux are sensitive to these factors; and 3) these factors are profoundly interrelated. Thus, biogenic gas fluxes seem a natural tool in determining the overall degree of perturbation of the soil system. In the field and laboratory, gas fluxes have responded similarly to a variety of disturbances.

**Compaction**. Compaction can alter soil microbial activity by decreasing aeration and favoring anaerobic over aerobic processes. In several studies,

compaction has resulted in increased N-gas emissions and/or decreased evolution of $CO_2$. In cores taken from a Coastal-Plain loamy sand and compacted in the laboratory, Torbert and Wood (1992) observed lower total soil microbial activity and higher N losses with increasing bulk density. N loss was determined by measurement of soil N over the course of the experiment, rather than direct measurement of gas fluxes. N loss was attributed to microbial denitrification induced by poor aeration.

Similarly, in cores collected from an Alfisol subjected to wheel pressure, 26-56 % of injected $^{15}N$ was lost to denitrification versus only 0-5 % in uncompacted cores (Becker et al., 1989). The maximum denitrification rate of the compacted soil (5000 g-N $ha^{-1}$ $day^{-1}$) was fifteen times higher than the maximum rate in uncompacted soil. Bakken et al. (1987) found that tractor traffic during wet conditions in agricultural soil increased N loss by denitrification 3-4- fold versus controls, an effect attributed to reduced porosity, aeration, and soil structure. Denitrification was not affected by dry weather traffic.

**Drainage and Water -Table Fluctuations.** The effects of compaction are due largely to its influence on aeration and soil moisture. Rates of most aerobic microbial processes are positively correlated with soil moisture up to some level, above which moisture becomes limiting. Linn and Doran (1984) noted that in many studies involving a wide range of soil types, 60 % saturation delineates the point of maximum aerobic microbial activity; and the optimum level for denitrification is above 60-70 %. Neilson and Pepper (1990) found that soil moisture became limiting at higher levels (77-97 % saturation, depending on soil amendments and bulk density). Differences among studies are due to both soil characteristics and methods; but most studies have shown similar trends, including a limiting value for respiration below saturation.

Colbourn and Harper (1987) concluded that drainage of an agricultural soil reduced total denitrification 35 % and promoted the production of $N_2O$ over $N_2$. $N_2O$ is likely to be reduced to $N_2$ when redox potential is low and diffusion inhibited because of a high water table. Unfortunately, only one plot represented each the drained and undrained condition, and there was no pretreatment calibration period. Results should be interpreted only in the context of other studies.

Alternate wet and dry cycles can provide optimum conditions for $CO_2$ evolution, methanogenesis, and denitrification. Reddy and Patrick (1975) examined the effect of varying cycles of alternate aerobic and anaerobic conditions on organic-matter decomposition and N loss in flooded soil samples incubated for 128 days. Anaerobicity was achieved by flushing with argon. Nitrogen loss was maximized by the greatest number (of shortest duration) cycles. Nitrogen loss was probably stimulated by alternating periods of nitrification (aerobic), which supplied nitrate, and denitrification (anaerobic), which caused gasification of the nitrate to $N_2$ or $N_2O$. Alternating cycles—in comparison with continuous anaerobicity—did not stimulate decomposition. However, decomposition under continuous anaerobicity was about half that of the aerobic treatment.

Reddy and Patrick's (1975) results were in agreement with earlier studies (Wijler and Delwiche, 1954; Patrick and Wyatt, 1964; MacRae et al., 1967) revealing high N loss under alternate drained and flooded conditions. However, earlier work had also shown stimulation of organic-matter decay by alternate wetting and drying (Gooding and McCalla, 1945; Stevenson, 1956; Birch, 1960). Reddy and Patrick's work, thus, suggests that factors other than oxygen (perhaps labile carbon or diffusion) stimulate decomposition under alternate wetting and

drying. Their study isolated only those factors associated with the presence or absence of oxygen and did not fully represent field conditions.

Windsor et al., (1992) determined that a significant amount of methane evolved from subarctic fens in episodic peaks associated with fluctuations in water-table depth. Several explanations, including the following, were offered: 1) a rise in the water table expelled methane from pore spaces, and 2) lowering the water table triggered the release of stored methane or increased methane diffusion. Several investigators (Birch, 1958; Stevenson, 1956; Soulides and Allison, 1961) have demonstrated that soil drying increases the availability of organic carbon, which provides energy for virtually all microbial transformations. Disturbances that affect water table depth include drainage, clearcutting (which can raise a water table due to lack of transpiration), and bedding (which elevates the planting surface).

## Atmospheric Effects of $CH_4$ and $N_2O$

Nitrous oxide and methane absorb infrared radiation and contribute to the natural containment of heat in the earth's atmosphere. While $N_2O$ and $CH_4$ concentrations in the atmosphere are 3 orders of magnitude lower than that of $CO_2$, they absorb infrared radiation much more strongly and their contributions to "global-warming potential" are disproportionately large (Lashof and Ahuja, 1990; Rhode, 1990; Figure 4). The atmospheric concentration of $N_2O$ is increasing about 0.27 % per year (Prinn et al, 1990) and $CH_4$ at about 1.0 % per year (Cicerone and Oremland, 1988; Blake and Rowland, 1988). Data from ice cores and direct atmospheric measurements suggest that after a prolonged period of stability at approximately 0.7 ppm, the $CH_4$ concentration in the atmosphere began to increase in the early 1800's, and rate of increase has accelerated until present (Rasmussen and Khalil, 1984; Ehalt, 1988). The current concentration of $CH_4$ in the atmosphere is about 1.7 ppm (Ehalt, 1988).

$N_2O$ and, to a much lesser extent, $CH_4$ also influence stratospheric ozone chemistry. $N_2O$ can migrate to the stratosphere, where it degrades readily into—and may be the primary source of—nitric oxide (McElroy and McConnell, 1971; Crutzen, 1974). Nitric oxide destroys ozone catalytically, and Crutzen (1974) considers production of $N_2O$ by microorganisms in the ocean and soil to be "one of the most critical factors for the ozone budget of the earth." Marine and terrestrial contributions of $N_2O$ have not been well partitioned. $CH_4$ may mitigate slightly against ozone deletion by neutralizing chlorine atoms, which would otherwise react detrimentally with ozone (Crutzen, 1974).

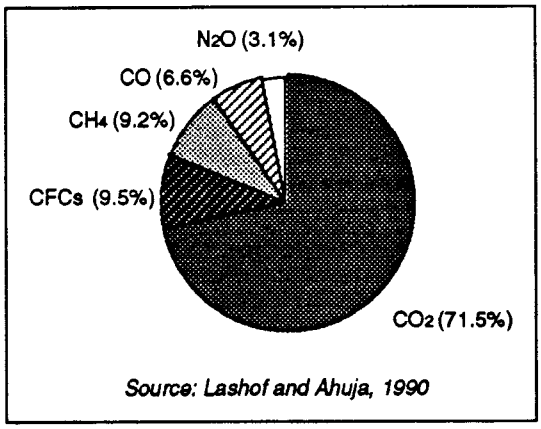



Figure 4. Relative contributions to global warming potential in 1985

These discoveries have prompted efforts to better quantify biogenic sources and sinks of $N_2O$ and $CH_4$ and to assess the influence of human activities on their atmospheric budgets. For example, cultivation and fertilization have been found to decrease consumption of $CH_4$ in North American grasslands and increase production of $N_2O$ (Mosier et al, 1991). This study and others (eg, Keller et al, 1990; Slemr et al, 1984; Stuedler et al, 1989) suggest that fertilization and changes in land use are contributing to the observed increase in atmospheric concentrations of $N_2O$ and $CH_4$, but to an unknown degree.

Prinn et al. (1990) state that the cause of increasing atmospheric $N_2O$ "appears to be a combination of a growing tropical source (probably resulting from tropical land disturbance) and a growing northern mid-latitude source (probably resulting from a combination of fertilizer use and fossil fuel combustion)." Their recently developed global model estimates that land—including wetlands—contributes approximately 80 % of the total $N_2O$ flux to the atmosphere The oceanic contribution of 20 % was obtained by subtracting the sum of tenuous land estimates from the supposed total global flux. Contradictory evidence (Elkins et al, 1978; Cohen and Gorden, 1979) suggests a larger role for the oceans (and a smaller one for land), assuming the estimated total flux to the atmosphere is correct.

A global budget developed by Sheppard et al. (1982) estimated contributions to the total $CH_4$ flux from open ocean and continental shelf areas of less than 1 %. If so, land management may affect atmospheric $CH_4$ disproportionately.

The observed increases in $N_2O$ and $CH_4$ appear to have resulted from a variety of poorly identified anthropogenic sources. Preliminary partitioning of total fluxes has been based on many admittedly tenuous assumptions, few measurements, and extravagant extrapolations of flux rates beyond their scale of measurement. Therefore, policy recommendations are unclear, and detailed inventory and classification of ecosystems are needed. The primary value of recently developed models is to give structure to current knowledge and point the way for future investigations. The present study will contribute to the scant database on sources and sinks of $CO_2$, $CH_4$, and $N_2O$.

## Study Objectives

The effects of skid-trail formation and related soil disturbances on tree growth have been thoroughly studied. However, little information is available concerning the effectiveness of remediation treatments such as tillage and

fertilization. Theoretically, biogenic gas fluxes should be strong composite indicators of soil condition and, thus, useful in detecting residual disturbance effects—even after individual physical or chemical characteristics are restored by remediation.

**Objectives**

1) Determine if harvest damage is apparent despite mitigation by bedding and fertilization.

2) Determine the effects of harvest damage, bedding, and fertilization on aerobic and anaerobic biological activity in the soil.

3) Determine specific microsite conditions and soil properties that influence the production of $CO_2$, $CH_4$ and $N_2O$.

## MATERIALS AND METHODS

### Study Area

Study sites were located in the Wambaw district of the Francis Marion National Forest in Berkely County, South Carolina. The forest is centered at 33° 10′ north, 79° 42′ west, about 25 miles northeast of Charleston, South Carolina in the lower Coastal Plain. In Winter of 1991, McKee et al. (1991) installed plots throughout the forest on phosphorous-deficient Ultisols, "previously occupied by merchantable pine stands." Their study was established to assess soil disturbance resulting from post-Hurricane Hugo timber salvage

Two sites were selected for the present study to represent a range of drainage conditions in Ultisol pine flats. One site was located on a poorly drained Bethera series soil and the other on a moderately well drained Goldsboro series. Drainage conditions varied within each site. The Bethera series is a Typic Paleaqult and Goldsboro an Aquic Paleudult. Soils were formed on marine and fluvial sediments, deposited in the Pliestocene Epoch (within the past million years) (USDA Soil Conservation Service, 1980). Rainfall is evenly distributed throughout the year and total annual rainfall averages about 120 cm.

### Treatments and Experimental Design

There were three levels of mechanical soil disturbance (none, bedding, and bedding over residual harvest damage). Three plots—one for each level of mechanical soil disturbance—comprised a block. There were four such blocks over two sites, two blocks per site. Each plot was split into fertilized and unfertilized halves, for a 3 x 2 factorial arrangement of treatments (Figure 5).

Because the experiment was installed after the damage had occurred, randomization within the blocks was constrained. Soil cores and examination of

plots revealed no difference in horizonation, drainage, or vegetative cover in the vicinity of disturbed and undisturbed areas. Skid-trail placement appeared to be unrelated to these characteristics, following a similar radial pattern on all sites. While randomization is necessarily constrained in a post-hoc damage assessment; the conditions are real, and lasting damage to additional areas for experimental purposes is avoided.

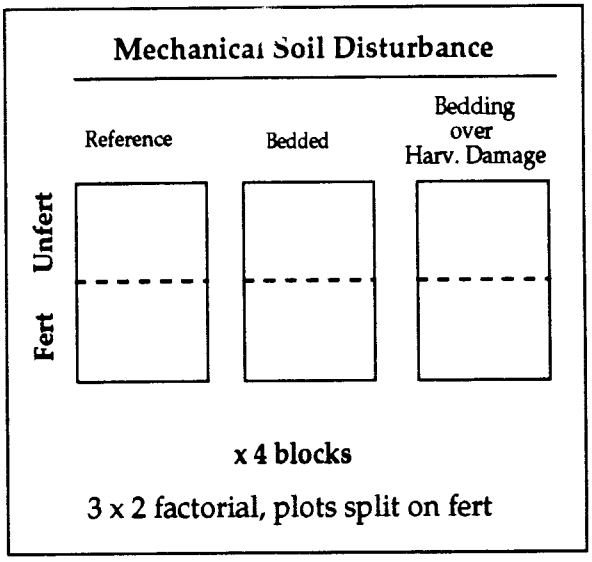



Figure 5. Treatments

## Bedding

The bedding treatment chosen involved two mechanical passes: 1) disking to a depth of eight inches, and 2) bedding with a fire plow at close spacing (about four feet between row centers) (McKee et al, 1991). The beds were approximately 15 inches tall from furrow to crest. The use of a fire plow resulted in the A-horizon material, with litter layer and herbaceous vegetation, being stripped away from the interbed rows, inverted, and deposited into a bed. The $B_t$ horizon was exposed in the interbed rows.

Microsites were selected within the bedded plots to serve as dedicated sampling locations throughout the study. To allocate limited sampling capacity as efficiently as possible, two sampling microsites were located in the beds of each subplot and one in the interbed rows (Figure 6). The interbed rows were sampled less intensively than other microsite types because exposure of the $B_t$ horizon produced a relatively homogeneous and infertile environment, where low rates of biological activity were anticipated.

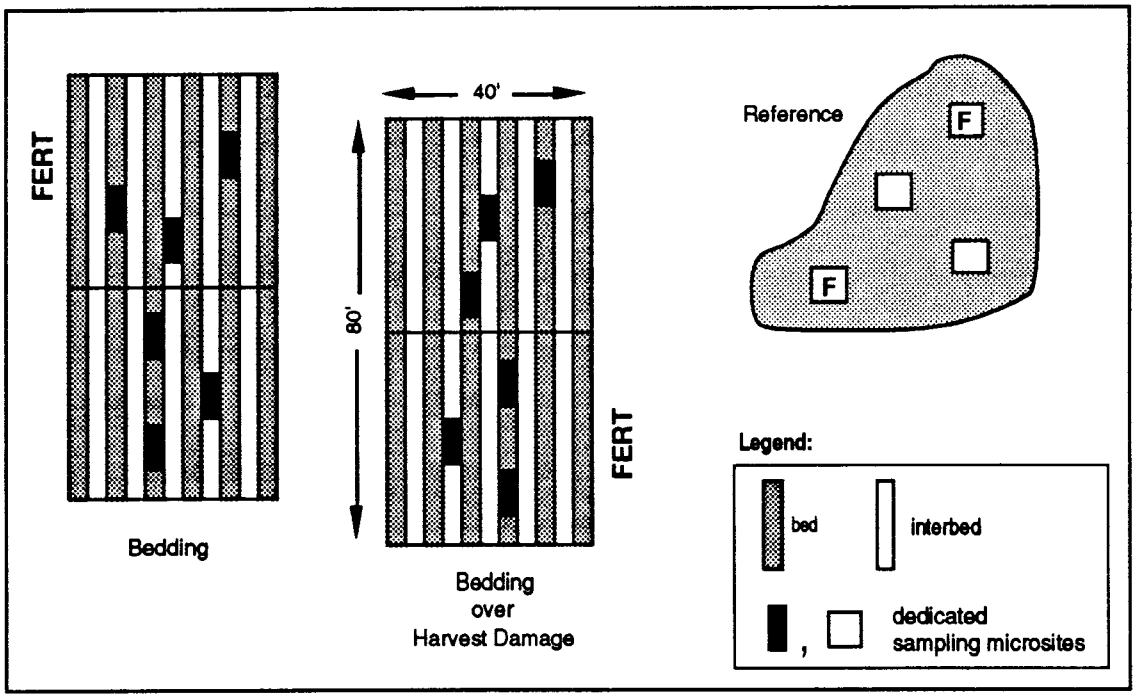



Figure 6. Physical arrangement of block and placement of permanent sampling microsites (idealized)

Bed formation was inconsistent on the ends and outer rows of the plots, so random selection of sampling microsites was constrained largely to exclude these areas. In the beds themselves, microsites were selected from sections that were within ± 5 cm of the mean bed height for each plot. Stumps were excluded.

Interbed microsites were selected randomly, excluding sections where the adjacent beds converged too closely for installation of sampling equipment.

At the start of this study, the beds had not yet settled and the surface was too rough for installation of gas-sampling equipment. Therefore, each microsite was prepared by breaking apart the large clods with a pick to produce a well formed bed. No soil was added or removed.

**Harvest Damage**

The harvest damage represented in this study resulted from the formation of skid trails during post-Hurricane Hugo timber salvage and was variable in terms of time since formation, antecedent conditions at the time of formation, and intensity of rutting and compaction. The Goldsboro site was classified as primarily rutted and the Bethera site as primarily compacted (Tippett, 1992), but both forms of disturbance were present at each. High variability in the harvest-damage factor provided for robust conclusions in the event of significant results.

**Fertilization**

Half of each plot was fertilized with 100 kg/ha of nitrogen, phosphorous, and potassium. Elemental sources were urea for nitrogen, triple super phosphate for phosphorous, and potash for potassium. On the sampling microsites, application rates were carefully controlled. Before fertilizer pellets were broadcast on fertilized subplots, sampling microsites were covered with plastic sheets. The sheets were then removed and the pellets that had fallen on them discarded. Fertilizer pellets previously ground in a Wiley mill were dissolved (when possible) or suspended in approximately two gallons of water and applied to the microsites with a watering bucket.

**Undisturbed Reference Areas**

Undisturbed reference areas—not associated with the McKee et al. (1991) study—were established near the other two plots in each block. Within each reference

area, four 0.5 x 1 m sampling microsites were randomly located (Figure 6). The two fertilized microsites were centered within 2 x 2 m fertilized areas, treated with 100 lbs/ha N-P-K as in the bedded plots. The excess fertilized area surrounding the microsites served as a buffer. In reference areas, each level of fertilization was represented in calculations by the average of two microsites.

### Differentiation of Microsite Types

To address certain questions, it was necessary to examine soils at the microsite—rather than the plot—level. For example, factors that may have affected soil condition in the beds alone (where the seedlings were planted) were of particular interest. Throughout the study areas, four microsite types were identified: 1) undisturbed microsites (intact forest floor), 2) beds with residual harvest damage, 3) beds without residual harvest damage, and 4) interbed rows. Each type was represented by an equal number of subjects.

No distinction was made between the existence or absence of residual harvest damage in the interbed rows. This distinction was considered unmeaningful for two reasons: 1) in the interbed rows, a sheared and compacted Bt horizon was exposed regardless of the area's previous condition; and 2) the interbed rows were frequently submerged, resulting in low rates of and spatial variability in gas-exchange rates.

Thus, for microsite analysis, there were four microsite types and two levels of fertilization for a total of eight microsite conditions. Four blocks were recognized. Within each block, as previously described, the microsites were randomly located in the area occupied by each combination of microsite type and fertilization.

## Sampling Schedule

To determine seasonal variation in process rates and the consistency of treatment effects over time, measurements were conducted over four trial periods in 1992

(Table 5). A trial period consisted of two visits to the Francis Marion National Forest, one to each of the two sites. The two blocks contained within a site were typically visited within two days of each other. Logistical problems delayed the second site visit in Trial 1.

Table 5. Time line 1992.

| | Trial 1 | | | | | Trial 2 | | | Trial 3 | Trial 4 | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | G | | | B | | G | B | | G B | G B | | |
| | Jan | Feb | Mar | Apr | May | Jun | Jul | Aug | Sep | Oct | Nov | Dec |
| Date Measured | 18<br>19 | | | 7<br>9 | | 11<br>13 | 2<br>7 | | 9 16<br>1117 | 6 20<br>8 21 | | |

**G** - Goldsboro Site
(moderately well drained)

**B** - Bethera Site
(poorly drained)

## Measurement of Gas Fluxes in the Field

### Equipment Selection and Testing

Gas fluxes were measured with a static chamber (Figure 7) in which temporal changes in concentrations of carbon dioxide, methane, and nitrous oxide were attributed to exchange across the soil surface. By using large chambers and leaving them in place for short periods, edge effect and heating were minimized. Foil shields provided further protection against solar heating. Battery powered fans, activated before each sampling, compensated for the potential uneven distribution of gases in the chamber.

To determine if there was any leakage or absorption of nitrous oxide by the chamber materials or water seal, one of the chambers was tested for its ability to maintain a nitrous oxide concentration of 1ppm for an extended period. The open bottom of the chamber base was sealed to a plexiglass sheet (the same material as the chamber walls) with silicon sealant. There was no reduction in

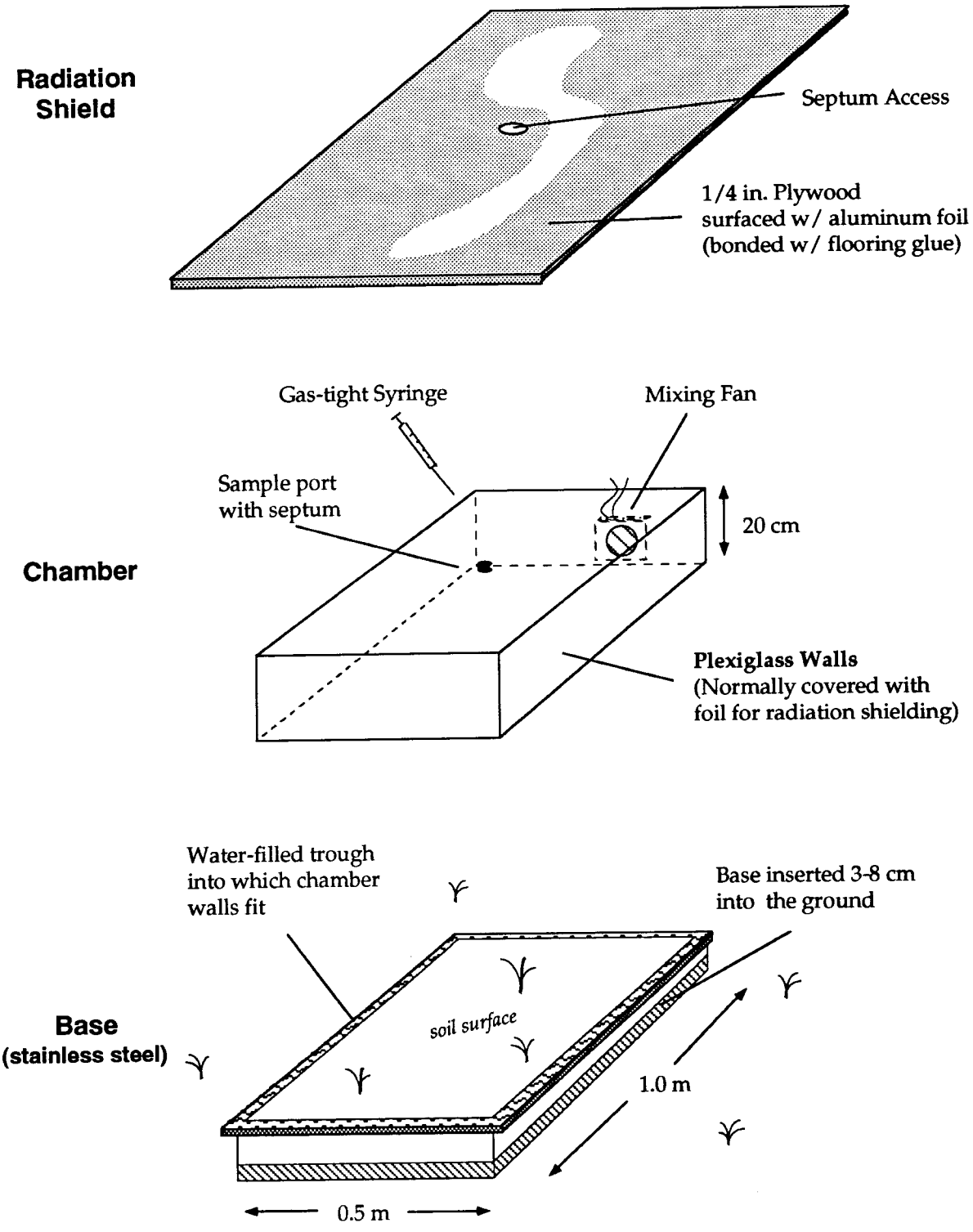



Figure 7. Static chamber for sampling of gas fluxes

concentration over 60 hours—well in excess of the planned 2-hour incubation time.

A 10 ml crimp-top vial (West Company) was selected for storage of gas samples. Crimp-top vials offered three advantages over other alternatives[2]. 1) Gases could be stored at pressures well above atmospheric. Therefore, when a portion was withdrawn for analysis, excess sample was expelled from the syringe, and there was little possibility of contaminating the gas in the syringe with room air. 2) Should a storage vial leak (e.g., through a needle puncture), "overpressuring" afforded some insurance against sample contamination. 3) The vials held enough gas to permit multiple withdrawals and analyses of the same sample. The vials were evacuated by inserting a needle attached to an electric vacuum pump for 30 seconds. The suction strength of the pump was approximately 93 centibars.

A small (24 gauge) needle was used in order to minimize the size of puncture holes in the vial septa. The septa were made of a minimally-reactive butyl rubber formulation. Tests indicated that sample integrity was maintained in the vials for at least two months.

**Chamber Installation**

The chamber base was installed the day before sampling by hammering it into the ground so that the walls extended 3 to 8 cm below the surface. A board was placed over the metal to prevent damage and distribute force of hammering evenly. The rim was leveled and the trough filled with approximately 2 liters of water. Since microtopography influenced the volume of the chamber headspace, a six-pin wooden profiler was placed across the width of the base at eight

---

[2] The primary alternative was a Vacutainer™ blood-storage vial, in which sample had to be stored at or near atmospheric pressure (because the cap is not strongly secured). If a syringe needle was introduced to withdraw sample, the gas would be distributed over an increased volume and the pressure reduced below atmospheric. Thus, when the syringe was withdrawn, room air would flow in due to the pressure differential, contaminating the sample.

intervals, and the elevation of the soil surface relative to the rim (within 0.5 cm) recorded for a grid of 48 points over the sampled area. Vegetation was clipped to height even with the rim.

**Sampling**

Barometric pressure and temperature were recorded before each set of incubations. At the beginning of an incubation, the plexiglass top was lowered onto the base and the radiation shield placed immediately over the chamber. The sampling port was left open, permitting air to escape until pressure in the chamber equilibrated with that of the atmosphere, then sealed. Thirty-five ml of air were withdrawn from the chamber into a 50 cc Hamilton™ gas-tight syringe and injected into an evacuated 10 ml crimp-top vial for storage. Additional gas samples were withdrawn at approximately 40 and 120 minutes and the times recorded. (Roughly 40 minutes was required to initialize eight incubations and return to the first chamber.) A subset of the chambers was sampled more intensively—up to ten times per incubation—to determine if gas evolution was linear over time. After each incubation, the bases were removed. On the first sampling date, the corners of the sampled area were marked with pin flags for precise relocation of the chambers.

Eight chambers and sixteen bases were available, plus one spare of each. Limited equipment availability imposed constraints on the sampling schedule that are reflected in the sampling chronology (Appendix I). Typically, the sixteen microsites comprising a block were sampled in one day, eight in the morning and eight in the afternoon.

**Calculations**

Gas concentrations were determined by gas chromatography (described in the following section). The mass of a particular gas present in the chamber was calculated using a derivation (Equation 2) of the ideal gas law (Equation 1). The

gas' flux rate across the soil surface was then estimated by fitting a least-squares linear equation of mass versus time. The ideal gas law is expressed in the following equation.

**The Ideal Gas Law:**

Equation 1. $$PV = nRT$$

or to solve for quantity: $$n = \frac{PV}{RT}$$

Where $P$=pressure, $V$=volume, $n$=moles of gas,
$R$=the universal gas constant, and $T$=temperature (°K)

Zumdahl (1986) states that, "most gases obey this equation closely at pressures below 1 atmosphere," noting that the equation has been derived empirically from experimental measurements on the properties of gases. The universal gas constant ($R$) can be expressed in a variety of units and has a value of 0.08206 L atm/°K mol or 62.4 L mb/°K mol. Equation 1 was modified to calculate the mass of a single gas in an air mixture, by accounting for the fact that moles are equivalent to grams ÷ molecular weight, and by employing Dalton's Law of Partial Pressures:

Equation 2. $$g = \frac{(m.w.)P_{partial}V}{RT}$$

The pressure referred to in Equation 2 is not total atmospheric pressure, but the partial pressure of a particular gas. Dalton's Law of Partial Pressures states that, "for a mixture of gases in a container, the total pressure exerted is the sum of the pressures that each gas would exert if it were alone" (Zumdahl, 1986). The partial pressure of a gas was calculated by multiplying its concentration by the total atmospheric pressure (Equation 3.)

Equation 3. $$P_{partial} = P_{total} \times ppm$$

PPM (parts per million) is a unitless measure of concentration that indicates the partial pressure exerted or the partial volume occupied by a gas. Partial pressure and partial volume thus have the same numerical values.

Gas fluxes in the bedded plots were estimated by area-weighted averaging of beds and interbed rows (Equation 4). The weighting values (0.7 for the beds and 0.3 for the interbed rows) reflect the fraction of total surface area typically occupied by each microsite type and were determined by characterization of surface profiles across a subset of the bedded plots. The area-weighted flux calculation also accounted for a 12 % increase in surface area resulting from increased micro-relief in bedded areas.

Equation 4. $$Flux_{plot} = ((0.7)Bed + (0.3)Interbed) \times 1.12$$

## Determination of Gas Concentrations

### Gas Chromatography

Gas concentrations were determined by analysis on a Shimadzu GC 14-A gas chromatograph (G.C.), equipped with a 10-port valve to facilitate gas analysis. The valve had two purposes: 1) to permit repeated injections of precise volume, and 2) to eliminate extraneous peaks after the peak of interest. The valve contained a sample loop connected to an injection port and was initially open to the atmosphere. When the valve was switched, a portion of the sample loop was incorporated into the carrier gas stream. Since this portion of the loop was of fixed volume, the primary factor that affected the mass of a gas it contained was atmospheric pressure, which varies little. Two columns (a 3 ft Poropak™ N with 80/100 mesh and a 10 ft Poropak™ Q with a 100/120 mesh) were connected in

series through the valve. The former served as a "filtering column" and the latter an analytical column. After the gas to be analyzed had eluted from the first column, the valve was switched and the first column backflushed to eliminate extraneous peaks (e.g., acetylene) that sometimes followed. Configurations for the G.C. are given in Table 6.

Table 6. Gas-chromatograph configuration for each gas studied

| Gas | Detector | Carrier Gas | Column T (°C) | Detector T (°C) | Flow Rate (ml/min) |
|---|---|---|---|---|---|
| Nitrous Oxide | Electron Capture Detector (ECD) | Argon 95% Methane 5% (P5 Mixture) | 70 | 340 | 40 |
| Carbon Dioxide | Thermal Conductivity Detector (TCD) | Helium | 50 | 110 | 35 |
| Methane | Flame Ionization Detector (FID) | Helium | 50 | 250 | 35 |

**Same column arrangement for all analyses:** 3 ft. Poropak N™ & 10 ft. Poropak Q™ inseries. Further description provided in the text.

## Standard Preparation

Standards were prepared using the principle of partial volumes. A 2-liter flask was filled with either helium or air and adjusted to ambient atmospheric pressure. The partial volume required to achieve a particular concentration was calculated and withdrawn from the flask. The same volume of a pure gas was

injected, and the flask, which contained glass mixing spheres, placed on a shaker for 15 minutes. Low concentration (0.5 and 1.0 ppm) Scott™ brand nitrous-oxide standards were also used. Before analysis on the G.C., standards were injected into evacuated vials in the same manner as samples in the field. Any effects associated with storage in the vials (such as dilution by the air inevitably remaining despite evacuation) were thus applied equally to standards and samples.

## Soil Sampling

Soil samples were collected immediately before returning to the lab to minimize storage time before biological assays and other time-sensitive analyses. Typical times elapsed between sample collection and analysis are reflected in Table 7.

Table 7. Typical time elapsed before analyses of soil samples

| *Days After Arrival* | *Activity* |
|---|---|
| 1 | Mix composite samples, prepare reagents. |
| 2 | KCl extraction for $NO_3^-$ and $NH_4^+$. |
| 3 | Soluble organic carbon—same day as KCl extraction if possible. |
| 4 | Denitrifying enzyme activity. |
| 5 | $N_2$:$N_2O$ ratio. |

Soil was sampled to a depth of 23 cm with a push tube (2 cm in diameter). Six samples were collected from the area of each chamber, placed in a polyethylene bag, and immediately into a cooler. To maintain temperatures close to those of the field, samples were transported in large coolers in which the

balance of space was filled with bagged, freshly-collected soil. On returning to the lab, the samples were placed in an incubator set to the average soil temperature observed during the trip. The soil in each bag was mixed thoroughly.

## Soil Chemical and Physical Properties

### Nitrate and Ammonia

Soil nitrate and ammonia were determined by KCl extraction and analysis on a Technicon™ autoanalyzer. Due to the ephemeral nature of nitrate and ammonia pools, the extraction was performed as soon as possible after sample collection (Table 7). Ten grams of fresh, well-mixed soil and 100 ml of 2 *M* KCl solution were placed in a flask. The flask was shaken for one hour, then allowed to sit until the solution had partially cleared. A portion of this solution was decanted into a 15 ml Falcon™ centrifuge tube, which was spun horizontally at 1,800 rpm for 20 minutes. The supernatant was decanted into an acid-washed plastic storage vial, which was stored at 4°C until analysis.

### Kjeldahl Nitrogen and Phosphorous

Kjeldahl nitrogen and phosphorous were determined colorimetrically on a Technicon autoanalyzer following a hot-acid digestion (Technicon Industrial Systems, 1978)

### pH

Ten grams of air-dry soil were mixed with 10 ml water and the electrodes of a pH meter were inserted into the homogenized slurry.

### Total Organic Carbon

Organic carbon content was determined by combustion of soil in a muffle furnace at 450°C for 4 hours. Two grams of oven-dry soil that had been sifted through a 2 mm sieve were analyzed. Organic carbon was taken to be the material volatilized during combustion, and reported as a percent of total soil

dry weight. The samples analyzed were collected on the final site visit (in the manner described in the "soil sampling" section).

**Water Soluble Organic Carbon**

Water soluble organic carbon (as an index of carbon available to microbes) was determined by a cold-water extract procedure adapted from Burford and Bremner (1975). Ten grams of fresh soil and 20 ml of deionized water were placed in a 50 ml polyethelene centrifuge tube and shaken for 15 minutes. The tubes were centrifuged horizontally at 1,800 rpm for 5 minutes. The supernatant was transferred to a second tube, which was centrifuged for an additional 60 minutes at the same speed. The supernatant from the second centrifugation was filtered by suction through a 47 mm-diameter, 0.2 μm Metricel™ membrane filter (Gelman Instrument Co.), previously washed with 100 ml of water. The filtrate was analyzed on a Shimadzu organic carbon analyzer, which vaporizes then combusts the sample and analyzes for resultant carbon-dioxide evolution with an infrared detector.

**Bulk Density**

Samples for estimation of bulk density were collected with an impact corer that extracted a cylindrical soil core of radius 3.81 cm and length 7.62 cm. The core was placed in a paper bag, dried at 105°C for 48 hours and weighed. Bulk density was reported in $g \cdot cm^{-3}$.

At the conclusion of the study, a single core was collected from each sampling microsite in the beds and interbed rows. Two sets of cores (consisting of one core from the A horizon and one from the $B_t$ horizon) were collected from each undisturbed reference plot near the sampling microsites.

**Temperature and Moisture**

Soil temperature was measured with a dial thermometer inserted 23 cm into the soil. Soil moisture was measured by time domain reflectometry (TDR) (Topp et

al., 1980). Two parallel, 22.9-cm rods were inserted vertically into the soil, 5 cm apart. The estimate of soil moisture was integrated over the upper 22.9 cm of the soil column and in the plane between the two rods. A TDR reading yields an estimate of the fraction of the total soil volume occupied by water.

## Biological Laboratory Incubations

### Denitrifying Enzyme Activity

Denitrifying enzyme activity (DEA) was determined by a biological assay (Tiedje, 1982) in which conditions for denitrifiers were optimized and the atmosphere was amended with acetylene to ensure that the end product of denitrification would be $N_2O$. A nutrient solution was prepared that contained the following: 1 $g \cdot L^{-1}$ chloramphenicol, 200 $mg \cdot L^{-1}$ $NO_3$-N (from $KNO_3$), and 2 $g \cdot L^{-1}$ sucrose-C. Chloramphenicol is a protein synthesis inhibitor that minimizes microbial population responses to environmental modifications during an incubation, extending the period during which activity rates remain constant.

Ten grams of fresh, well-mixed soil and 10 ml of the solution previously prepared were added to a 125 ml serum bottle. The bottles were sealed, and then evacuated and refilled with helium three times. Helium pressure was permitted to equilibrate with that of the atmosphere. Sixteen bottles were connected to the helium tank and vacuum pump by a manifold. After the third filling with helium, the manifold was opened to the atmosphere through a flow meter. When outflow from the bottles ceased, the pressure in bottles was assumed to be equilibrium with the atmosphere. Ten milliliters of helium were withdrawn from each bottle and replaced with 10 ml of acetylene. The bottles were placed in an incubator (previously heated to 36°C) and shaken at 100 rpm on an orbital shaker. Four-milliliter gas samples were withdrawn after approximately one, two, and six hours and injected into 1-ml crimp-top vials.

$N_2O$ concentrations were determined by gas chromatography. Denitrifying enzyme activity was reported as the rate of $N_2O$-N evolution relative to the dry weight of the soil in the bottle (ng-$N_2O$-N·g-soil$^{-1}$ hr$^{-1}$).

**$N_2$:$N_2O$-N Ratio**

$N_2$ is a potential end product of denitrification, yet it is not readily measured against the atmospheric background of 78% $N_2$. Therefore, only $N_2O$ evolution was measured in the field. The magnitude of $N_2$ evolution relative to that of $N_2O$-N was estimated in a two-stage laboratory incubation.

In the first stage, the rate of $N_2O$ evolution from 30 g of fresh soil, placed in a 125 ml serum bottle, was measured over approximately 12 hours. The atmosphere in the bottle was sampled approximately one and twelve hours after the bottle was sealed. The bottle was opened, ventilated with fresh air, and resealed. In the second stage the same procedure was used, with the exception that the atmosphere was amended with acetylene to block the reduction of $N_2O$ to $N_2$. Presumably, $N_2$ evolution that had gone undetected in the first stage was revealed through increased $N_2O$ evolution in the second stage. $N_2$ evolution was thus taken to be the second-stage $N_2O$ evolution minus that of the first stage. To determine if the rate of $N_2O$ evolution changed for a reason other than the presence of an acetylene block, controls were included in which no block was applied in the second stage; the procedure for controls was otherwise identical.

**Statistics**

The following statistical analyses were performed: 1) means comparisons among treatments applied to whole plots 2) means comparisons among microsite types, and 3) examination of relationships. Each addressed a different set of questions in support of the study objectives. All analyses were performed using SAS statistical software.

Means comparisons among treatments applied to whole plots (Table 8) were intended primarily to answer questions concerning treatments effects at the field level and to characterize processes in the Ultisol pine-flat ecosystem for inclusion in terrestrial-gas-flux databases. Means comparisons among microsite types (Table 9) were intended to answer questions concerning the distribution of resources and activity within treated areas, including the question of whether residual harvest damage was apparent in the beds. Microsite analysis required the assumption that the variation within a block was uniform throughout the area before any soil disturbance occurred. The fertilization effect was not tested at the plot level, as it was already tested at the microsite level. Relationships between microsite characteristics and process rates were examined to identify the ephemeral and more lasting determinants of processes at the microsite level.

All means comparisons were accomplished by F-test-protected Duncan's Multiple Range Test. If the F-test resulting from the initial analysis of variance indicated no significant differences, means separation was not performed. Relationships among process rates and other measured parameters were analyzed by multiple linear regression, using the stepwise option to assist in the selection of independent variables.

## Data Normalization

Before comparing means of gas-flux rates (either among plot- or microsite-level effects), the data were normalized to minimize variation due to changes in soil temperature within a trial period. Ideally, every measurement in a trial period would be taken simultaneously, so that all variation in flux rates would be due to location, treatment effects, or random factors. However, by taking measurements on different days and different times within the same day, a known source of variation (temperature) was introduced that could have obscured treatment effects. During some trial periods, temperatures varied

slightly by treatment as well—probably due to concomitant differences in surface albedo, soil moisture, etc..

Flux rates were normalized to preserve differences due to treatment-induced temperature variation, while removing differences due to changes in temperature over time. Average soil temperatures during each trial period were calculated for beds, interbed rows, and reference areas. Assuming a $Q_{10}$ value of 2, flux rates on individual microsites were then normalized to the average temperature observed for that microsite type during the trial period. $Q_{10}$ is the factor by which rates change for a 10°C increase in temperature.

Table 8. Means comparisons among treatments applied to whole plots: ANOVA followed by means separation with Duncan's Multiple Range Test

| Factor | Levels |
|---|---|
| Block | 1, 2, 3, 4 |
| Mech. Soil Dist. | Undisturbed,<br>Bedded,<br>Bedded Over Harvest Damage |
| Fertilization | Fertilized, Unfertilized |

| Source | Degrees of Freedom |
|---|---|
| Whole-Plot D.F. | (11) |
| Block | 3 |
| Mech. | 2 |
| Error a: | |
| Block x Mech | 6 |
| Split-Plot D.F. | (12) |
| Fert. | 1 |
| Mech x Fert | 2 |
| Error b: | |
| Block x Fert | 3 |
| Block x Mech x Fert | 6 |
| Total | 23 |

Table 9. Means comparisons among microsite types: 2-way ANOVA followed by means separations with Duncan's Multiple Range Test

| Factor | Levels |
|---|---|
| Block | 1, 2, 3, 4 |
| Microsite Type | Undisturbed Soil Column,<br>Beds (previously disturbed),<br>Beds (previously undisturbed),<br>Interbed Rows |
| Fertilization | Fertilized, Unfertilized |
| Replications | 1, 2 |

| Source | Degrees of Freedom |
|---|---|
| Block | 3 |
| Micro | 3 |
| Fert. | 1 |
| Micro x Fert | 3 |
| Error: | (53) |
| Reps | 1 |
| Reps x Fert | 1 |
| Reps x Block | 3 |
| Reps x Micro | 3 |
| Block x Micro | 9 |
| Block x Fert | 3 |
| Micro x Reps x Block | 9 |
| Micro x Block x Fert | 9 |
| Micro x Reps x Fert | 3 |
| Reps x Blocks x Fert | 3 |
| Micro x Reps x Block x Fert | 9 |
| Total | 63 |

## RESULTS

Results for each gas ($CO_2$, $CH_4$, and $N_2O$) are presented separately with the following analyses: 1) means comparisons among plot treatments and microsite types, 2) regression analyses with the gas flux as the dependent variable, and 3) summary graphs of unaltered data from all sampling locations.

Analyses focus upon main effects, rather than interactions, as interactions were seldom significant (Table 10). Regression analysis was complicated by the fact that data for some variables was available only for a limited number of cases (Table 11). The variables soil temperature, moisture, total organic matter, TKN, and bulk density were available for all cases; and multiple linear regression models were developed for each gas flux, selecting from these independent variables with the aid of the stepwise selection procedure in SAS. Models were developed using other independent variables (denitrifying enzyme activity, nitrate, ammonia, and water-soluble organic carbon) as well, and discussed where appropriate.

### Linearity of Gas Evolution in Chambers

A constant flux rate across the soil surface during sampling is desirable, as it indicates that effects sometimes associated with chambers (such as soil warming or decreased diffusion due to high concentrations of a gas in the chamber headspace) either did not occur or did not affect the flux rate. A linear increase or decrease in gas concentration in a chamber with time is evidence that the flux rate remained constant. Evolution of all gases was linear throughout the study period. Typical examples of gas concentration versus time in the chambers are shown in Figure 8. $R^2$ values were calculated for every case and were consistently near 1.0. Lower $R^2$ values sometimes occurred at very low rates of

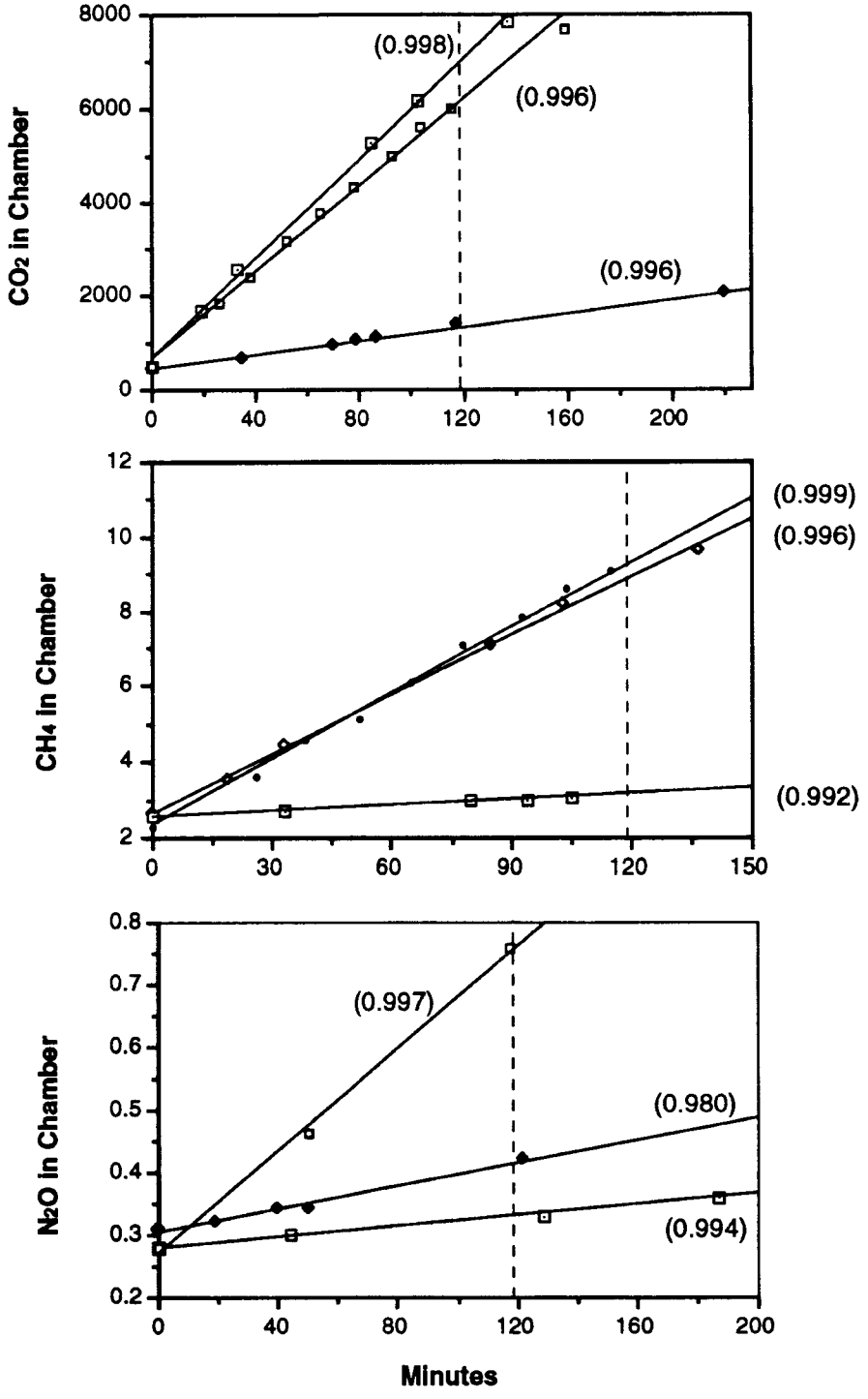



Figure 8. Typical examples of gas evolution in static chambers, showing linear rates of evolution. Some tests extended beyond the usual sampling period of 120 minutes (dashed line). $R^2$ values shown in parentheses.

gas evolution, as determination of gas concentrations was less precise at low levels.

Soil warming is often cited as a potential source of nonlinearity. Therefore, soil temperature was measured during some incubations—including those on the hottest days (approximately 40°C air temperature)—and found to remain constant while the chamber was in place.

Table 10. Summary of significance levels from analyses of variance in gas flux rates.

| | | *P-Value* | | | |
|---|---|---|---|---|---|
| *Effect* | | *Trial 1* | *Trial 2* | *Trial 3* | *Trial 4* |
| ***Mechanical Disturbances*** | | | | | |
| Field-Scale | $CO_2$ | 0.085* | 0.003*** | 0.014** | 0.011** |
| | $CH_4$ | 0.368 | 0.289 | 0.450 | 0.421 |
| | $N_2O$ | 0.507 | 0.001*** | 0.508 | 0.175 |
| Microsite Type | $CO_2$ | 0.000*** | 0.000*** | 0.000*** | 0.000*** |
| | $CH_4$ | 0.070* | 0.198 | 0.052* | 0.012** |
| | $N_2O$ | 0.544 | 0.000*** | 0.336 | 0.051* |
| ***Fertilization*** | | | | | |
| Field-Scale | $CO_2$ | 0.561 | 0.004*** | 0.035** | 0.118 |
| | $CH_4$ | 0.759 | 0.531 | 0.726 | 0.364 |
| | $N_2O$ | 0.956 | 0.629 | 0.452 | 0.745 |
| Microsite Type | $CO_2$ | 0.567 | 0.002*** | 0.044** | 0.192 |
| | $CH_4$ | 0.912 | 0.523 | 0.892 | 0.286 |
| | $N_2O$ | 0.940 | 0.486 | 0.489 | 0.737 |
| ***Mech x Fert Interaction*** | | | | | |
| Field-Scale | $CO_2$ | 0.826 | 0.083* | 0.698 | 0.138 |
| | $CH_4$ | 0.702 | 0.511 | 0.358 | 0.365 |
| | $N_2O$ | 0.206 | 0.760 | 0.290 | 0.144 |
| Microsite Type | $CO_2$ | 0.937 | 0.079* | 0.848 | 0.314 |
| | $CH_4$ | 0.515 | 0.941 | 0.815 | 0.172 |
| | $N_2O$ | 0.426 | 0.593 | 0.457 | 0.192 |

* $\alpha = 0.10$, ** $\alpha = 0.05$, *** $\alpha = 0.01$

Table 11. Variables available for regression analyses

| *Variable* | *Trial 1* | *Trial 2* | *Trial 3* | *Trial 4* | *Total Cases* |
|---|---|---|---|---|---|
| $CO_2$ | yes | yes | yes | yes | 256 |
| $CH_4$ | yes | yes | yes | yes | 256 |
| $N_2O$ | yes | yes | yes | yes | 256 |
| Temperature | yes | yes | yes | yes | 256 |
| Moisture | yes | yes | yes | yes | 256 |
| Org. Matter* | yes | yes | yes | yes | 256 |
| TKN* | yes | yes | yes | yes | 256 |
| Bulk Density* | yes | yes | yes | yes | 256 |
| DEA | yes | yes | no | yes | 192 |
| $NO_3^-$ | no | yes | no | yes | 128 |
| $NH_4^+$ | no | yes | no | yes | 128 |
| WSOC | no | no | no | yes | 64 |

* Measured after Trial 4 but assummed constant throughout study period. TKN = total Kjeldahl nitrogen. DEA = denitrifying enzyme activity. WSOC = water soluble organic carbon.

## Soil Characterization

### Organic Carbon

Total organic carbon was relatively insensitive to mechanical soil disturbance. The concentration of organic carbon in the interbed rows was 4.4 % versus an average of 6.8 % in the other microsite types, which did not differ significantly (Figure 9).

Water-soluble organic carbon was measured only in Trial 4, but was more sensitive to mechanical disturbance (Figure 9). The concentration of water-soluble organic carbon in beds with residual harvest damage was 37 % lower than in previously undamaged beds. This difference was significant at $\alpha = 0.05$. Neither the previously damaged or undamaged beds differed significantly from

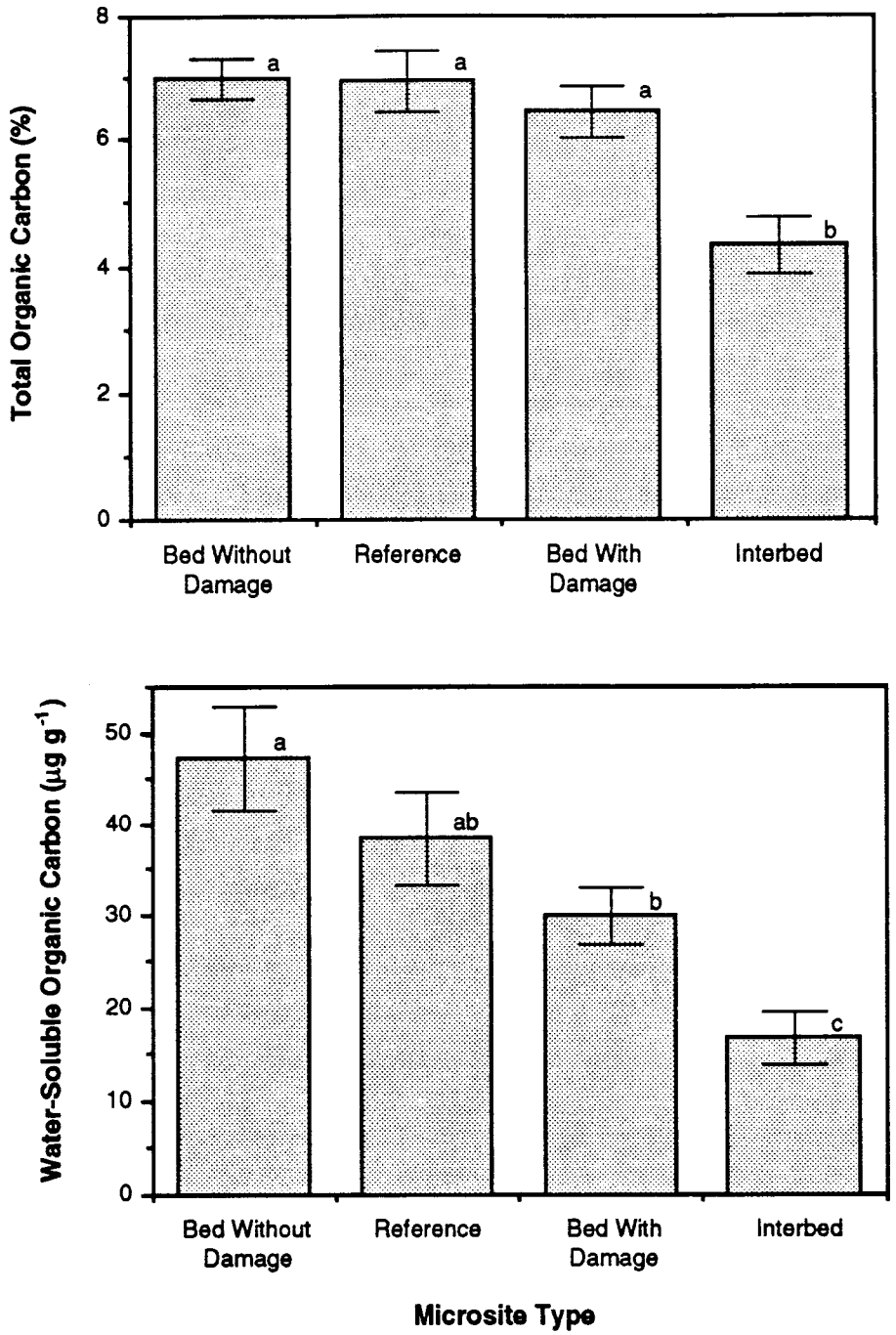



Figure 9. Total and water-soluble organic carbon from four microsite types: 1) reference (undisturbed), 2) beds without previous harvest damage, 3) beds with previous harvest damage, and 4) interbed rows. Means separated with F-test-protected Duncan's Multiple Range Test. Means with the same letter are not significantly different at $\alpha$=0.05. Determined from samples collected during Trial 4. Error bars are ± one standard error of the mean.

reference soils. Water-soluble organic carbon in interbed rows was lower than that of all other microsite types and 79 % lower than that of reference sites.

**Kjeldahl Nitrogen and Phosphorus**

Both of these measures were more sensitive to mechanical disturbance than total organic carbon. Concentrations of nitrogen and phosphorus in beds with residual damage was lower than previously undamaged beds (Figure 10). The concentration of nitrogen in interbed rows was 54 % lower than reference sites, and the concentration of phosphorus was 14 % lower than reference sites.

**Labile Nitrogen**

Measurements of nitrate and ammonia were available only for Trials 2 and 4. Both ammonia and nitrate were more abundant in Trial 2—immediately following fertilization—than in Trial 4. Nitrate concentrations in previously damaged and undamaged beds did not differ significantly (Figure 11). Nitrate in reference soils ranked lower than all other microsite types in both trial periods.

No significant differences in ammonia concentrations were detected among microsite types in either trial period (Figure 12). However, experiment-wise p-values were barely above the critical level of 0.10. In both trial periods, the rankings of ammonia concentrations closely mirrored those of nitrate. If an experiment-wise p-value of 0.11 is accepted as significant and comparison-wise error is controlled at 0.05, the concentration of ammonia in reference areas was significantly lower than all other microsite types in Trial 2.

**Denitrifying Enzyme Activity**

Dentrifying enzyme activity (DEA) was consistently low in interbed rows, but differences among other microsite types were marginal (Figure 13). Reference soils ranked lower than both bedded conditions in all trial periods, but differed significantly only from the beds with residual damage and only in Trial 4. Residual damage in beds had no apparent effect upon DEA.

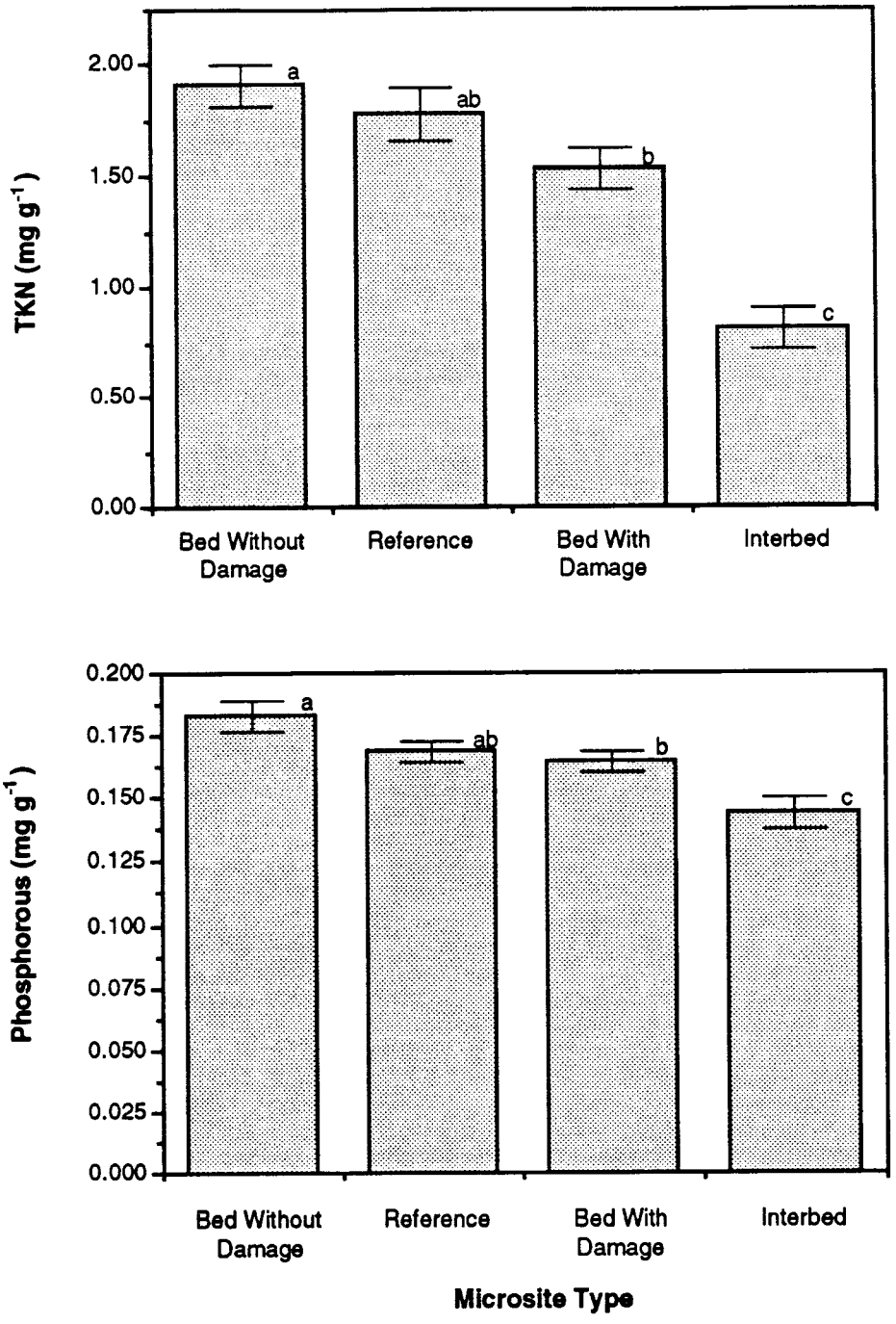



Figure 10. Kjeldahl nitrogen and phosphorous from four microsite types: 1) reference (undisturbed), 2) beds without previous harvest damage, 3) beds with previous harvest damage, and 4) interbed rows. Means separated with F-test-protected Duncan's Multiple Range Test. Means with the same letter are not significantly different at $\alpha$=0.05. Error bars are ± one standard error of the mean.

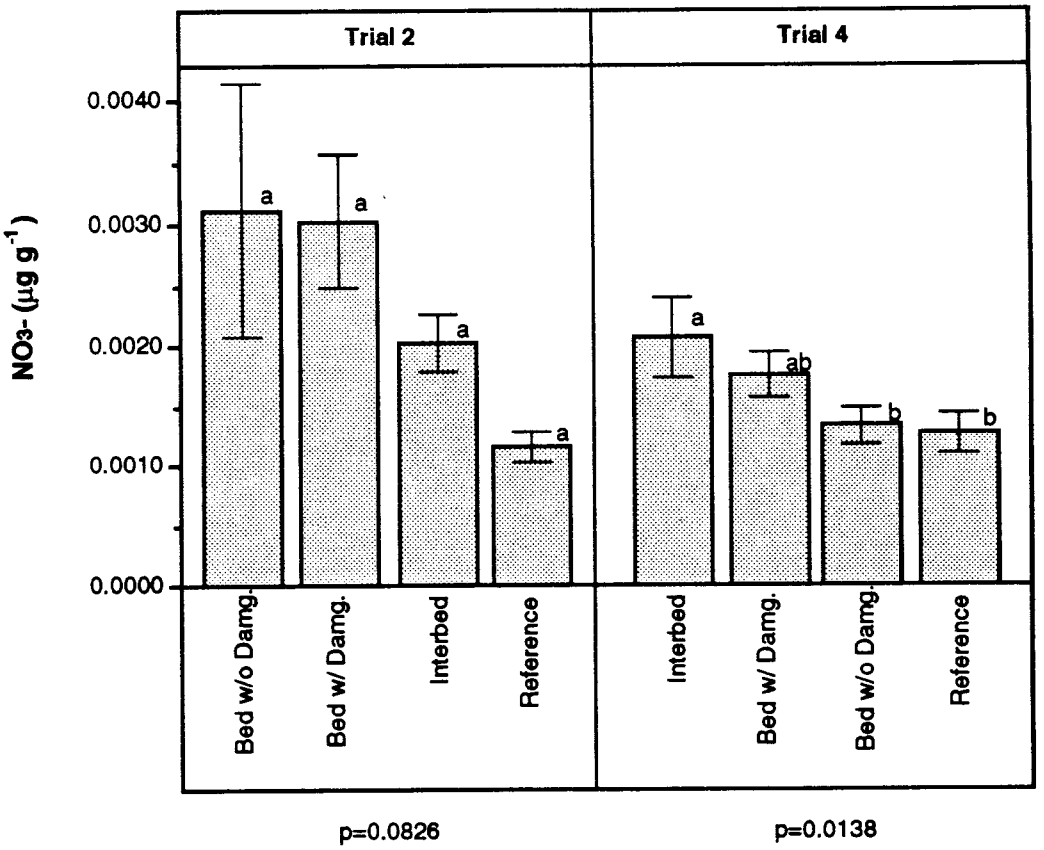



Figure 11. Nitrate concentration in soils from four microsite types: 1) reference (undisturbed), 2) beds without previous harvest damage, 3) beds with previous harvest damage, and 4) interbed rows. Means separated with F-test-protected Duncan's Multiple Range Test. Means with the same letter are not significantly different at alpha = 0.05. Error bars are ± one standard error of the mean.

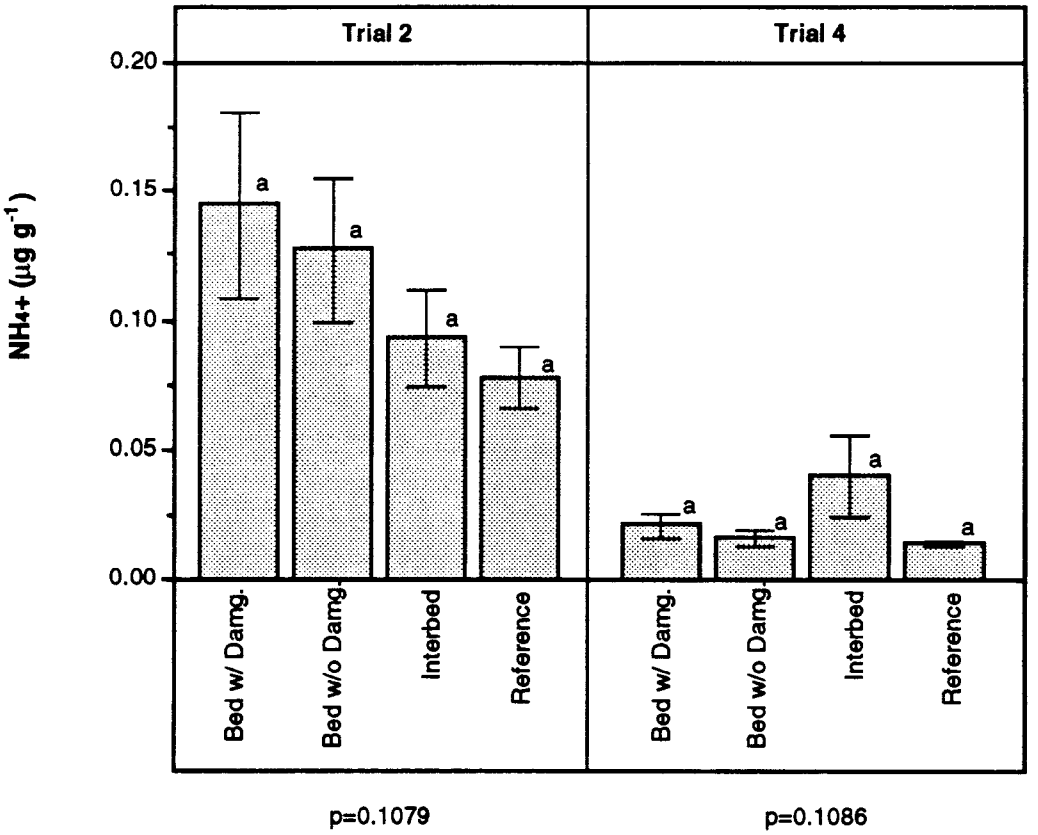



Figure 12. Ammonium concentrations in soils from four microsite types: 1) reference (undisturbed), 2) beds without previous harvest damage, 3) beds with previous harvest damage, and 4) interbed rows. Means separated with F-test-protected Duncan's Multiple Range Test. Means with the same letter are not significantly different at alpha = 0.05. Error bars are ± one standard error of the mean.

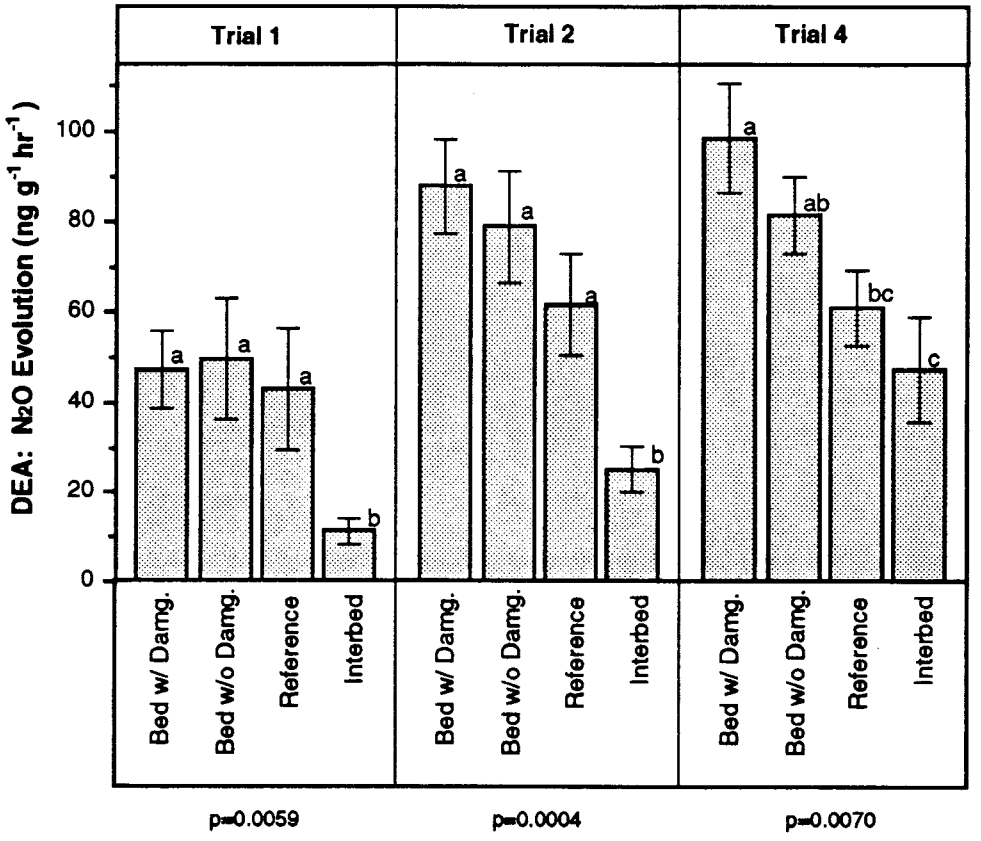



Figure 13. Denitrifying Enzyme Activity in microsite types: 1) reference (undisturbed), 2) beds without previous harvest damage, 3) beds with previous harvest damage, and 4) interbed rows. Means separated with F-test-protected Duncan's Multiple Range Test. Means with the same letter are not significantly different at alpha = 0.05. Error bars are ± one standard error of the mean.

**Bulk Density**

Two samples were collected from each reference plot to represent its four sampling microsites, so a balanced comparison of all microsite types was not possible. However, variance in bulk density was small. Bulk density in beds with residual harvest damage was significantly greater than previously undamaged beds ($p = 0.05$), when comparison was limited to these two microsite types (Figure 14). Bulk density of the A-horizon in reference soils averaged 0.99 $g \cdot cm^{-3}$, and bulk density in the interbed rows averaged 1.58 $g \cdot cm^{-3}$.

**Soil Moisture and Air-Filled Porosity**

Soil moisture was high in all trial periods during 1992 (Figure 15), varying between 35 and 45 % of total soil volume. Soil moisture was generally highest in reference microsites and lowest in previously undamaged beds. The combination of higher soil moisture and lower porosity in previously damaged beds resulted in values of air-filled porosity that were consistently below the critical level of 10 to 15 % (Blake and Page, 1948; Wyckoff and Botset, 1936), below which gas diffusion is severely restricted (Figure 16). In contrast, beds without residual damage were generally at the upper limit of— or above—the critical range.

## Gas Flux Rates

**Carbon Dioxide**

**Comparison of Treatments Applied to Whole Plots.** Rankings of the three mechanical soil treatments applied to whole plots (reference, bedding, and bedding over harvest damage) remained constant throughout the study period, with the most rapid efflux of $CO_2$ occurring on undisturbed reference plots and the slowest on bedded plots installed over skid trails (Figure 17). Though the rate of carbon fixation (net primary productivity) was unknown, bedded plots were conservative of carbon in terms of efflux. Bedded plots without residual

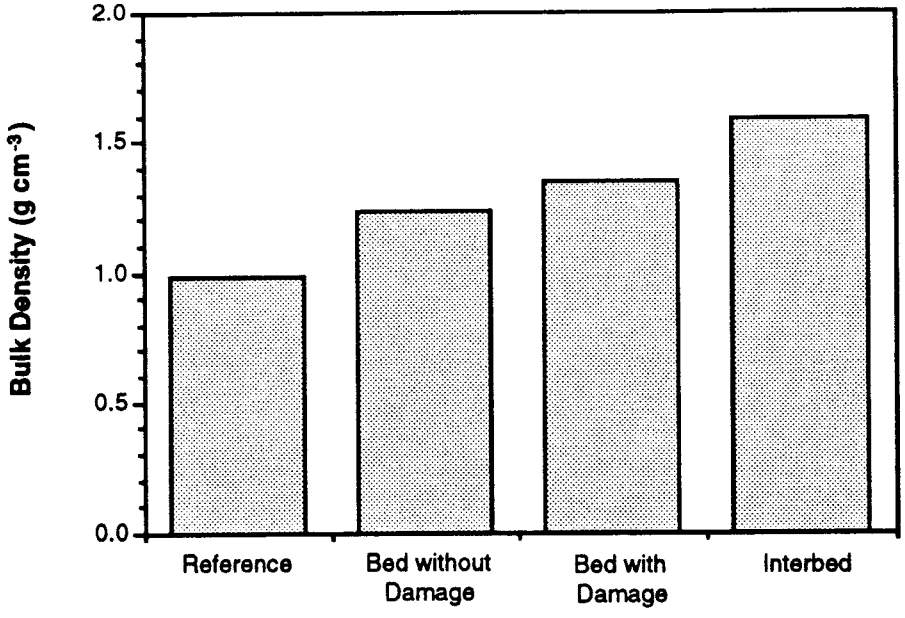



Figure 14. Bulk density of four microsite types: 1) reference (undisturbed), 2) beds without previous harvest damage, 3) beds with previous harvest damage, and 4) interbed rows.

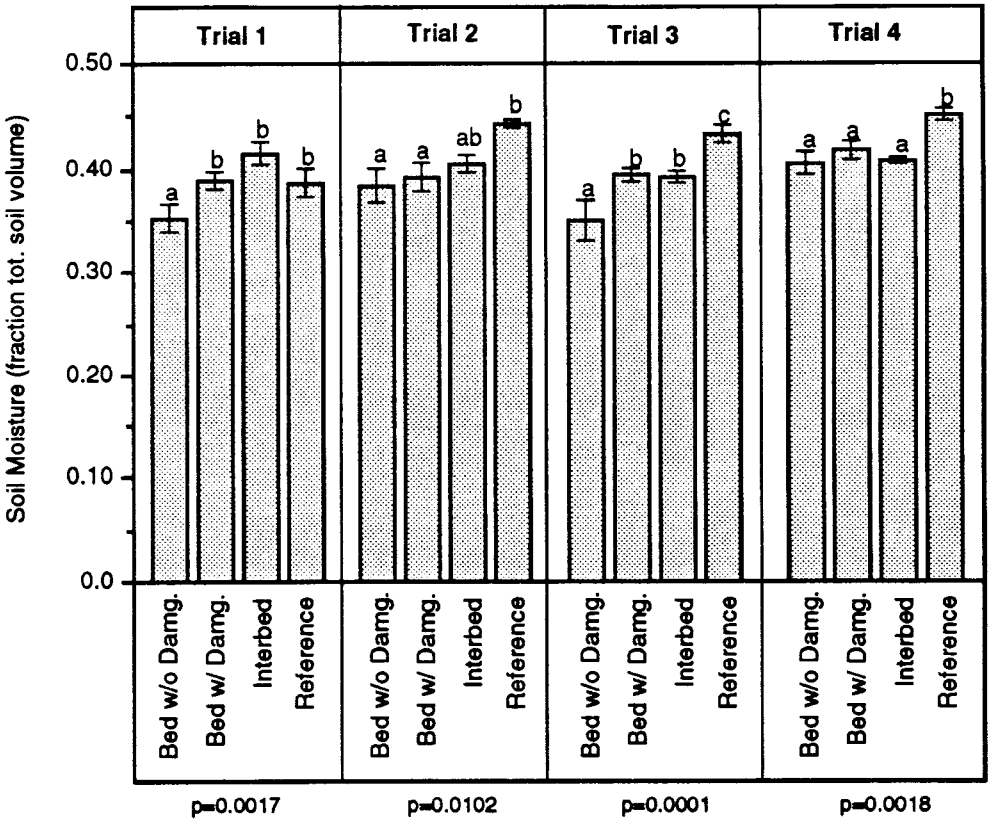



Figure 15. Soil moisture at four microsite types: 1) reference (undisturbed), 2) beds without previous harvest damage, 3) beds with previous harvest damage, and 4) interbed rows. Means separated with F-test-protected Duncan's Multiple Range Test at $\alpha$=0.05. Means with the same letter are not significantly different. Error bars are ± one error of the mean.

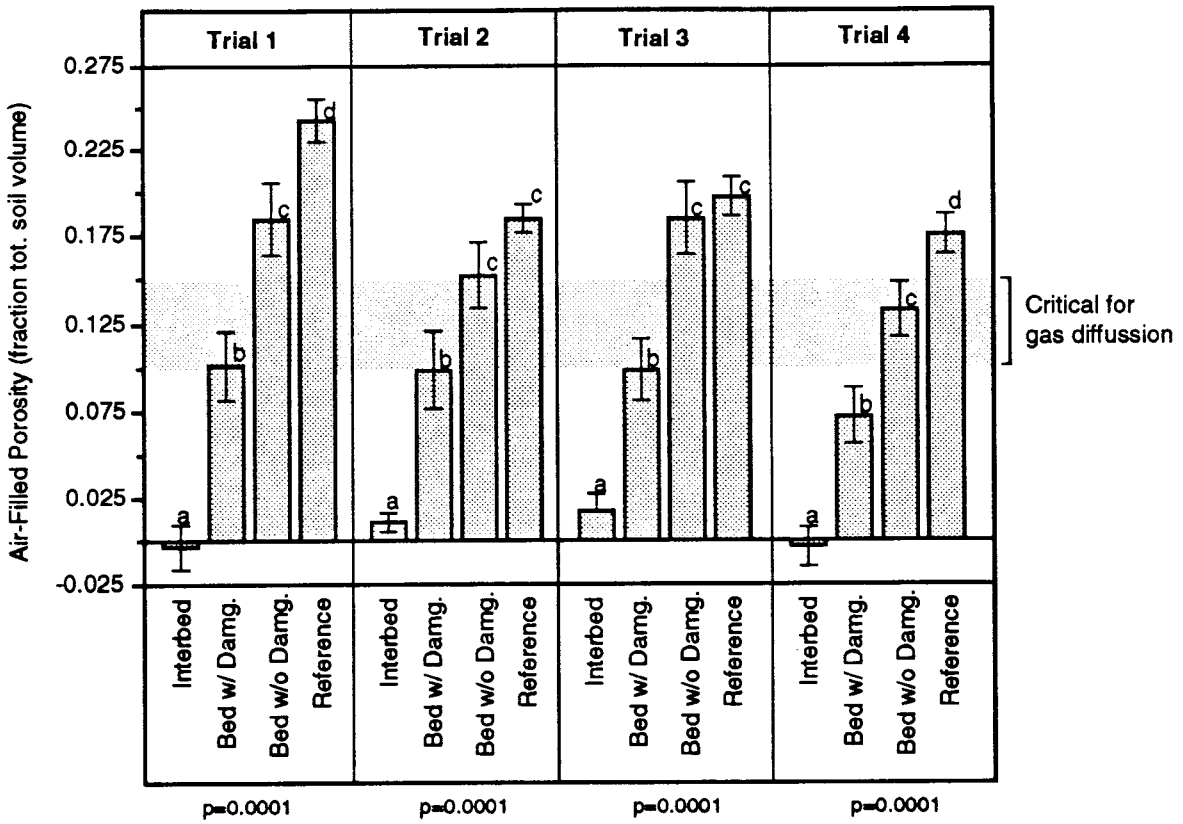



Figure 16. Air-filled porosity in four microsite types: 1) reference (undisturbed), 2) beds without previous harvest damage, 3) beds with previous harvest damage, and 4) interbed rows. Means separated with F-test-protected Duncan's Multiple Range Test at $\alpha$=0.05. Means with the same letter are not significantly different. Error bars are ± one error of the mean.

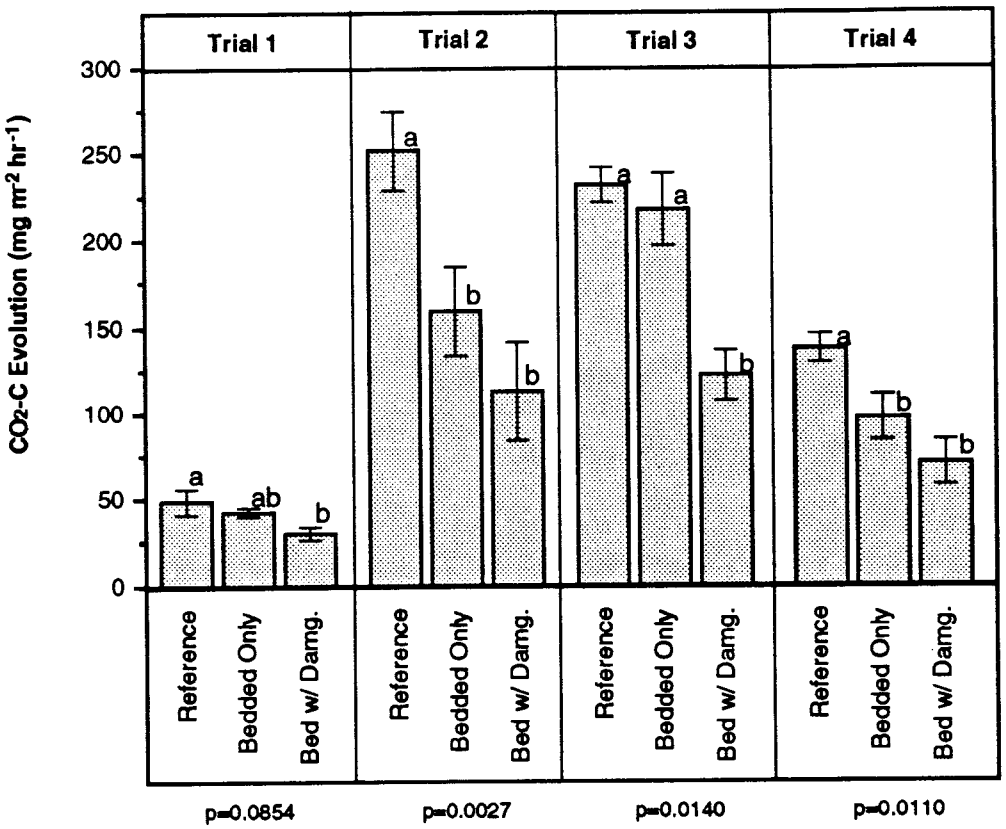



Figure 17. Carbon-dioxide evolution from plots subjected to three levels of mechanical soil disturbance: 1) reference (none), 2) bedding without previous harvest damage, and 3) bedding with previous harvest damage. Means were separated with F-test-protected Duncan's Multiple Range Test. Means with the same letter are not significantly different at alpha = 0.05. Error bars are ± one standard error of the mean.

disturbance were significantly lower than reference sites in two trial periods at $\alpha = 0.05$. Residual harvest damage reduced $CO_2$ evolution from bedded plots during two trial periods at $\alpha = 0.05$. Fertilization increased $CO_2$ evolution significantly at $\alpha = 0.05$ in the two trial periods immediately following fertilization (Figure 18).

**Comparison of Microsite Types.** Soil respiration ($CO_2$ evolution) was severely depressed in interbed rows, averaging only 15 % of undisturbed reference sites (Figure 19). Interbed rows occupied about 33 % of the surface area of bedded plots and were largely responsible for the reduced rate $CO_2$ efflux in bedded areas. $CO_2$ efflux from the actual beds (with no residual harvest damage) was similar to that of undisturbed reference sites. However, in beds installed over former skid trails, respiration was consistently depressed, averaging 38 % less than reference sites and 34 % less than beds without residual disturbance.

Fertilizer compensated partially for the negative effect of residual damage in the beds. Mitigation of residual damage by fertilizer was most pronounced in Trial 2, when a significant interaction ($p = 0.08$) occurred in analysis of microsite types between the main effects, mechanical disturbance and fertilization. During this trial period, $CO_2$ efflux from previously damaged beds that were unfertilized was about half that of similar beds that were fertilized (Figure 20). In Trials 3 and 4, the trend was similar but significant interactions did not result.

**Factors Related to $CO_2$ Evolution.** Temperature was examined first because of its strong influence upon rates of most biological processes. While temperature itself was certainly an important determinant, in this case, it also represented a seasonal effect. Low temperatures occurred only in winter and high temperatures in summer. For example, root respiration is likely to have

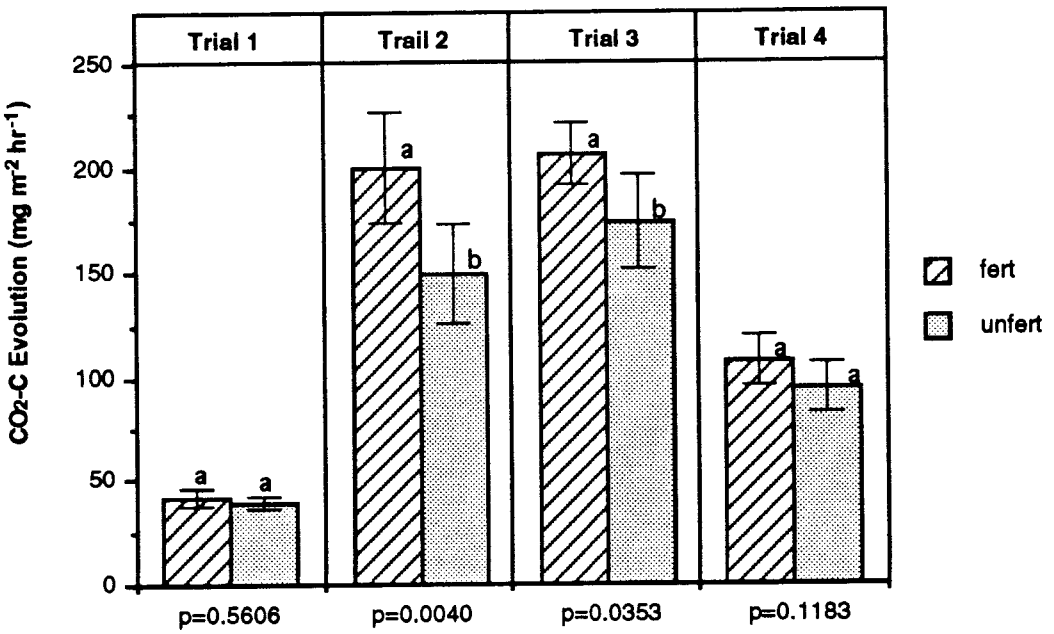



Figure 18. Carbon-dioxide evolution from fertilized and unfertilized areas. Fertilizer was applied after Trial 1. Means separated with F-test protected Duncan's Multiple Range Test. Means with the same letter are not significantly different at alpha = 0.05. Error bars are ± one standard error of the mean.

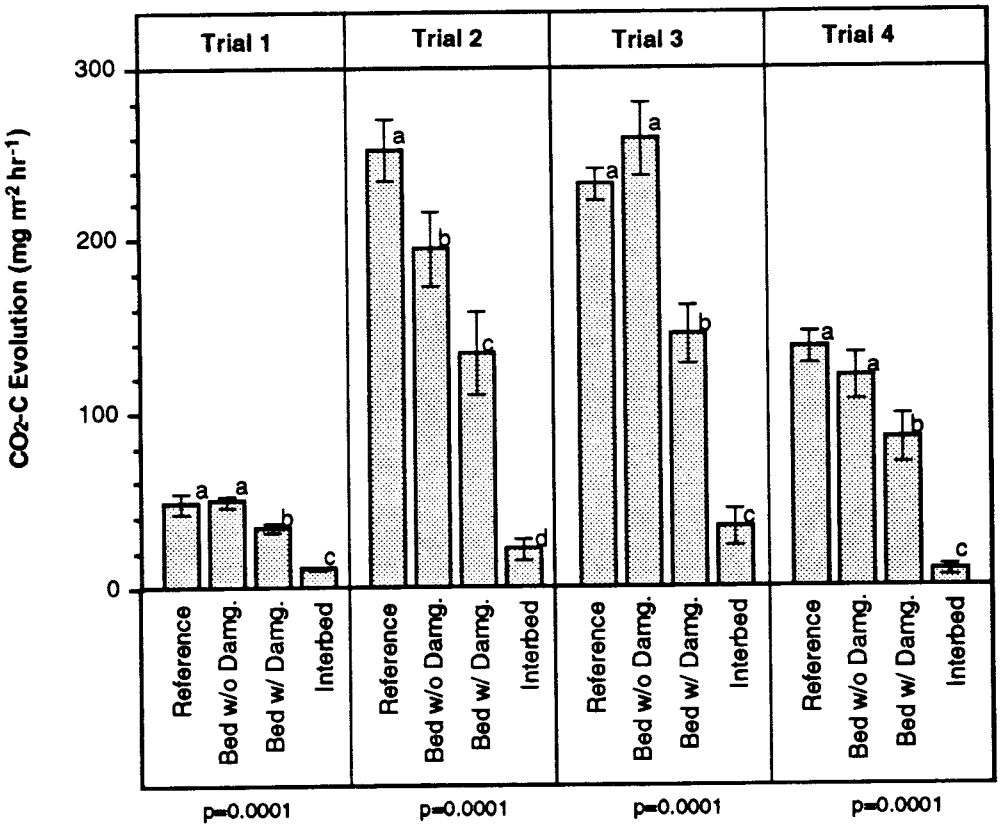



Figure 19. Carbon-dioxide evolution from four microsite types: 1) reference (undisturbed), 2) beds without previous harvest damage, 3) beds with previous harvest damage, and 4) interbed rows. Means separated with F-test-protected Duncan's Multiple Range Test at alpha = 0.05. Means with the same letter are not significantly different. Error bars are ± one standard error of the mean.

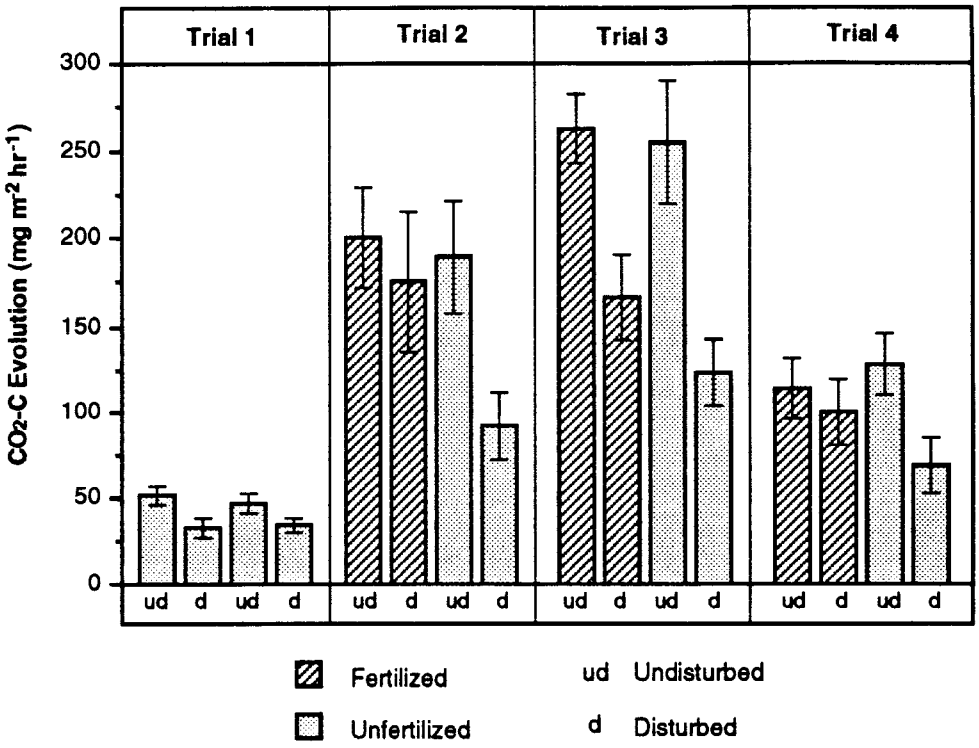



Figure 20. Carbon-dioxide evolution from beds alone: interaction between mechanical disturbance and fertilization. Comparison of microsite types revealed an interaction only in Trial 2, when fertilization appeared to have mitigated the deficit in gross biological activity due to previous disturbance. Fertilizer was applied between trials one and two. Error bars are ± one standard error of the mean.

contributed a larger fraction during the summer growing season, an effect due to dormancy of plants during winter, rather than a strict relation with temperature.

Using data from all microsite types, the relationship between $CO_2$ evolution and soil temperature was not well defined, but an exponential function seemed to clearly delineate the upper limit of soil respiration (Figure 21). $CO_2$ efflux from undisturbed reference sites was consistently near the upper limit for all sites and adhered to an exponential function (Figure 22). For the remaining microsite types, temperature accounted for less variation (Figures 22 and 23), indicating that other factors were limiting. These trends were quantified by exponential regressions, in which temperature accounted for 85 % of variation in $CO_2$ efflux from reference sites, but was less important with other microsite types (Figure 24).

Further accounting of the observed variation in flux rates was sought through multiple linear regression. Squaring of the temperature data resulted in the best transformation to account for its exponential relation with gas flux rates, though the transformed data were slightly more scattered, resulting in slight reductions in $r^2$ values over exponential regresssions. Examination of fluxes from individual microsites (Figures 25 and 26) revealed that their relative performance was often similar between trial periods, suggesting that some lasting soil characteristics associated with treatments were important sources of variation in flux rates.

A model was fitted using variables available for all individual gas flux measurements in the study (a total of 256 measurements). When all microsite types were included, the best model accounted for 60 % of variation and included the variables total nitrogen, temperature, bulk density, organic matter and moisture (Table 12). Temperature and total nitrogen each accounted for

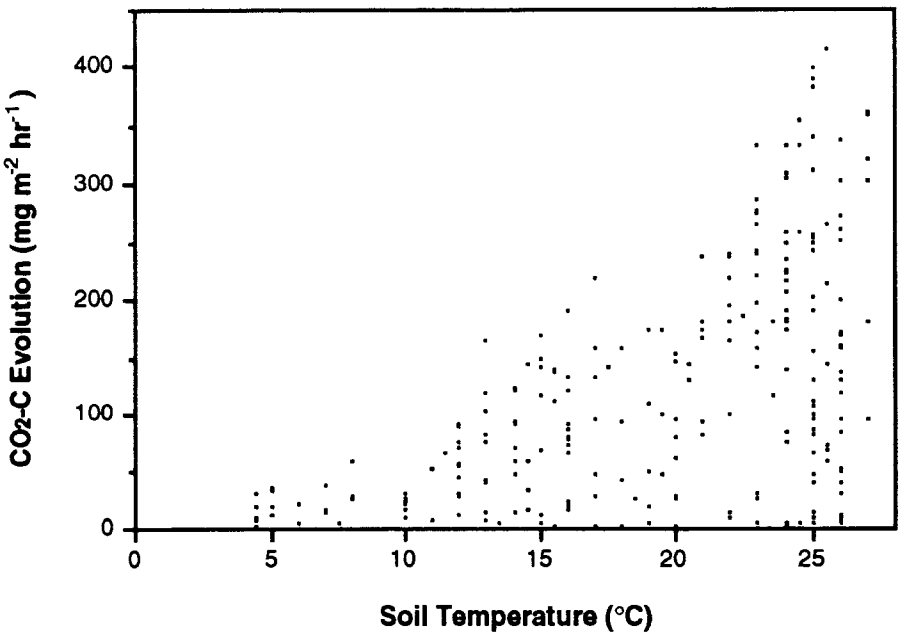



Figure 21. Carbon-dioxide evolution vs. temperature for all microsite measurements (256) over the entire study period. Four microsite types are included: 1) reference, 2) beds without previous disturbance, 3) beds with previous damage, and 4) interbed rows. The upper limit of $CO_2$ evolution appears to be defined by an exponential function.

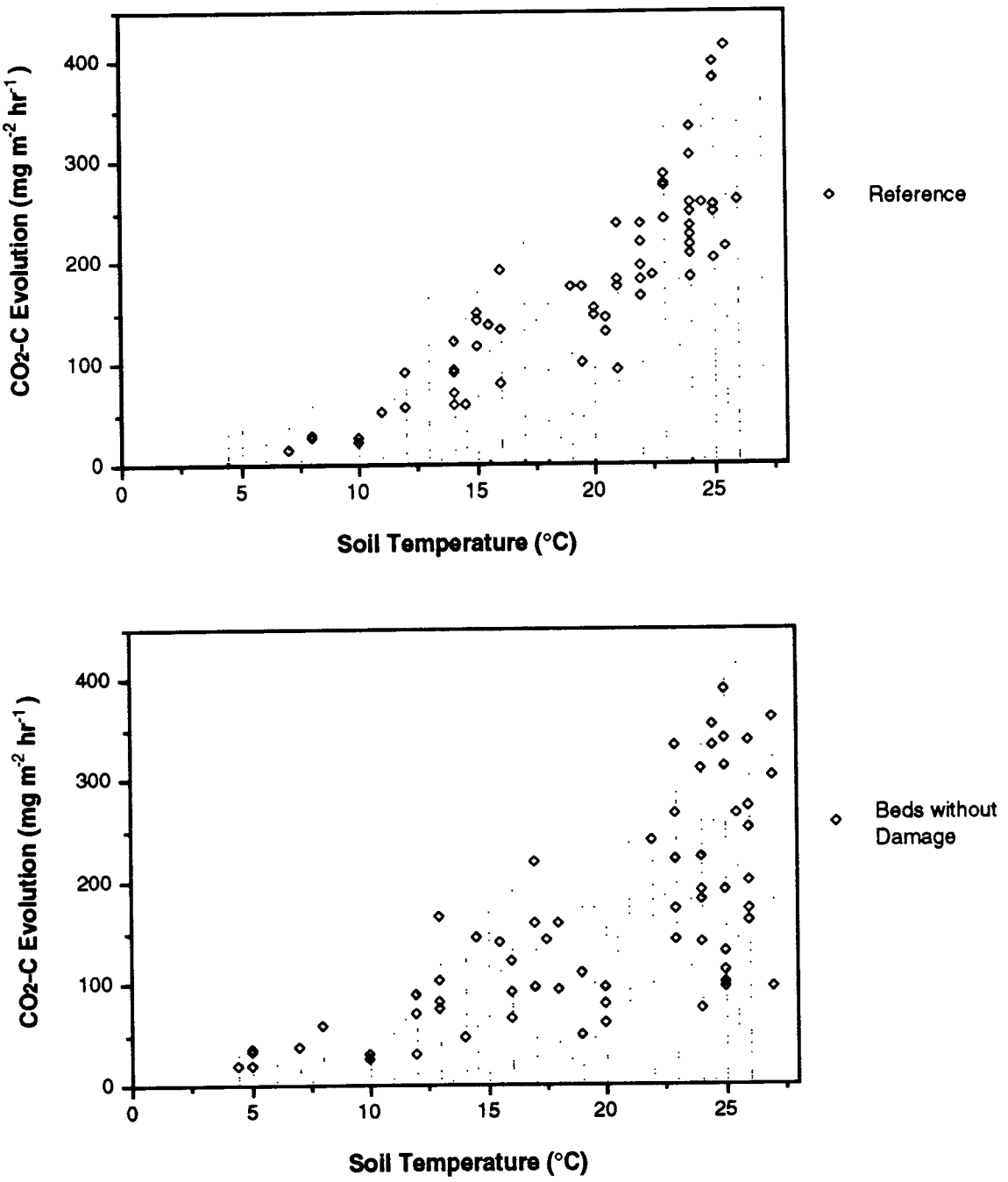



Figure 22. Carbon-dioxide evolution vs. temperature for all microsite measurements over the entire study period, with individual microsite types highlighted. Each microsite type is represented by 64 points.

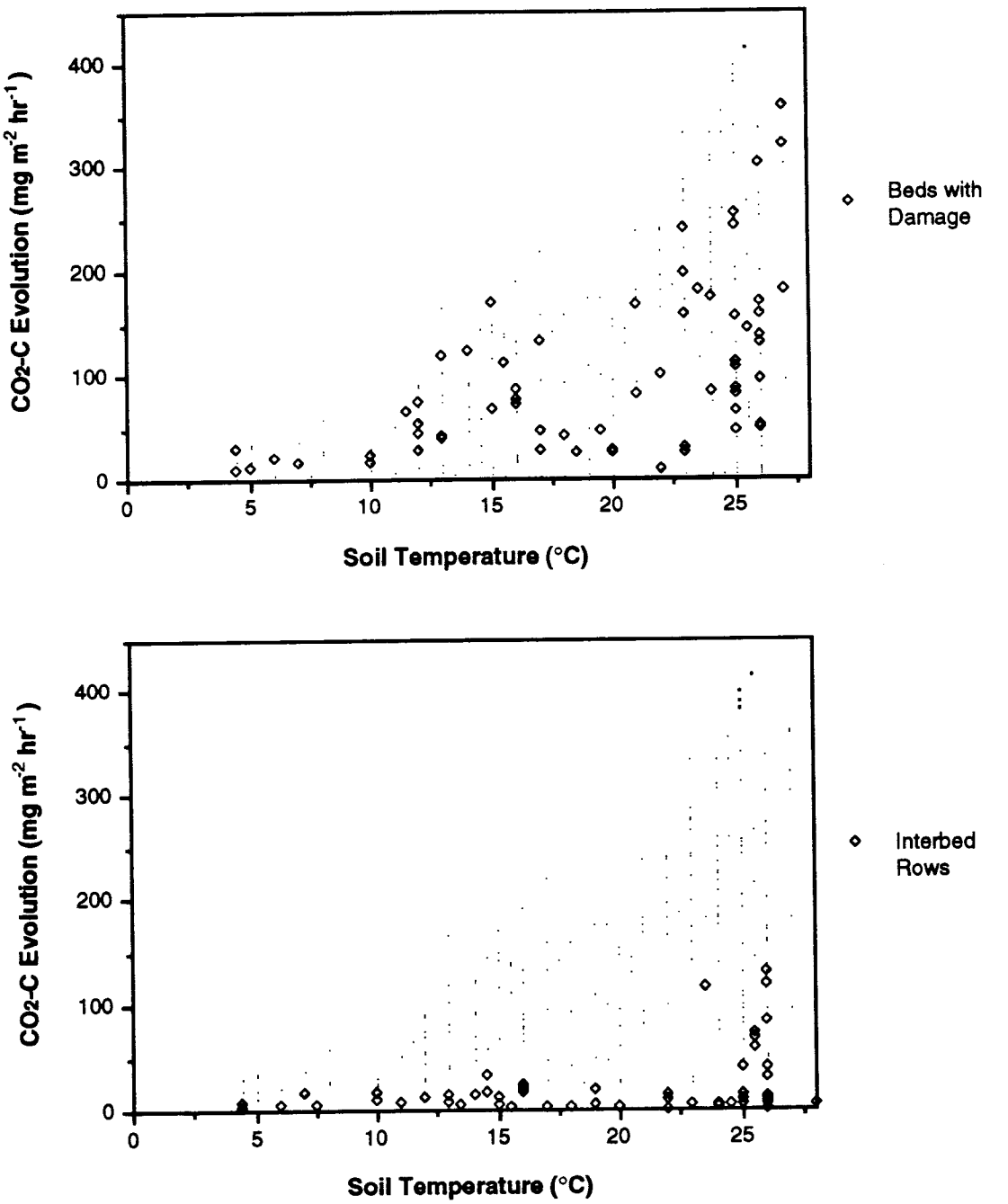
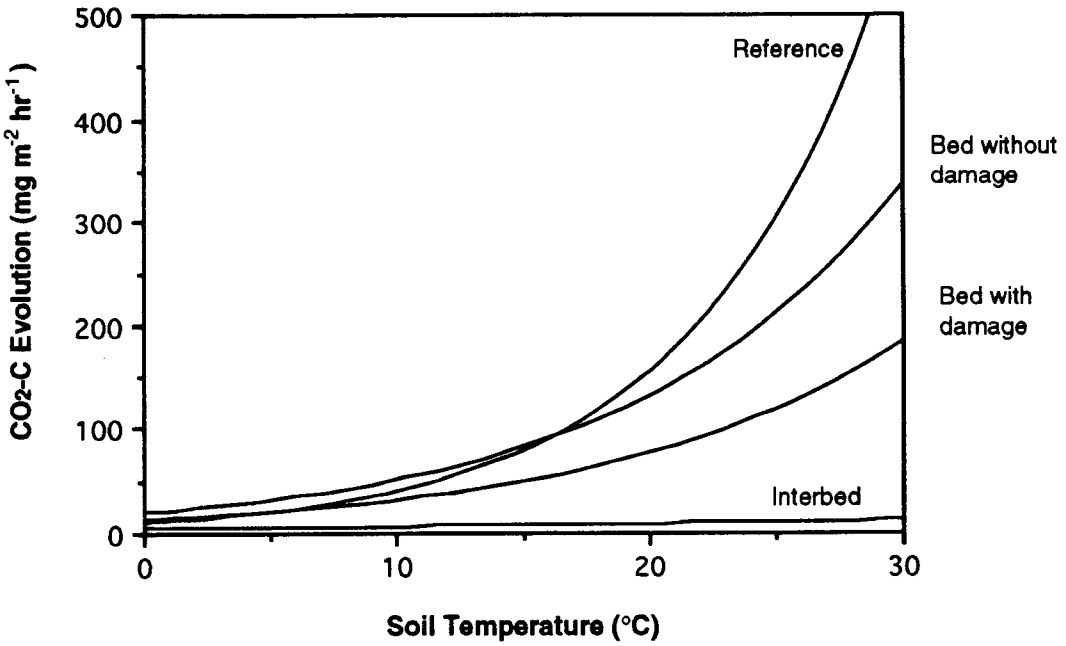



Figure 23. Carbon-dioxide evolution vs. temperature for all microsite measurements over the entire study period, with individual microsite types (beds with previous disturbance and interbed rows) highlighted. Each microsite type is represented by 64 points.




| Microsite Type | Exponential Regression of CO2 vs. Temperature |
|---|---|
| Reference | y = 10.406 * 10^(5.8476e-2x) R^2 = 0.850 |
| Bed without Damage | y = 19.753 * 10^(4.1050e-2x) R^2 = 0.642 |
| Bed with Damage | y = 13.072 * 10^(3.8227e-2x) R^2 = 0.413 |
| Interbed | y = 4.5840 * 10^(1.4062e-2x) R^2 = 0.022 |

Figure 24. Exponential regressions of carbon dioxide vs. temperature for four microsite types. Temperature accounts for 85% of the variation in carbon-dioxide evolution in the reference areas over the course of the study period. Reference areas define the upper limit of soil respiration. Sources of variation other than temperature are apparent in the bedded areas with and without previous damage. Carbon-dioxide evolution from soils in the interbed rows showed little relation to temperature, presumably because factors such as lack of organic matter and frequent inundation strongly limited soil respiration.

Figure 25. Carbon-dioxide evolution from individual microsites on the Goldsboro (moderately well drained) soil series. Each bar, from left to right, represents the same microsite in every trial period.

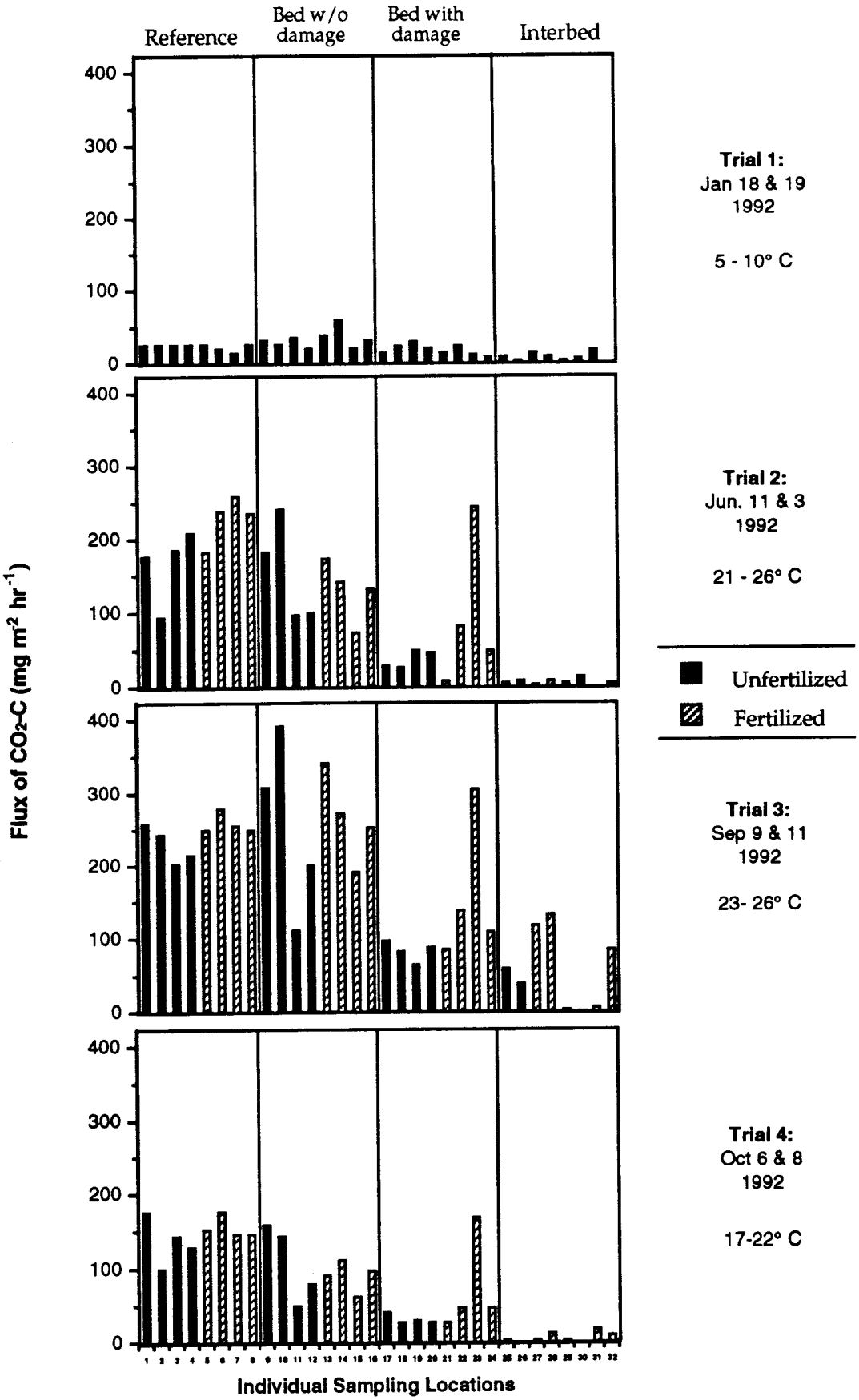
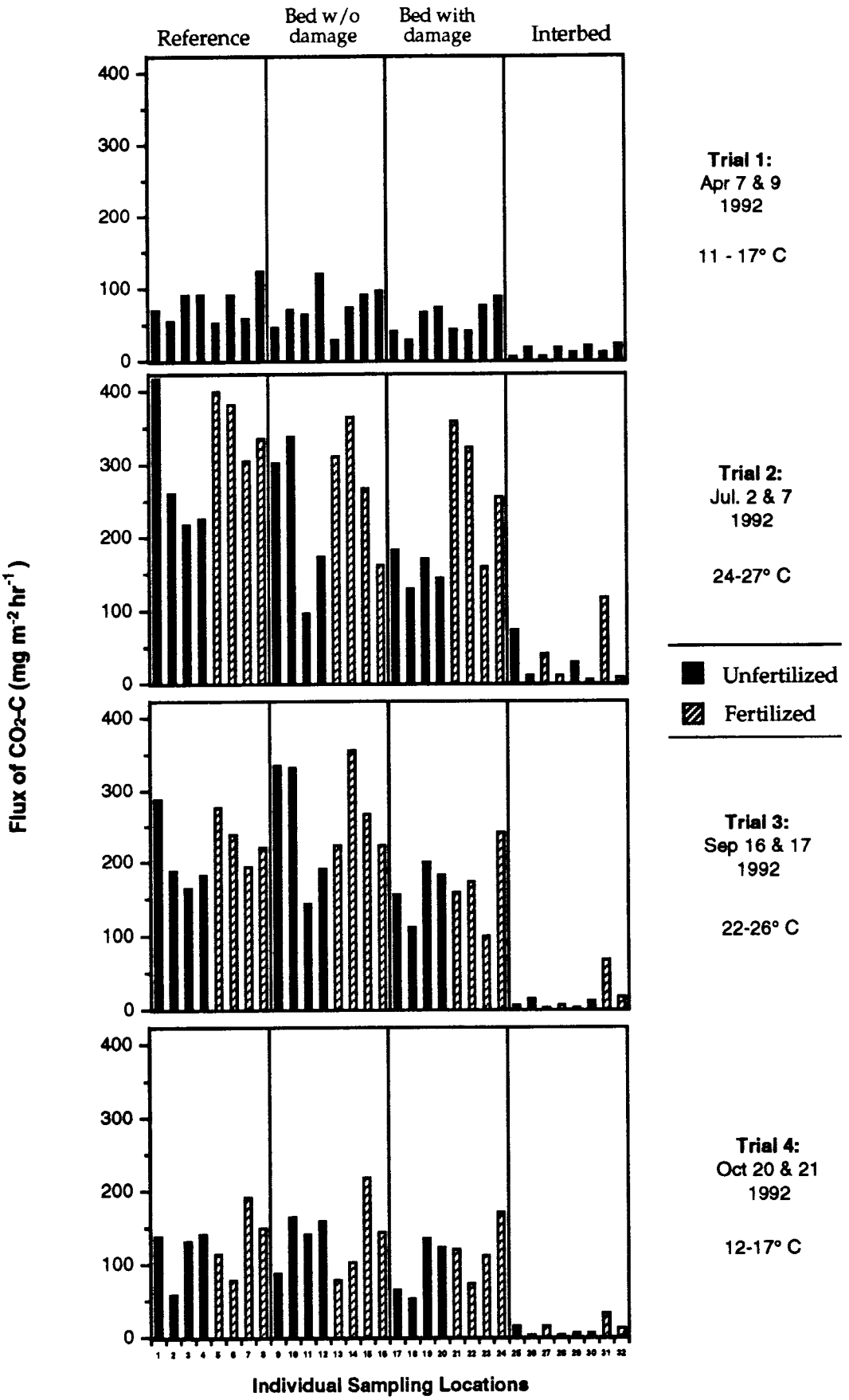

Reference
Bed w/o damage
Bed with damage
Interbed
Flux of $CO_2$-C (mg $m^{-2}$ $hr^{-1}$)
0
100
200
300
400
Individual Sampling Locations
Trial 1:
Jan 18 & 19
1992
5 - 10° C
Trial 2:
Jun. 11 & 3
1992
21 - 26° C
Trial 3:
Sep 9 & 11
1992
23- 26° C
Trial 4:
Oct 6 & 8
1992
17-22° C
Unfertilized
Fertilized

Figure 26. Carbon-dioxide evolution from individual microsites on the Bethera (poorly drained) soil series. Each bar, from left to right, represents the same microsite in every trial period.


Reference
Bed w/o damage
Bed with damage
Interbed
400
300
200
100
0
Flux of $CO_2$-C (mg $m^{-2}$ $hr^{-1}$)
Trial 1:
Apr 7 & 9
1992
11 - 17° C
Trial 2:
Jul. 2 & 7
1992
24-27° C
Unfertilized
Fertilized
Trial 3:
Sep 16 & 17
1992
22-26° C
Trial 4:
Oct 20 & 21
1992
12-17° C
Individual Sampling Locations

about 1/4 of the variation, with bulk density, organic matter, and moisture accounting for minor fractions (Table 13). Inclusion of other variables did not substantially improve the fit when the model was fitted across all microsite types.

Table 12. Multiple linear regression for dependent variable $CO_2$ flux from all microsite types.

R-squared = 0.5987

| | DF | Sum of Squares | Mean Square | F | P > F |
|---|---|---|---|---|---|
| Regression | 5 | 1566160.78 | 313232.16 | 74.60 | 0.0001 |
| Error | 250 | 1049705.16 | 4198.82 | | |
| Total | 255 | 2645865.93 | | | |

| Variable | Parameter Estimate | Std. Error | Type II Sum of Sqrs. | F | P > F |
|---|---|---|---|---|---|
| Intercept | 149.0564 | 51.2819 | 35473.1069 | 8.45 | 0.0040 |
| $Temp^2$ | 0.2383 | 0.0186 | 689078.7106 | 164.11 | 0.0001 |
| Moisture | -1.6587 | 0.7537 | 20336.7756 | 4.84 | 0.0287 |
| OM | -16.5450 | 4.7647 | 50628.4223 | 12.06 | 0.0006 |
| TKN | 113.8073 | 17.7201 | 173195.2654 | 41.25 | 0.0001 |
| BD | -106.8841 | 20.8470 | 110374.2712 | 26.29 | 0.0001 |

When regression analysis was confined to specific microsite types, temperature was the most important determinant of $CO_2$ flux rates (Table 13), indicating that, in the previous regression, total nitrogen had represented primarily differences among microsite types. Much of the variation was still unaccounted for, so regression was confined further to cases for which the additional variables ($NO_3$, $NH_4$, DEA, and water soluble-soluble organic carbon) were available. These variables each accounted for a small fractions of the variation (5 % or less) and were not highly significant.

Table 13. Multiple linear regressions for the dependent variable $CO_2$ flux by microsite type.

| Microsite Type (model parameters) | Variable | Parameter Estimate | Partial $r^2$ | Prob > F |
|---|---|---|---|---|
| ALL TYPES | INTERCEPT | 149.05637 | -- | 0.0040 |
| n = 256 | $TEMP^2$ | 0.23827 | 0.2522 | 0.0001 |
| $r^2$ = 0.60 | MOISTURE | -1.65868 | 0.0078 | 0.0287 |
| p = 0.0001 | ORG. MATTER | -16.54498 | 0.0279 | 0.0006 |
| | TKN | 113.80733 | 0.2720 | 0.0001 |
| | BULK DENSITY | -106.88408 | 0.0388 | 0.0001 |
| REFERENCE | INTERCEPT | -21.74721 | -- | 0.6446 |
| n = 64 | $TEMP^2$ | 0.46433 | 0.7826 | 0.0001 |
| $r^2$ = 0.78 | MOIS | -1.96160 | 0.0062 | 0.1334 |
| p = 0.0001 | OM | 13.5983 | 0.0509 | 0.0001 |
| BED w/o damage | INTERCEPT | 24.06700 | -- | 0.6974 |
| n = 64 | $TEMP^2$ | 0.32784 | 0.5052 | 0.0001 |
| $r^2$ = 0.64 | MOIS | -3.77775 | 0.0658 | 0.0016 |
| p = 0.0001 | TKN | 72.47123 | 0.0654 | 0.0017 |
| BED w/ damage | INTERCEPT | -39.96428 | -- | 0.2562 |
| n = 64 | $TEMP^2$ | 0.20999 | 0.3506 | 0.0001 |
| $r^2$ = 0.5129 | OM | -30.66535 | 0.1098 | 0.0005 |
| p = 0.0001 | TKN | 163.57045 | 0.0526 | 0.0001 |
| INTERBED | INTERCEPT | 1.90700 | -- | 0.7930 |
| n = 64 | $TEMP^2$ | 0.04119 | 0.1071 | 0.0083 |
| $r^2$ = 0.1071 | | | | |
| p = 0.0083 | | | | |

## Methane

**Comparison of Treatments Applied to Whole Plots.** No significant differences in methane flux rate were detected among mechanical or fertilization treatments applied to plots (Figures 27 and 28). However, flux from bedded plots ranked consistently higher than that of reference sites in all trial periods.

**Comparison of Microsite Types.** Comparison of microsite types demonstrated that methane evolution from beds and interbed rows was always greater than or equal to that of reference sites (Figure 29). However, differences were of marginal significance. Few differences were detected at the 0.05 significance level., but overall p-values were 0.07 or less in three of the four trial periods. Methane evolution from beds without residual harvest damage was

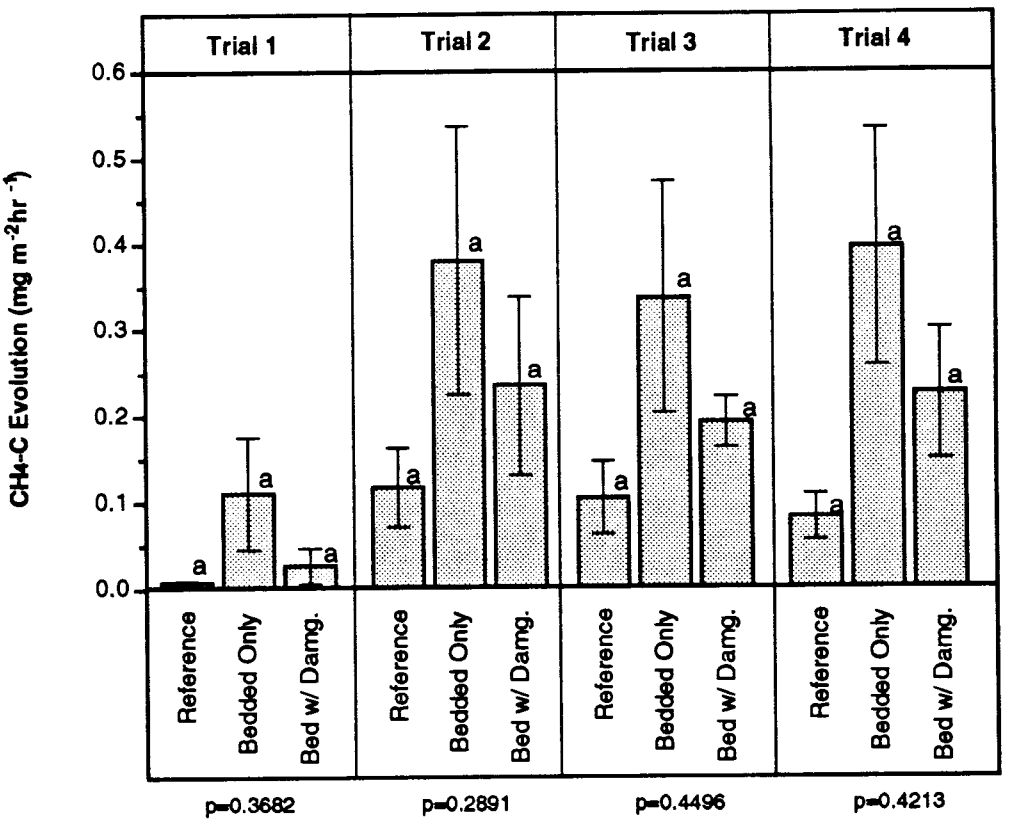



Figure 27. Methane evolution from plots subjected to three levels of mechanical soil disturbance: 1) reference (none), 2) bedding without previous harvest damage, and 3) bedding with previous harvest damage. Means separated with F-test-protected Duncan's Multiple Range Test. Means with the same letter are not significantly different at alpha = 0.05. Error bars are ± the standard error of the mean.

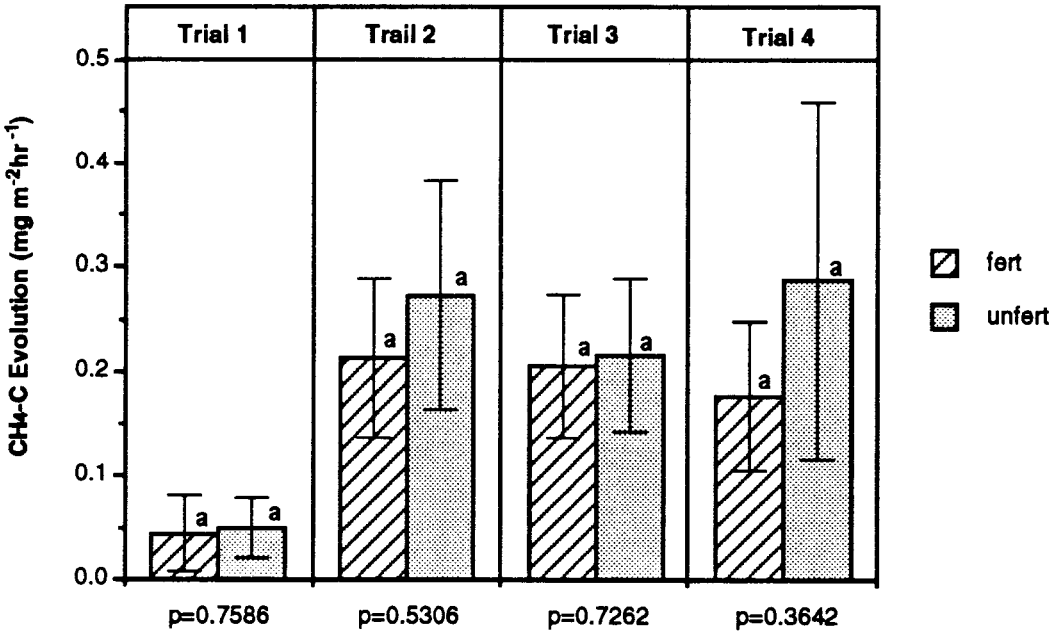



Figure 28. Methane evolution from fertilized and unfertilized plots. Fertilizer was applied after Trial 1. Means separated with F-test-protected Duncan's Multiple Range Test. Means with the same letter are not significantly different at alpha = 0.05. Error bars are ± one standard error of the mean.

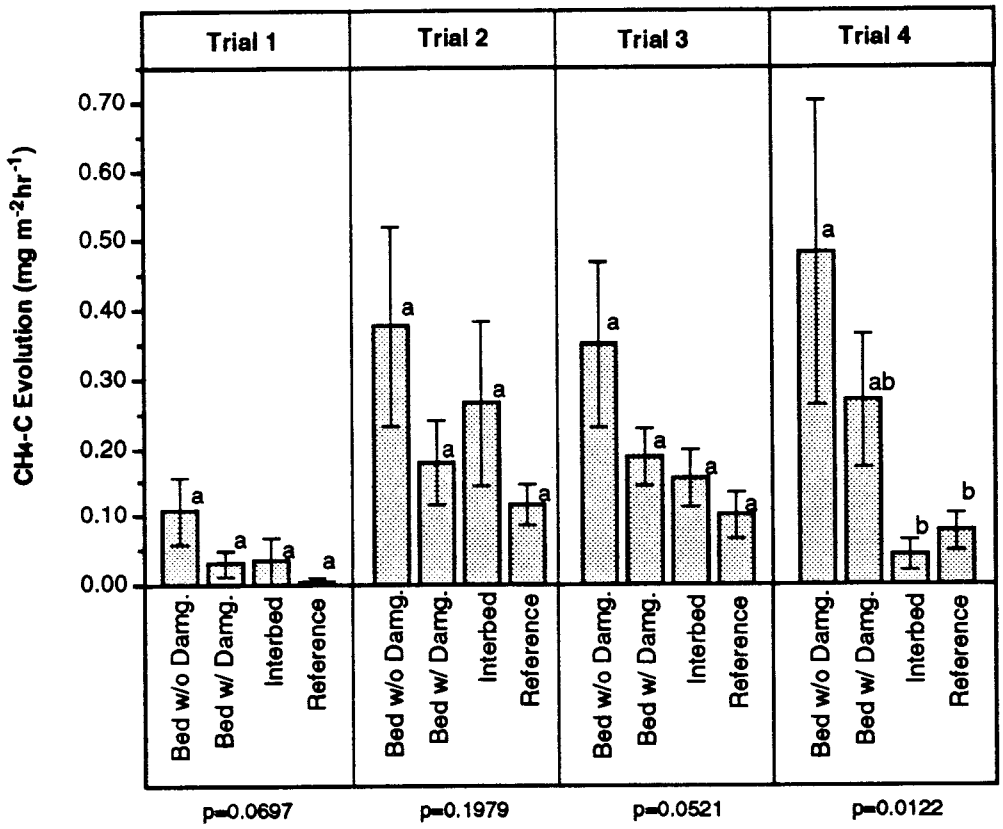



Figure 29. Methane evolution from four microsite types: 1) reference (undisturbed), 2) beds without previous harvest damage, 3) beds with previous harvest damage, and 4) interbed rows. Means separated with F-test-protected Duncan's Multiple Range Test at $\alpha$=0.05. Means with the same letter are not significantly different. Error bars are ± one error of the mean.

significantly higher than reference sites in three of four trial periods at $\alpha$=0.1 (Figure 29). Means from interbed rows were usually ranked higher than reference sites but did not differ statistically. Methane flux from beds with residual damage averaged only 40 % that of previously undamaged beds. This effect was significant at $\alpha$ = 0.10 in two trial periods.

**Factors Related to $CH_4$ Evolution.** Temperature was positively correlated with the rate of methane evolution (Figure 30), but there was no temperature-related upper bound on the flux rate, suggesting that factors other than temperature were usually limiting. Reference sites were less responsive to temperature increase. Because methane evolution tended to occur in "spikes" at scattered locations (Figures 31 and 32) and activity at many microsites was negligible, the correlation with temperature was weaker than visual inspection of the data might have suggested.

Selecting from the variables available for all gas flux measurements in the study (with the aid of the stepwise procedure in SAS) the best model accounted for about 20 % of the variation in methane flux rates (Table 14) and included the variables moisture, temperature, organic matter, total nitrogen, and bulk density. No single variable accounted for more than 7 % of the variation when analyzing across all microsite types (Table 15). In the limited number of cases for which the variable denitrifying enzyme activity was available (192), it accounted for about 7 % of variation and was highly significant, but its inclusion did not substantially improve the overall fit of the model. Factors related to physical condition, nutrient status, and enzyme activity, together, generally accounted for more variation than temperature.

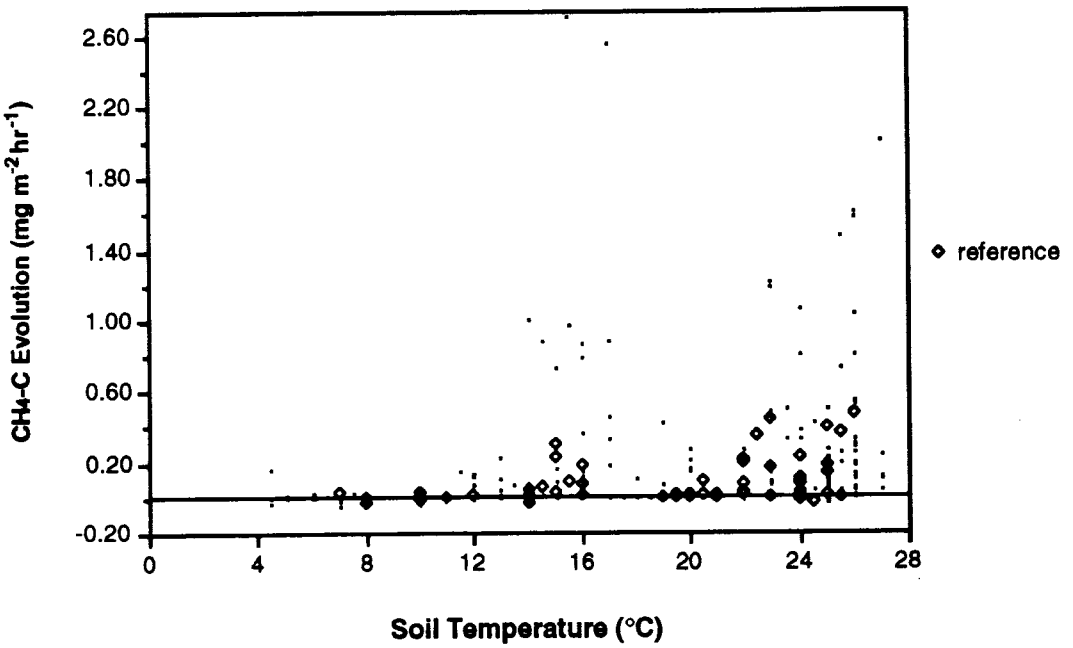



Figure 30. Methane evolution versus temperature for all microsite measurements over the entire study period, with reference microsites highlighted.

Figure 31. Methane evolution from individual microsites on the Goldsboro (moderately well drained) soil series. Each bar, from left to right, represents the same microsite in every trial period.

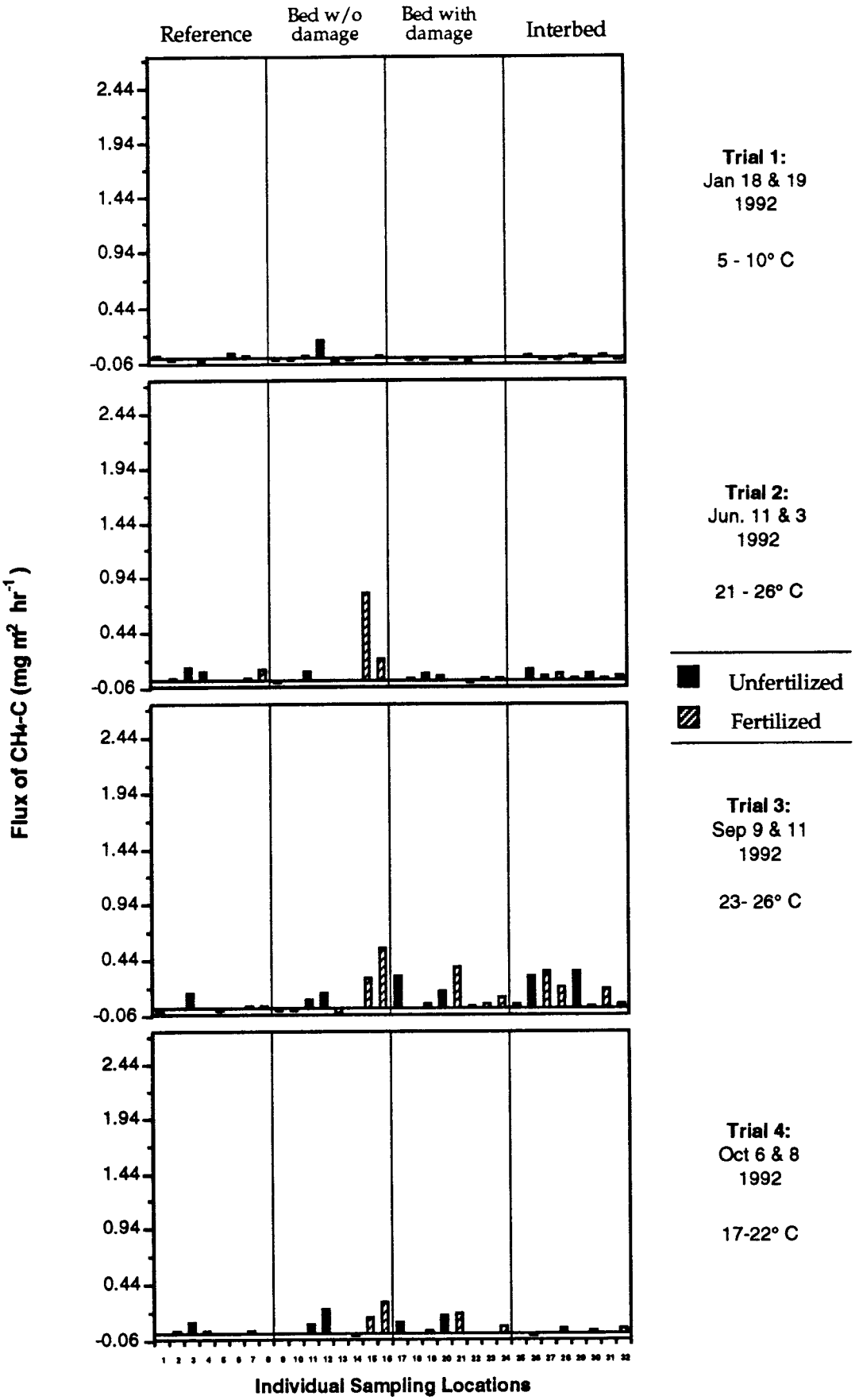
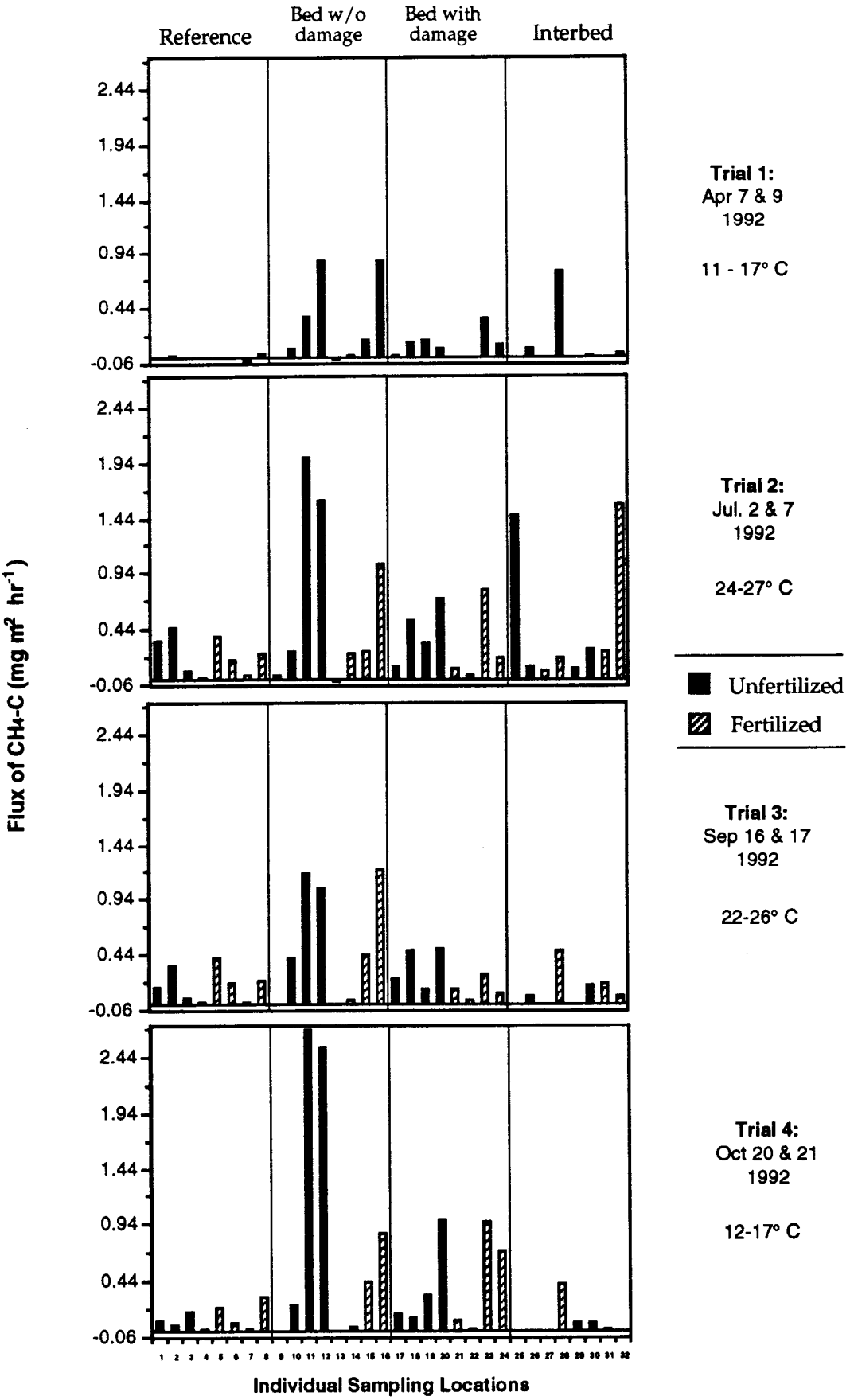

Reference
Bed w/o damage
Bed with damage
Interbed
2.44
1.94
1.44
0.94
0.44
-0.06
Flux of CH4-C (mg m2 hr-1)
Trial 1:
Jan 18 & 19
1992
5 - 10° C
Trial 2:
Jun. 11 & 3
1992
21 - 26° C
Unfertilized
Fertilized
Trial 3:
Sep 9 & 11
1992
23- 26° C
Trial 4:
Oct 6 & 8
1992
17-22° C
Individual Sampling Locations

Figure 32. Methane evolution from individual microsites on the Bethera (poorly drained) soil series. Each bar, from left to right, represents the same microsite in every trial period.


Reference
Bed w/o damage
Bed with damage
Interbed
2.44
1.94
1.44
0.94
0.44
-0.06
Flux of CH4-C (mg m2 hr-1)
Trial 1: Apr 7 & 9 1992
11 - 17° C
Trial 2: Jul. 2 & 7 1992
24-27° C
Unfertilized
Fertilized
Trial 3: Sep 16 & 17 1992
22-26° C
Trial 4: Oct 20 & 21 1992
12-17° C
Individual Sampling Locations

**Flux Rates from Individual Microsites.** Additional interpretations are possible through inspection of the unaltered data from individual microsites. This experiment was not designed to test differences among the Bethera and Goldsboro soil series, though there were two replications on each. However, rates of methane flux from the Bethera series, which is more poorly drained, were often an order of magnitude larger (Figures 31 and 32). Frequently, no methane evolution was detected on the Goldsboro soil.

Also apparent is that tillage and mechanical disturbance (in both beds and interbed rows) caused scattered extreme peaks or "spikes" in the rate of methane evolution, which did not occur at reference sites. The erratic spatial distribution may have obscured differences among means in statistical analyses. Note also that methane evolution did not respond as consistently to changes in temperature and season as carbon dioxide.

Table 14. Multiple linear regression for dependent variable $CH_4$ flux from all microsite types.

R-squared = 0.1981

| | DF | Sum of Squares | Mean Square | F | P > F |
|---|---|---|---|---|---|
| Regression | 5 | 6.9105919 | 1.3821184 | 12.35 | 0.0001 |
| Error | 250 | 27.9780030 | 0.1119120 | | |
| Total | 255 | 34.8885949 | | | |

| Variable | Parameter Estimate | Std. Error | Type II Sum of Sqrs. | F | P > F |
|---|---|---|---|---|---|
| Intercept | -1.267723 | 0.267577 | 2.512047 | 22.45 | 0.0001 |
| Moisture | 0.008618 | 0.003321 | 0.753844 | 6.74 | 0.0100 |
| Temp | 0.022047 | 0.003899 | 3.578411 | 31.98 | 0.0001 |
| OM | -0.118602 | 0.024615 | 2.598236 | 23.22 | 0.0001 |
| TKN | 0.531365 | 0.091540 | 3.770826 | 33.69 | 0.0001 |
| BD | 0.246577 | 0.107585 | 0.587864 | 5.25 | 0.0227 |

Table 15. Multiple linear regressions for the dependent variable $CH_4$ flux by microsite type.

| Microsite Type (model parameters) | Variable | Parameter Estimate | Partial $r^2$ | Prob > F |
|---|---|---|---|---|
| ALL TYPES | INTERCEPT | -1.26772 | -- | 0.0001 |
| n = 256 | TEMP | 0.00862 | 0.0258 | 0.0100 |
| $r^2$ = 0.20 | MOISTURE | 0.02205 | 0.0640 | 0.0001 |
| p = 0.0001 | ORG. MATTER | -0.11860 | 0.0714 | 0.0001 |
| | TKN | 0.53136 | 0.0200 | 0.0001 |
| | BULK DENSITY | 0.24658 | 0.0168 | 0.0227 |
| REFERENCE | INTERCEPT | 0.02571 | --* | 0.8144 |
| n = 64 | $TEMP^2$ | 0.00022 | 0.128 | 0.0010 |
| $r^2$ = 0.36 | TKN | 0.11632 | 0.169 | 0.0001 |
| p = 0.0001 | BULK DENSITY | -0.24631 | 0.042 | 0.0171 |
| BED w/o damage | INTERCEPT | -0.77233 | -- | 0.1103 |
| n = 64 | MOIS | 0.04151 | 0.2274 | 0.0001 |
| $r^2$ = 0.39 | OM | -0.31489 | 0.0468 | 0.0002 |
| p = 0.0001 | TKN | 0.92360 | 0.1190 | 0.0011 |
| BED w/ damage | INTERCEPT | -0.96766 | -- | 0.0542 |
| n = 64 | TKN | 0.35618 | 0.1001 | 0.0034 |
| $r^2$ = 0.14 | BULK DENSITY | 0.43179 | 0.0405 | 0.0951 |
| p = 0.0098 | | | | |
| INTERBED | INTERCEPT | -0.87702 | -- | 0.0106 |
| n = 64 | $TEMP^2$ | 0.00039 | 0.0949 | 0.0104 |
| $r^2$ = 0.19 | MOISTURE | 0.02023 | 0.0902 | 0.0117 |
| p = 0.0019 | | | | |

* Partial $r^2$ values in this cell calculated separately in another program and do not sum exactly to the overall model $r^2$.

## Nitrous Oxide

**Comparison of Treatments Applied to Whole Plots.** Nitrous-oxide fluxes were below the limit of detection in undisturbed reference areas throughout the study, and at all microsites in trials one and four. Small variations in flux values about zero were random and in a range that could be due errors in gas chromatography. Nitrous-oxide evolution was detected only in the bedded areas (with and without residual harvest damage).

During Trial 2, $N_2O$ evolution occurred throughout the bedded areas and was 160 % higher in those with residual harvest damage (Figure 33). Though

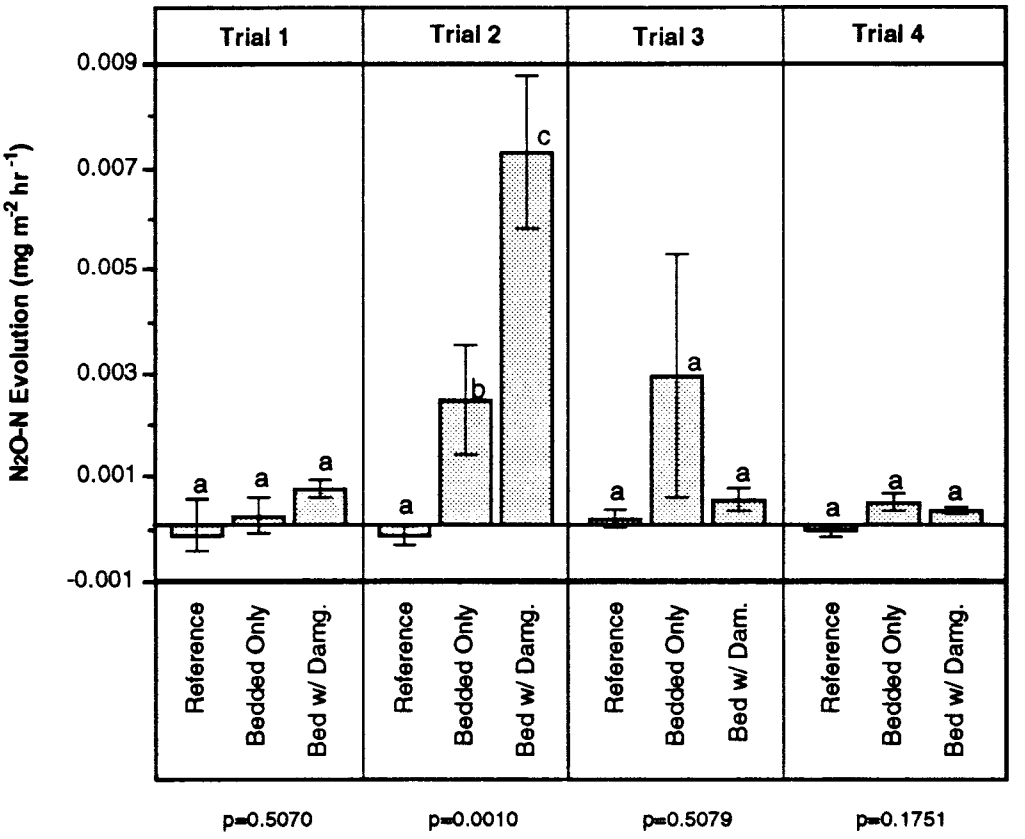



Figure 33. Nitrous-oxide evolution from plots subjected to three levels of mechanical soil disturbance: 1) reference (none), 2) bedding without previous harvest damage, and 3) bedding with previous harvest damage. Means separated with F-test-protected Duncan's Multiple Range Test. Means with the same letter are not significantly different at alpha = 0.05. Error bars are ± one standard error of the mean.

$N_2O$ evolution was detected in Trial 3 also, it was limited to activity at two or three isolated microsites (Figures 37 and 38) and did not result in significant differences among treatment means. Fertilization had no apparent effect on $N_2O$ efflux (Figure 34). The peak in activity during Trial 2 occurred in both fertilized and unfertilized areas.

**Comparison of Microsite Types.** In the beds themselves, $N_2O$ efflux was detected only during Trial 2 and was 210% higher in beds with residual harvest damage (Figure 35). Rates in beds without residual harvest damage and interbed rows were not significantly greater than reference areas, even though $N_2O$ evolution in reference areas was below the limit of detection. In Trial 3, there was no difference among microsite types. The larger mean of interbed microsites was due to two isolated high values.

**Factors Related to $N_2O$ Evolution.** Temperature was positively correlated with the rate of $N_2O$ evolution (Figure 36) but weakly so because activity was negligible at most microsites throughout the study. No temperature-related upper bound on flux rate was apparent. Selecting from the variables available for all gas flux measurements in the study, the best model accounted for only 6 % of the variation and included the variables temperature and bulk density (Table 16). When analysis was confined to Trial 2 (when $N_2O$ evolution was widely observed) and additional variables were included, $NO_3^-$ accounted for 9 % of the variation and was positively related to flux rate (Table 17).

**$N_2$:$N_2O$-N Ratio.** $N_2O$ evolution was detected in 28 of 96 incubations in either the first or second stage. Of the 28 cases in which denitrification activity was detected, $N_2$ evolution averaged 4.1 times greater than $N_2O$-N, and the $N_2$:$N_2O$-N ratio ranged from 0.23 to 16 (Figure 39).

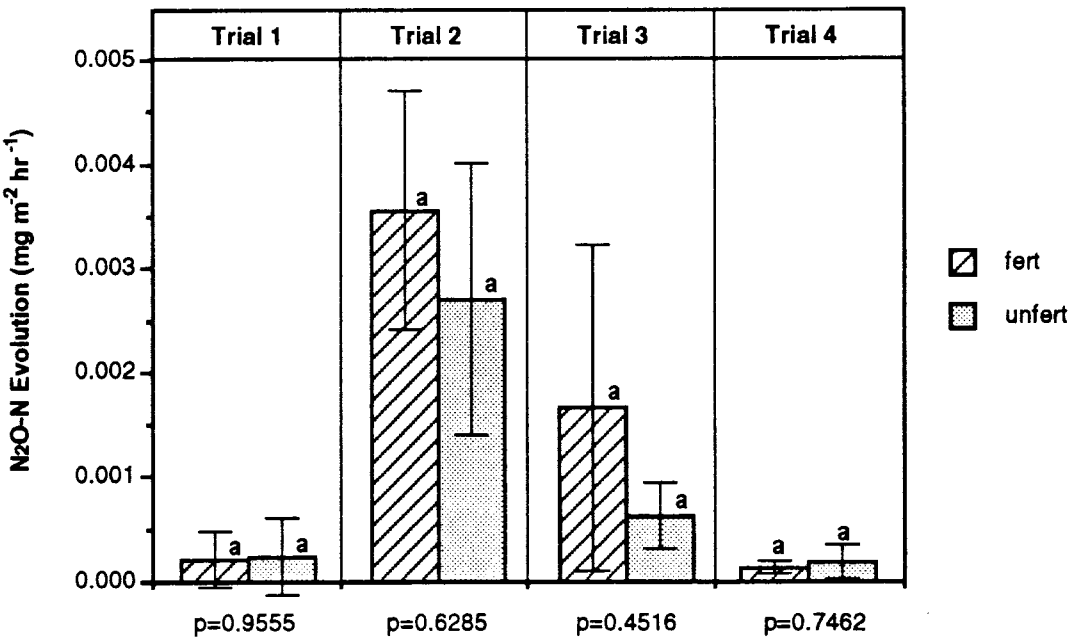



Figure 34. Nitrous-oxide evolution from fertilized and unfertilized plots. Fertilizer was applied after Trial 1. Means separated with F-test-protected Duncan's Multiple Range Test. Means with the same letter are not significantly different at alpha=0.05. Error bars are ± one standard error of the mean.

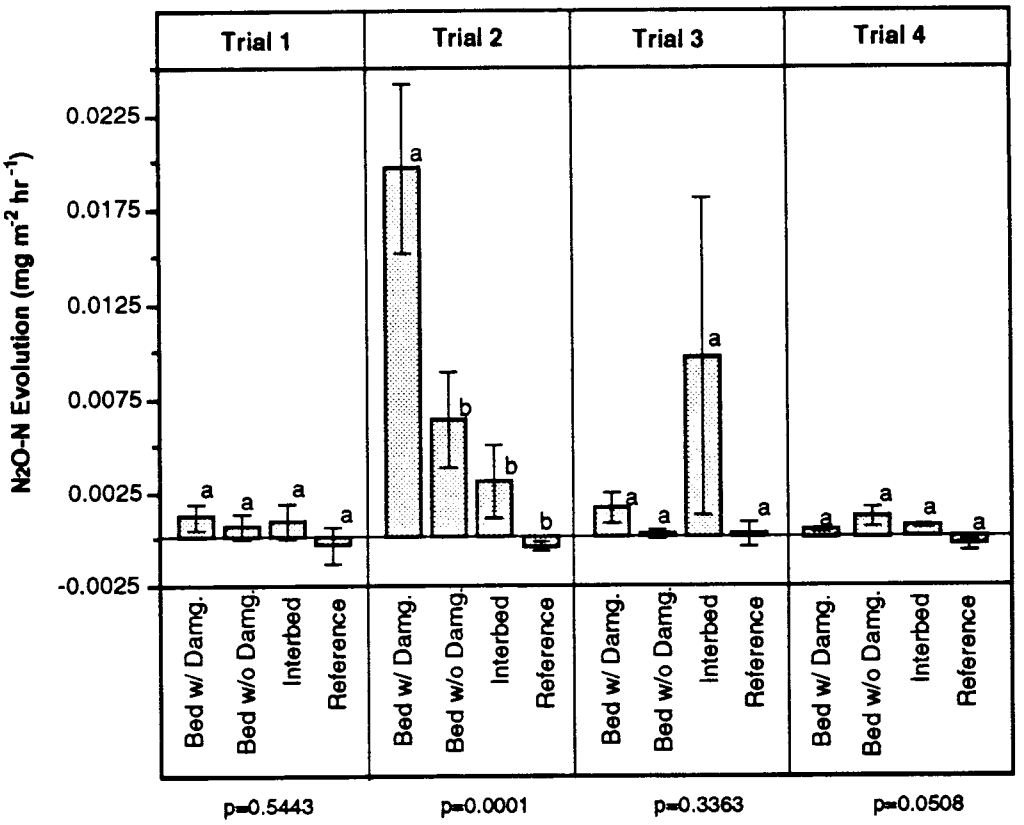



Figure 35. Nitrous-oxide evolution from four microsite types: 1) reference (undisturbed), 2) beds without previous harvest damage, 3) beds with previous harvest damage, and 4) interbed rows. Means separated with F-test-protected Duncan's Multiple Range Test. Means with the same letter are not significantly different. Error bars are ± one standard error of the mean.

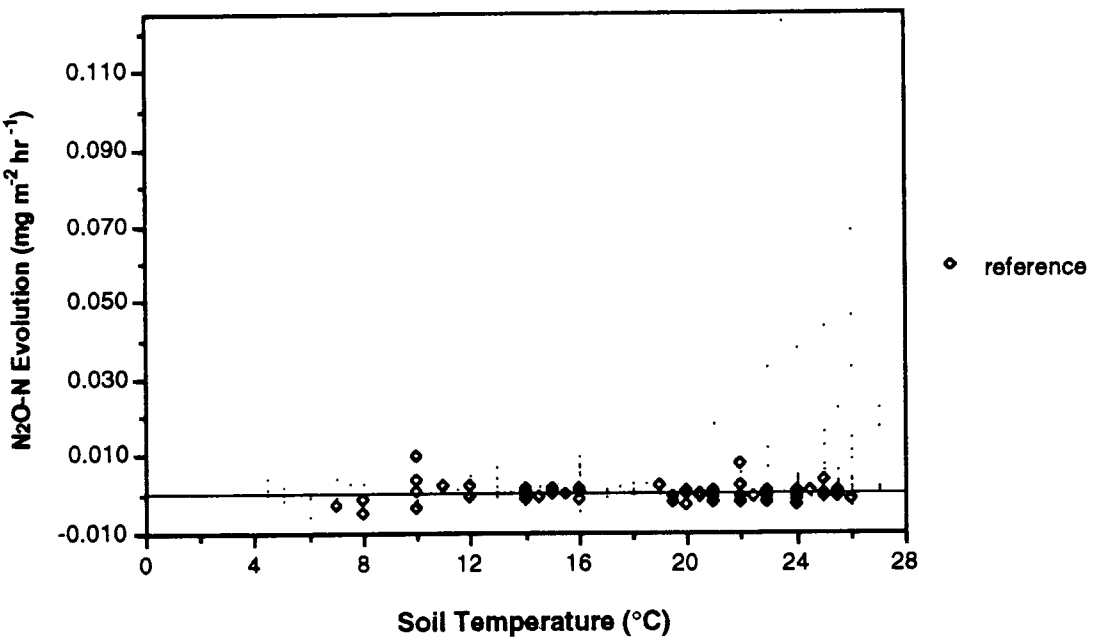



Figure 36. Nitrous oxide evolution vs. temperature for all microsite measurements with reference microsites highlighted.

Figure 37. Nitrous-oxide evolution from individual microsites on the Goldsboro (moderately well drained) soil series. Each bar, from left to right, represents the same microsite in every trial period.

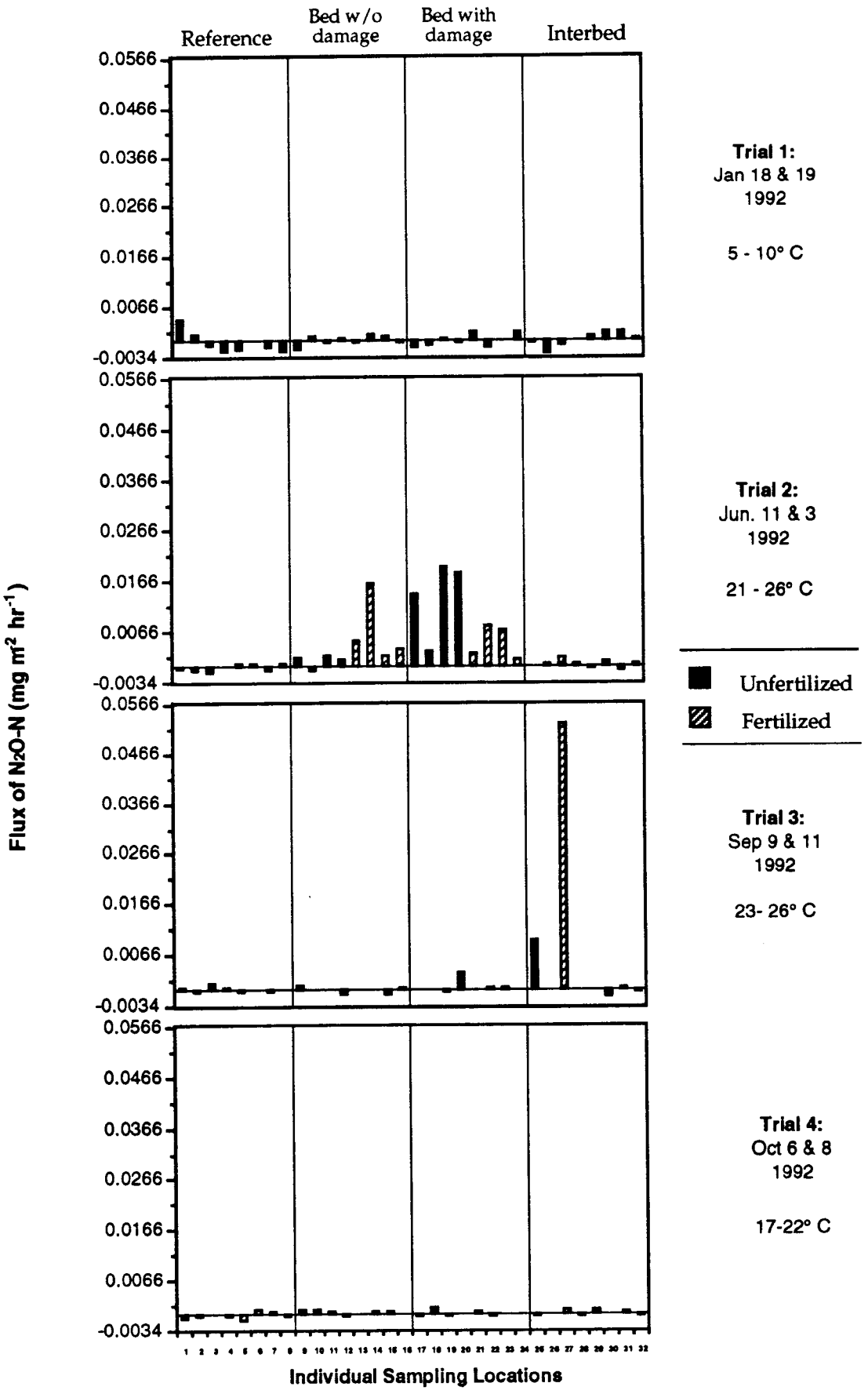
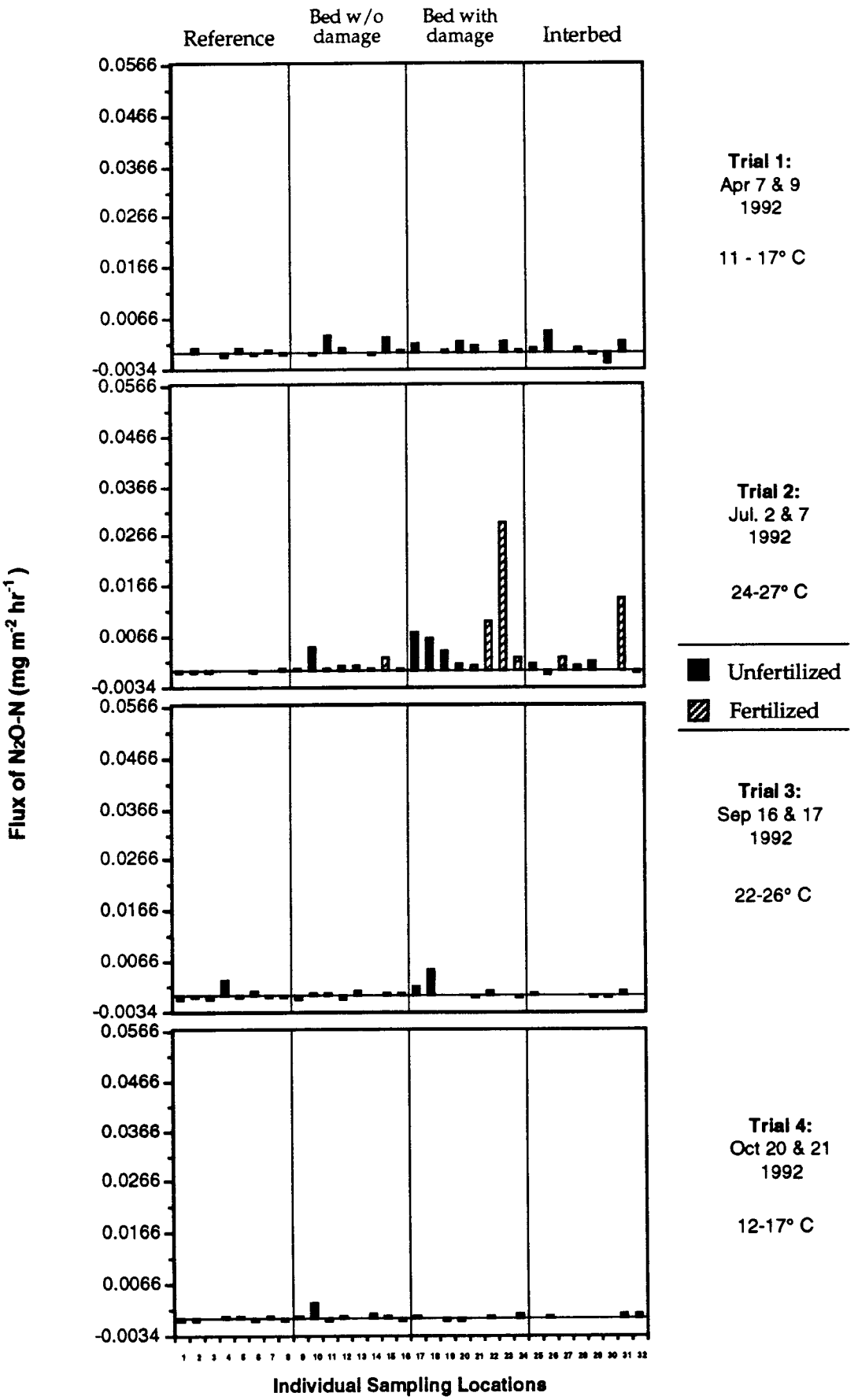

Reference
Bed w/o damage
Bed with damage
Interbed
0.0566
0.0466
0.0366
0.0266
0.0166
0.0066
-0.0034
Flux of N2O-N (mg m-2 hr-1)
Trial 1: Jan 18 & 19 1992
5 - 10° C
Trial 2: Jun. 11 & 3 1992
21 - 26° C
Unfertilized
Fertilized
Trial 3: Sep 9 & 11 1992
23- 26° C
Trial 4: Oct 6 & 8 1992
17-22° C
Individual Sampling Locations

Figure 38. Nitrous-oxide evolution from individual microsites on the Bethera (poorly drained) soil series. Each bar, from left to right, represents the same microsite in every trial period.


Reference
Bed w/o damage
Bed with damage
Interbed
0.0566
0.0466
0.0366
0.0266
0.0166
0.0066
-0.0034
Flux of N2O-N (mg m-2 hr-1)
Trial 1: Apr 7 & 9 1992
11 - 17° C
Trial 2: Jul. 2 & 7 1992
24-27° C
Unfertilized
Fertilized
Trial 3: Sep 16 & 17 1992
22-26° C
Trial 4: Oct 20 & 21 1992
12-17° C
Individual Sampling Locations

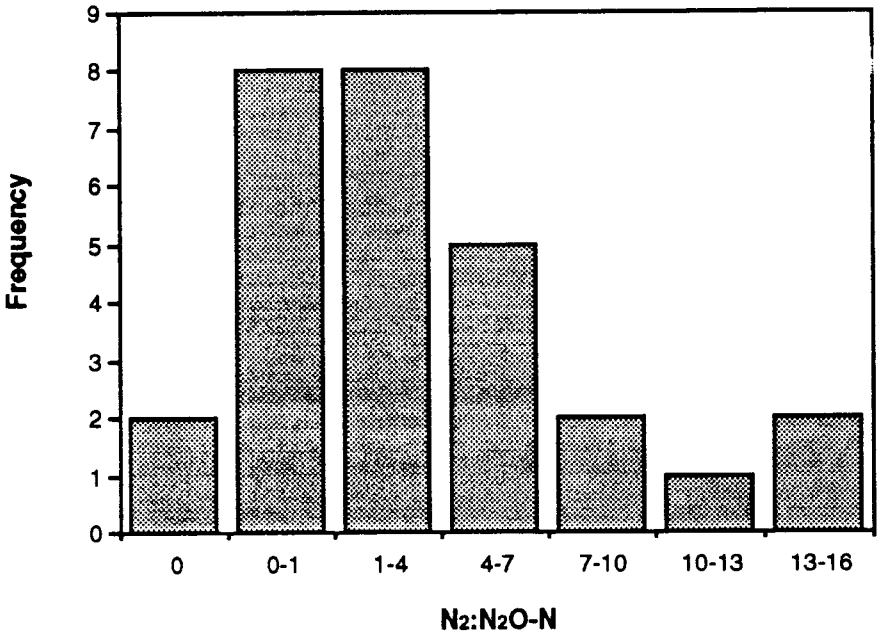



Figure 39. Frequency distribution of $N_2$:$N_2O$-N in $N_2O$-producing soil samples. The mean value of $N_2$:$N_2O$-N in laboratory incubations was 4.1.

Table 16. Multiple linear regression for dependent variable $N_2O$ flux from all microsite types.

R-squared = 0.0560

| | DF | Sum of Squares | Mean Square | F | P > F |
|---|---|---|---|---|---|
| Regression | 2 | 0.0016587 | 0.0008294 | 7.51 | 0.0007 |
| Error | 253 | 0.0279552 | 0.0001105 | | |
| Total | 255 | 0.0296139 | | | |

| Variable | Parameter Estimate | Std. Error | Type II Sum of Sqrs. | F | P > F |
|---|---|---|---|---|---|
| Intercept | -0.0081477 | 0.0035228 | 0.0005911 | 5.35 | 0.0215 |
| $Temp^2$ | 0.0000095 | 0.0000030 | 0.0010875 | 9.84 | 0.0019 |
| Bulk Density | 0.0054444 | 0.0025542 | 0.0005020 | 4.54 | 0.0340 |

Table 17. Multiple linear regression for dependent variable $N_2O$ flux from all microsite types. Regression confined to Trial 2.

R-squared = 0.0910

| | DF | Sum of Squares | Mean Square | F | P > F |
|---|---|---|---|---|---|
| Regression | 1 | 0.0010443 | 0.0010443 | 6.20 | 0.0154 |
| Error | 62 | 0.0104373 | 0.0001683 | | |
| Total | 63 | 0.0114816 | | | |

| Variable | Parameter Estimate | Std. Error | Type II Sum of Sqrs. | F | P > F |
|---|---|---|---|---|---|
| Intercept | 0.0032887 | 0.0022246 | 0.0003679 | 2.19 | 0.1444 |
| Nitrate | 1.6235642 | 0.6518667 | 0.0010443 | 6.20 | 0.0154 |

## DISCUSSION

### Residual Damage Effects

Soil damage persisted into the establishment phase of the subsequent rotation, despite mitigation by bedding and fertilization. Residual harvest damage in planting beds depressed gross soil biological activity and methane evolution, while increasing production of nitrous oxide. Suppression of biological activity coincided with a lower concentration of water-soluble carbon (Figure 9), depleted nitrogen and phosphorus (Figure 10), higher bulk density (Figure 14), and critically low levels of air-filled porosity (Figure 16).

Biogenic gas fluxes were more sensitive to residual damage, in magnitude of response, than traditional measures such as bulk density, total organic carbon, and Kjeldahl nitrogen and phosphorus (Table 18). The reduction in $CO_2$ and $CH_4$ production—despite generally smaller differences in other soil properties—could have resulted from either cumulative stress or a limiting factor(s). In beds with residual damage, reductions in water-soluble organic carbon and air-filled porosity were especially large (Table 18). Depletion of water-soluble organic carbon suggested the possibility of a labile carbon limitation. Displacement of litter and A-horizon material during the salvage operation had no apparent effect on total soil carbon (Figure 9), but probably removed a disproportionate share of labile carbon in fresh detritus. Air-filled pore space in previously damaged beds averaged 10 % or less throughout the study (Figure 27), a level below which gas diffusion through most porous media is virtually zero (Blake and Page, 1948; Wyckoff and Botset, 1936).

Table 18. Percent difference in variables in previously damaged beds compared with previously undamaged beds.

| *Variable* | *% difference* | *Comment* |
|---|---|---|
| *Stable † Soil Properties* | | |
| [organic C] | - 8 | (NS) |
| [Kjeldahl N] | - 19 | |
| [Kjeldahl P] | - 10 | |
| bulk density | + 10 | |
| *Unstable Soil Properties* | | |
| [water-soluble organic C] | - 37 | Measured in Trial 4 only. |
| $[NH_4^+]^*$ | + 22 | (NS) |
| $[NO_3^-]^*$ | + 15 | (NS) |
| denitrifying enzyme activity* | + 9 | (NS) |
| soil moisture* | + 7 | Significant in 2 of 4 trials. |
| air-filled porosity* | - 43 | |
| *Process Rates* | | |
| $CO_2$ evolution* | - 34 | |
| $CH_4$ evolution* | - 54 | (NS) |
| $N_2O$ evolution | + 210 | $N_2O$ detectable in Trial 2 only |

† unlikely to change significantly in one year. * mean over all available trial periods.
NS - difference not significant at a = 0.05.
Unless otherwise noted, differences are significant.

Carbon-dioxide production was by far the most stable process, and the difference between disturbed and undisturbed beds was significant in all trial periods at $\alpha = 0.05$ (Figure 19). Because $CO_2$ originates from a vast diversity of organisms, including a large contribution from plant living plant roots, the cumulative stress hypothesis seems an appropriate explanation. Though no studies were found examining $CO_2$ efflux after application of a mitigation treatment, depressed $CO_2$ evolution (Torbert and Wood, 1992) and cellulose decay (Aust et al., 1988) have been observed elsewhere in compacted and heavily trafficked soils.

Methanogens, which are capable of utilizing only the simplest organic compounds (Conrad, 1989), should have been especially vulnerable to a labile carbon limitation, but other mechanisms are plausible as well. If gas diffusivity was limited in previously damaged beds, the residence time of methane in the soil would have been lengthened, favoring attenuation of methane efflux by biological oxidation. Due to large spatial variance, the average 54 % reduction in methane production associated with residual damage was of marginal statistical significance ($\alpha = 0.1$ in two of four trial periods).

Nitrous oxide evolution is an ephemeral process in this and most other ecosystems. During Trial 2, when $N_2O$ evolution was detected, the rate from planting beds with residual damage was 210 % higher than from previously undamaged beds, and the difference was highly significant. Several other studies (Bakken et al., 1987; Becker et al., 1989; Torbert and Wood, 1992) have demonstrated accelerated N loss in compacted and heavily trafficked soils. One explanation is that redox potential drops below the critical level for $NO_3^-$ respiration because gas diffusion is inhibited.

If nitrate had been available, these soils should have provided ample anaerobic microsites, given their high organic matter contents, frequent rainfall in 1992, and water table within one meter of the surface throughout most of the year. However, $N_2O$ evolution was absent on many sites where methanogenesis was occurring, a process that requires a much lower redox potential. This suggests that denitrification was probably limited by nitrate. Without fertilization, nitrate is extremely scarce in acid forest soils, because of both rapid utilization and nitrifiers intolerance of acidity. While $NO_3^-$ concentrations in previously damaged and undamaged areas did not differ, lower overall biological activity in beds with residual damage may have lead to lower

assimilatory demand for nitrate by plants, microorganisms, and fungi—leaving a greater share for dissimilatory nitrate reduction by anaerobic bacteria.

$CO_2$ evolution should be widely applicable as a measure of cumulative stress in soils impacted by heavy machine traffic. Methane efflux is useful only in soils with strong reducing conditions. Given the low diversity of methanogens and strict prerequisites for methanogenesis, it is a much more narrowly focused indicator. Nitrous-oxide production is ephemeral and, thus, requires intense sampling for accurate characterization. It can constitute none or virtually all of gaseous N loss, depending on several temporally and spatially variable factors (Table 4). However, when $N_2O$ evolution occurs, its cause is limited to a few possibilities—for example, increased nitrate availability. $N_2O$ evolution is useful in evaluating N-enriched soils, especially if acetylene is used *in-situ*, or in the laboratory analogs, to approximate total denitrification.

## Bedding Effects

In this case, bedding was used as a damage mitigation treatment, but it is also a common silvicultural practice in the Coastal-Plain pinery. This discussion focuses upon both the creation of distinctive microsite conditions within bedded areas, as well as the overall effect of bedding on elemental efflux from a site. Bedded areas without residual damage are compared with undisturbed reference sites to isolate the effect of bedding alone.

### Microsite Conditions / Soil Quality

Rates of $CO_2$ evolution from planting beds and reference microsites were similar, however, the formation of beds was responsible for large ephemeral increases in nitrous-oxide and methane production (Table 19). Concentrations of carbon, nitrogen, and phosphorus were nearly equivalent in beds and undisturbed reference soils, reflecting the redistribution of A-horizon material into beds. Labile carbon and nitrogen concentrations in beds ranked consistently higher

than in reference microsites, but this trend was significant only in the case of nitrate during Trial 2. Higher bulk density near the surface in beds may have resulted from deposition of Bt-horizon and lower-A-horizon soil near the surface.

Table 19. Percent difference in variables in bed microsites compared with reference microsites.

| *Variable* | *% difference* | *Comment* |
|---|---|---|
| *Stable Soil Properties*† | | |
| [organic C] | + 1 | (NS) |
| [Kjeldahl N] | + 7 | (NS) |
| [Kjeldahl P] | + 8 | (NS) |
| bulk density | + 25 | |
| *Unstable Soil Properties* | | |
| [water-soluble organic C] | + 23 | (NS) |
| $[NH_4^+]$* | + 42 | (NS) |
| $[NO_3^-]$* | + 87 | Significant in Trial 2 only. |
| denitrifying enzyme activity* | + 26 | (NS) |
| soil moisture* | - 13 | |
| air-filled porosity* | -18 | Significant in 2 of 4 trials. |
| *Process Rates* | | |
| $CO_2$ evolution* | - 5 | Significant in 1 of 4 trials. |
| $CH_4$ evolution* | + 674 | Significant in 3 of 4 trials††. |
| $N_2O$ evolution | + ∞ | No activity in reference areas, but effect NS. $N_2O$ detected in Trial 2 only. |

† unlikely to change significantly in one year.
* mean over all available trial periods. †† $p \leq 0.07$
NS - difference not significant at a = 0.05.
Unless otherwise noted, differences are significant.

Despite lack of significance in direct measurements, higher concentrations of labile carbon and nitrogen could explain the stimulation of methanogenesis and nitrous-oxide production in beds. One objective of bedding is to control competition for planted seedlings. Competition control is achieved by

incorporating herbaceous vegetation into the soil through tillage, leaving the surface temporarily devoid of plants. Three factors would have favored higher carbon and nutrient availability in beds: 1) the addition of fresh organic material, 2) the "priming" effect of tillage, and 3) decreased competition for nutrients by plants. Though the responsible factors were not clearly identified, biogenic gas fluxes demonstrated functional differences among soils that otherwise appeared similar.

**Elemental Efflux from Bedded Sites**

Changes in gas flux rates were substantial at the field level. Severe suppression of biological activity in interbed rows (Figure 19), combined with the limited surface area for gas exchange with the atmosphere in beds, lead to reduced rates of $CO_2$ efflux from bedded plots (Figure 17). Rather than accelerate carbon loss, bedding may have actually lead to conservation of carbon. This finding is applicable only to the method of bedding used in this study, in which a barren, infertile Bt horizon is exposed in the interbed rows. Without an estimate of net primary production (NPP), the net change in carbon inventory is uncertain. NPP was probably reduced in the first year due to suppression of vegetation.

Because spatial variability of methane evolution was so large, the experimental design was marginally adequate in detecting potential treatment effects. $CH_4$ evolution from bedded plots was not significantly greater than that of reference plots (Figure 27). However, scattered high peaks in methane evolution occurred exclusively in bedded areas (Figures 31 and 32), and evolution from bed microsites was greater than that from reference sites in 3 of 4 trial periods ($p \leq 0.07$). While carbon loss through methanogenesis was too small to affect the total carbon inventory, it could have rendered the most labile carbon fractions less available to other organisms.

Bedding stimulated $N_2O$ production. While $N_2O$ production was ephemeral and spatially variable, it was limited exclusively to bedded areas, resulting in significantly higher efflux from bedded plots during Trial 2 (Figure 33). Accelerated $N_2O$ production from the planting beds during Trial 2 (Figure 35) coincided with higher nitrate availability ($p=0.08$).

## Fertilization Effects

Carbon dioxide was the only gas flux that responded significantly to fertilization. Fertilization stimulated $CO_2$ evolution in trials 2 and 3 following fertilization (Figure 18) and appeared to compensate partially for depressed $CO_2$ evolution in previously damaged beds (Figure 20). Fertilizer may have stimulated carbon mineralization and/or root respiration (by increasing plant vigor) but these contributions were not partitioned. A similar response in seedling growth was observed by Hatchell (1981) in pine flats of the Francis Marion Nation Forest, where a combination of bedding and fertilization mitigated harvest damage more effectively than bedding alone. The positive response to fertilizer is inconsistent with a strict labile carbon limitation on soil respiration.

There are several possible explanations for the apparent lack of nitrous-oxide production following fertilization.

1) It occurred but was not detected. Fertilization frequently results in short-term, pulsed release of N gases. In this study, sampling was delayed about a month after fertilization.
2) There was little or no "nitrification link" for conversion of the applied ammonia to nitrate.
3) Nitrogen evolved primarily as $N_2$ rather than $N_2O$ and, thus, went undetected. In fact, laboratory incubations suggested that only about 20 % of N evolved in the form of $N_2O$. Because of the uncertainties

associated with estimates of $N_2O$ efflux, interpretations concerning N-gas loss are limited.

## Regression Analyses

### Determination of Values from Individual Microsites

Detailed characterization of individual microsites allowed for investigation of relationships among gas flux rates and other factors through regression analyses. To obtain meaningful regression equations, it was essential that measurements represent, as accurately as possible, the individual 0.5- x 1- m microsites.

Some parameters were measured very precisely at the microsite level, for example, gas fluxes, where the entire surface area of the microsite was sampled, and soil temperature, in which spatial variance was minimal. To ensure precise measurement of properties determined from soil samples, six samples were collected with a push tube from each microsite and composited. Dried soil samples were crushed, mixed, and sieved, ensuring a uniform sample and, thus, a fairly high degree of precision in determination of total organic matter and Kjeldahl nitrogen and phosphorus.

Obtaining a representative portion from composited, field-moist samples was problematic and lead to relatively large measurement error. Field-moist samples were used in determination of denitrifying enzyme activity and concentrations of nitrate, ammonia, and water-soluble organic carbon. Though these samples were mixed as thoroughly as possible, they still contained discrete inclusions of organic matter, clay, and other materials that could have disproportionately influenced values determined from the small portions withdrawn for analysis. Error associated with poor mixing may have been critical in regression analysis, where each value represented a specific microsite. The additional variation was less critical in comparison of means averaged over many measurements. Hence, poor relationships with variables determined from

field-moist soil samples could have been due to either measurement error or legitimately weak relationships.

**Factors Related to Gas Flux Rates**

In undisturbed soils, soil respiration was a robust function of temperature across soil series, drainage classes, and microtopographic variation. In an exponential regression that included all measurements from reference microsites in the study, temperature accounted for fully 85 % of variation in soil respiration. Factors other than temperature became limiting with increasing severity of disturbance (Figure 24). In particular, Kjeldahl nitrogen was correlated with $CO_2$ evolution in multiple linear regression across all microsite types. Rather than a nitrogen limitation *per se,* the importance of Kjeldahl nitrogen could reflect a limitation associated with organic matter quality, inclusions of Bt-horizon soil near the surface, or other secondary factors. Forty percent of variation in soil respiration across microsite types was not accounted for in regression analysis.

Regression analysis largely failed to identify factors responsible for rates of $N_2O$ and $CH_4$ evolution. Several explanations are possible. 1) Dynamic factors such as water-table fluctuations and antecedent rainfall not characterized in this study were the primary determinants. 2) Even if one of the parameters measured such as labile nitrogen or carbon was responsible, the size of the labile pool itself may have been less important than the rate of supply to it by processes such as decomposition and nitrification. 3) Values for some measures (such as those determined from field-moist soil samples) did not accurately represent specific microsites.

## Accuracy of Gas-Flux Estimates

The chamber design and sampling protocol used in this study overcame shortcomings of some other designs and were especially well suited to forest soils strewn with coarse debris. Nevertheless, some biases and uncertainties

such as root disturbance, modification of microclimate, and underestimation of gaseous N losses remained.

**Shortcomings of the Static Chamber Method**

Root respiration, which commonly comprises 30 to 50 % of total soil respiration, may have been partially suppressed. While clipping herbaceous vegetation in the chamber mitigated against unwanted photosynthetic consumption of carbon-dioxide, it could have also reduced root respiration by severing the supply of photosynthate to roots (Davidson and Milthorpe, 1966; Frossard, 1976). These studies suggested that root respiration could decline by approximately 30 to 50 % twelve hours after clipping. In the present study, foliage was typically clipped the afternoon before sampling. Assuming roots contributed 40 % of total respiration, clipping could have reduced $CO_2$ efflux by 20 % . This is probably an overestimate, as there would have been contributions from surrounding trees, shrubs, and herbaceous plants, whose roots extended into the sampled area.

By inserting the chamber base 3 to 8 cm into the soil and severing some roots, contributions from surrounding plant roots may have been reduced as well. Root disturbance can be minimized by 1) minimizing the depth of insertion, 2) using a foam or similar collar that is not inserted into the soil (e.g., Matthias et al., 1980), or 3) leaving chamber bases in place permanently, so roots can recover after the initial disturbance. Methods that minimize root disturbance can also compromise the integrity of the air-tight seal. A proper balance must be struck and the potential biases understood.

Another shortcoming of static chambers is that they eliminate or alter atmospheric pressure fluctuations that normally occur at the soil surface due to turbulence or air movement (Mosier, 1990), perhaps leading to underestimation of actual fluxes.

In estimating N-gas losses, only the $N_2O$ component was reliably determined. The average $N_2$:$N_2O$-N of active samples in laboratory incubations was 4.1 (Figure 39), suggesting that measurement of $N_2O$ alone greatly underestimated gaseous N loss. This laboratory-determined ratio is only an approximation of the $N_2$:$N_2O$-N ratio in the field.

**Advantages of the Static Chamber Method**

The advantages of the method chosen exceeded its shortcomings. The large sampling area of the chamber (0.5 $m^2$) minimized potential edge effects and "averaged out" some of the extreme small-scale spatial variability in forest soils, often attributable to "hot spots" of activity at organic-matter accumulations. The water seal into which chamber walls were inserted was designed with sufficient depth and tolerance that precise leveling of the chamber was unnecessary, facilitating rapid installation of the bases. The surface area of the water within the chamber and its effect on internal chamber volume were negligible. Soil warming was also avoided by solar shielding.

Alternative methods were either inappropriate or infeasible. Micrometeorological methods involve characterizing concentration gradients above the ground and calculating fluxes based on estimated "transfer coefficients" and wind speed measurements (de Jong et al., 1979; Mosier, 1990). Such methods integrate over large areas and are, thus, useless in evaluation of small plots or disturbed microsites. Dynamic (flow-through) chambers are used almost exclusively to measure $CO_2$ fluxes, because of the low cost of infrared detectors and $CO_2$ absorption media. The gas-profile method involves examining gas concentrations at various depths in the soil, and calculating fluxes to the surface based upon diffusivity estimates (deJong et al., 1979). This approach was considered inappropriate because it has seldom, if ever, been

applied to methane and nitrous-oxide fluxes, and partitioning of fluxes in subsurface layers was unnecessary.

## CONCLUSIONS

### Regarding Residual Damage Effects in Planting Beds

1. Residual soil damage from wet weather logging persisted into the establishment phase of the subsequent rotation, despite mitigation by bedding and fertilization.
2. Soil biological activity (as evidenced by $CO_2$ evolution) was consistently sensitive to residual damage, declining an average of 34 % in response to multiple stresses. $CO_2$ evolution was, thus, a strong composite indicator of soil condition.
3. Though aeration was reduced to a critically low level, methane evolution was not accelerated. Instead, it was unexpectedly reduced an average of 54 %. The marginal significance of this effect ($\alpha = 0.10$ in two of four trial periods) and high spatial variance point to the need for more intensive sampling to characterize this process. Measurement of $CH_4$ fluxes is promising for damage assessment in highly reduced soils.
4. Nitrous-oxide evolution was elevated 210% in beds with residual damage during the one trial period in which appreciable $N_2O$ evolution occurred. This effect was consistent with earlier studies showing increased N loss due to trafficking and compaction. Nevertheless, $N_2O$ evolution was an unreliable indicator of soil condition in forest soils because of its ephemeral nature and lack of methodology to measure the $N_2$ fraction of gaseous N loss.
5. The soil characteristics most strongly influenced by residual damage were water-soluble organic carbon, reduced 37 %, and air-filled pore space, reduced an average of 43 % to a level at which gas diffusion

through most porous media is virtually zero. Calculation of air-filled pore space is far more useful in soil damage assessment than measurement of moisture or bulk density alone.

**Regarding Bedding Effects**

1. Carbon efflux from bedded areas was significantly lower than undisturbed reference sites in 2 of 4 trial periods, due to low rates of biological activity in interbed rows and limited surface area for gas exchange in beds.
2. Methane evolution was greater in planting beds than undisturbed reference microsites in 3 of 4 trial periods, though the responsible factors were not clearly identified. In field-level comparisons—averaging rates from beds and interbed rows—methane evolution from bedded plots was not significantly greater than that of reference plots.
3. Nitrous-oxide production was accelerated by bedding in the one trial period when appreciable $N_2O$ evolution was detected, probably due to increased availability of labile nitrogen.
4. Differences in methane and nitrous-oxide production among beds and reference microsites revealed important functional differences among soils that otherwise appeared similar.

**Regarding Fertilization Effects**

1. Fertilization increased $CO_2$ evolution in the two trial periods following fertilizer application and compensated partially for the deleterious effects of residual damage in beds.
2. Rates of methane and nitrous-oxide production were not measurably affected by fertilization.

## LITERATURE CITED


Anderson, I.C. and J.S. Levine. 1986. Relative rates of nitric oxide and nitrous oxide production by nitrifiers, denitrifiers, and nitrate respirers. Appl. and Envr. Microbiol, 51(5):938-945.

Aust, W.M., S.F. Mader, and R. Lea. 1988. Abiotic changes of a cypress-tupelo swamp following helicopter and rubber tired skidder timber harvest. 5th biennial southern silvicultural conference.

Aust, W.M., 1990. A case study in soil rutting. In J.E. Johnson, ed. Timber Harvesting and Forest Soil Rutting. Virginia Polytechnic Institute, School of Forestry and Wildlife Resources. Blacksburg, Virginia, USA.

Aust, W.M., T.W. Reisinger, J.A. Burger, and B.J. Stokes. 1991. Evaluation of Soil and Site Disturbance for Selected Harvesting Systems Operating on the Francis Marion National Forest: Final Report to U.S.D.A. Forest Service. Dept. of Forestry, Virginia Polytechnic Institute, Blacksburg, Virginia.

Aust, W.M., T.W. Reisinger, J.A. Burger, and B.J. Stokes. 1993. Soil physical and hydrological changes associated with logging a wet pine flat with wide-tired skidders. S. J. Appl. For., 17(1): 22-25.

Baker-Blocker, A., T.M. Donahue, and Khalil, H.M. 1977. Methane flux from wetland areas. Tellus, 29:245-250.

Bakken, L.R., T. Børrenson, and A. Njøs. 1987. Effect of soil compaction by tractor traffic on soil structure, denitrification, and yeild of wheat (*Triticum aestivum* L.). J. Soil Sci., 38: 541-552.

Becker, K.W., E. Janssen, and B. Meyer. 1989. The $^{15}N$-balance method for calculating denitrification losses in arable fields and its verification by direct measurement of gaseous $^{15}N$ losses. In Proceedings, International workshop on denitrification: soil, rhizosphere, and aquifer. Giessen, FRG, March 17-19, 1989. Mitteilungen-der-Deutschen-Bodenkundlichen-Gesellschaft, 60:59-64.

Behera, N., S.K. Joshi, and D.P. Pati. 1990. Root contribution to total soil metabolism in a tropical forerst soil from Orissa, India. Forest Ecology and Management, 36:125-134.

Berg, P.J. 1975. Developments in the establishment of second rotation radiata pine at Riverhead Forest. New Zealand J. For., 20:272-282.

Birch, H.F. 1958. The effect of soil drying on humus decomposition and nitrogen availability. Plant and Soil, 10:9-31.

Birch, H.F. 1960. Nitrification in soils after different periods of dryness. Plant and Soil, 12:81-96.

Blake, D.R. and F.S. Rowland. 1988. Continuing worldwide increase in tropospheric methane, 1978 to 1987. Science 239:1129-1131.

Blake, G.R. and J.B. Page. 1948. Direct measurement of gaseous diffusion is soils. Soil Sci. Soc. Amer. Proc., 13:37-42.

Boois, H.M.N. 1974. Measurement of seasonal variation in the oxygen uptake of various litter layers of an oak forest. Plant and Soil, 40:545-555.

Brietenbech, G.A., A.M. Blackmer, and J.M. Bremner. 1980. Effects of different nitrogen fertilizers on emission of nitrous oxide from soil. Geophys. Res. Lett., 7:85-88.

Bremner, J.M. and A.M. Blackmer. 1978. Nitrous oxide emission from soils during nitrification of fertilizer nitrogen. Science, 199:295-296.

Bremner, J.M., G.A. Breitenbeck, and A.M. Blackmer. 1981. Effect of anhydrous ammonia fertilization on emission of nitrous oxide from soils. J. Environ. Qual., 10:77-80.

Burger, J.A. 1979. The effects of harvest and site preparation on the nutrient budget of an intensively managed southern pine forest. Ph.D. Thesis., University of Florida. University Microfilms, Ann Arbor, MI.

Burger, J.A., J.V. Perumpral, J.L. Torbert, R.E. Kreh, and S. Minaei. 1983. The effect of track and rubber-tired vehicles on soil compaction. Paper presented at meeting of American Society of Agricultural Engineers, December 13-16, 1983, Paper no. 83-1621. American Society of Agricultural Engineers, St. Joseph, Michigan, USA.

Burger, J.A. 1990. Ecological effects of soil rutting. In J.E. Johnson, ed. Timber Harvesting and Forest Soil Rutting. Virginia Polytechnic Institute, School of Forestry and Wildlife Resources. Blacksburg, Virginia, USA.

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

Campbell, R.G., J.R. Willis, and J.T. Marj. 1973. Soil disturbance by logging with rubber-tired skidders. Journal of Soil and Water Conservation, 28:218-220.

Chancellor, 1977. Compaction of Soil by Agricultural Equipment. Division of Agricultural Sciences, University of California, Berkley, California.

Cheatle, R.J. 1989. Tree growth on a compacted Oxisol. Soil andTillage Research, 19(2-3):331-344.

Cicerone, R.J. and R.S. Oremland. 1988. Biogeochemical aspects of stmospheric methane. Global Biochemical Cycles, 2:299-327.

Cohen, Y. and L. Gordon. 1979. Nitrous oxide production in the ocean. J. Geophys Res., 84:347-353.

Colbourn, P. and I.W. Harper. 1987. Denitrification in drained and undrained arable clay soil. J. Soil Sci., 38:531-539.

Coleman, D.C. 1973. Compartmental analysis of "total soil respiration": an exploratory study. Oikos, 24:361-366.

Conrad, R. 1989. Control of methane production in terrestrial ecosystems. In M.O. Andrae and D.S. Schimel, eds., Exchange of Trace Gases between Terrestrial Ecosystems and the Atmosphere. John Wiley and Sons Ltd., New York, NY.

Crutzen, P.J. 1974. Estimates of possible variations in total ozone due to natural causes and human activities. Ambio, 3:201-210.

Daddow, R.L. and G.E. Warrington. 1983. Growth-Limiting Bulk Densities as Influenced by Soil Texture. USDA Forest Service, Watershed Systems Development Group Report. WSDG-TN-00005. Fort Collins, Co.

Davidson, J.L. and F.L. Milthorpe. 1966. The effect of defoliation on the carbon balance in *Dactylis glomerata*. Annals of Botany, 30:184-198.

deJong, E. R.E. Redman, and E.A. Ripley. 1979. A comparison of methods to measure soil respiration. Soil Science, 127(5):300-306.

Dickerson, B.P. 1976. Soil compaction after tree-length skidding in northern Mississippi. Soil Sci. Soc. Am. J., 40:965-966.

Donnelly, J.R. and J.B. Shane. 1986. Forest ecosystem responses to artificially induced compaction. I. Soil physical properties and tree diameter growth. Can. J. For Res., 16:750-754.

Drury, C.F., D.J. McKenney, and W.I. Findlay. 1992. Nitric oxide and nitrous oxide production from soil: water and oxygen effects. Soil Sci. Soc. Amer. J., 56:766-770.

Dryrness, C.T. 1965. Soil surface condition following tractor and high-lead logging in the Oregon Cascades. Journal of Forestry, 63:272-275.

Edwards, N.T. and P. Sollins. 1973. Continuous measurement of carbon dioxide evolution from partitioned forest floor components. Ecology, 54:406-412.

Ehalt, D.H. 1988. How has the atomospheric concentration of $CH_4$ changed? In F.S. Rowland and I.S.A. Isaksen, eds., The Changing Atmosphere, John Wiley and Sons, NY.

Elkins, J.S., M. Wofsy, M. McElroy, C. Kolb, and W. Kaplan. 1978. Aquatic sources and sinks for nitrous oxide. Nature, 275:602-606.

Ewel, K.C., W.P. Cropper Jr., and H.L. Gholz. 1987. Soil $CO_2$ evolution in Florida slash pine plantations. II. Importance of root respiration. Can. J. For. Res., 17:330-333.

Firestone, M.K. and E.A. Davidson. 1989. Microbiological basis of NO and $N_2O$ production and consumption in soil. In M.O. Andreae and D.S. Schimel, Exchange of trace gases between terrestrial ecosystems and the Atmosphere. John-Wiley and Sons, NY.

Foil, R. and C.W. Ralston. 1967. The establishment and growth of loblolly pine seedlings on compacted soils. Soil Sci. Soc. of Amer. J., 31:565-568.

Froelich, H.A. 1976. The influence of different thinning systems on damage to soil and trees. pp. 333-344. In Proc. XVI IUFRO World Congress, Division IV, Norway.

Froelich, H.A., D.W.R. Miles, and R.W. Robbins. 1985. Soil bulk density recovery on compacted skid trails in central Idaho. Soil Sci. Soc. of Amer. J., 49:1015-1017.

Frossard, J.S. 1976. *Relations entre l'éclairement des feuilles et l'absorption d'oxygène par les racines chez le tournesol (Helianthis annuus L.) Annales Agronomique,* 27:435-445.

Gambrell, R.P. and W.H. Patrick Jr. 1978. Chemical and microbiological properties of anaerobic soils and sediments. In D.D. Hook an R.M.M. Crawford, eds., Plant Life in Anaerobic Environments,. Ann Arbor Scientific Publications Incorporated, Ann Arbor, Mich, pp. 375-423.

Gent, J.A. Jr., R. Ballard, and A.E. Hassan. 1983. The impact of harvesting and site preparaton on the physical properties of lower Coastal Plain forest soils. Soil. Sci. Soc. Amer. J., 47:595-598.

Goncharov, V.M. and Ye.V. Shein. 1991. Soil hydrological regime in the technological rut zone. Moscow University Soil Sci. Bull., 46(2) 35-38.

Gooding, T.H. and T.M McCalla. 1945. Loss of $CO_2$ and $NH_3$ from crop residues during decomposition. Soil. Sci. Soc. Amer. Proc., 10:185-190.

Goreau, T.J., W.A. Kaplan, S.C. Wofsy, M.B. McElroy, F.W. Valois, and S.W. Watson. 1980. Production of $NO_2^-$ and $N_2O$ by nitrifying bacteria at reduced concentrations of oxygen. Appl. and Envr. Microbiol., 40(3):526-532.

Grable, A.R. and E.G. Seimer. 1968. Effects of bulk density, aggregate size, and soil water suction on oxygen diffusion, redox potentials, and elongation of corn roots. Soil Sci. Soc. Amer. Proc., 32:180-186.

Hatchell, G.E. 1981. Site preparation and fertilizer increase pine growth on soils compacted in logging. Southern Journal of Applied Forestry, 5:79-83.

Hatchell, G.E. and C.W. Ralston. 1971. Natural recovery of surface soils disturbed in logging. USDA Forest Service, Tree Planters Notes, 22(2):5-9.

Hatchell, G.E., G.W. Ralston, and R.R. Foil. 1970. Soil disturbances in logging. Journal of Forestry, 772-775.

Howes, B.L., J.W.H. Dacey, and G.M. King. 1984. Carbon flow through oxygen and sulfate reduction pathways in salt marsh sediments. Limnol. Oceanogr., 29(5):10370-1051.

Johansson, C. and I.E. Gallaby. 1984. Production of nitric oxide in loam under aerobic and anaerobic conditions. Appl. and Evnr. Microbiol., 47:1284-1289.

Karr, B.L., J.D. Hodges, and T.E. Neebeker. 1987. The effect of thinning methods on soil physical properties in North-Central Mississippi. S. J. Appl. For., 11:110-112.

Keller, M., T.J. Goreau, S.C. Wofsy, W.A. Kaplan, and M.B. McElroy. 1983. Production of nitrous oxide and consumption of methane by forest soils. Geophys. Res. Lett., 10:1156-1159.

Keller, M.K., M.E. Mitre, and R.E. Stallard. 1990. Consumption of atmospheric methane in soils of central Panama: effects of agricultural development. Global Biogeochemical Cycles, 4(1):21-27.

King, G.M. 1990. Regulation by light of methane emissions from a wetland. Nature, 345:513-515.

King, G.M, P. Roslev, and H. Skovgaard. 1990. Distribution and rate of methane oxidation in sediments of the Florida everglades. Appl. Environ. Microbiol., 56:2902-2911.

Kira, T. 1978. Carbon cycling. In T. Kira, Y. Ono, and T. Hosokawa, eds., Biological Production in a Warm Temperate Evergreen Oak Forest of Japan. JIBPY Synthesis, Univ. of Tokyo Press, Tokyo, 18:272-276.

Lashof, D.A. and D.R. Ahuja. 1990. Relative contributions of greenhouse gas emissions to global warming. Science, 344:529-531.

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. Soc. Amer. J., 48:1267-1272.

Lieth, H. and R. Ouellette. 1962. Studies on the vegetation of the Gaspé Peninsula. II. The soil respiration of some plant communities. Canadian Journal of Botany, 40:127-140.

Lundegardh, A. 1927, Carbon dioxide evolution from soil and crop growth. Soil Science, 23:417-453.

Mace, A.C. 1971. Recovery of forest soils from compaction by rubber-tired skidders. Minn. For. Res. Notes, no. 226. University of Minnesota, St. Paul.

MacRae, I.C., R.R. Ancajas, and S. Salandanan. 1967. The fate of nitrate nitrogen in some tropical following submergence. Soil Sci., 105:327-334.

Martin, C.W. 1988. Soil disturbance by logging in New England—review and management recommendations. N. J. of Appl. For., 5:30-34.

Matthias, A.D., A.M. Blackmer, and J.M. Bremner. 1980. A simple chamber technique for field measurement of emissions of nitrous oxide from soil. J. Environ. Qual., 9:251-256.

McElroy, M.B. and J.C. McConnell. 1971. Nitrous oxide: A natural source of stratospheric NO. The Journal of the Atmospheric Sciences, 28:1095-1098.

McKee. W.H. Jr. 1990. Restoration of rutted sites. In J.E. Johnson, ed. Timber Harvesting and Forest Soil Rutting. Virginia Polytechnic Institute, School of Forestry and Wildlife Resources. Blacksburg, Virginia, USA.

McKee, W.H. Jr., J.A. Burger, S. Patterson, M. Aust, and L.A. Morris. 1991. Mitigation of forest sites damaged by wet-weather logging. Proposal. Dept. of Forestry, Virginia Polytechnic Institute. Blacksburg, Virginia.

McKee, W.H. Jr. and R.D. Haselton. 1989. Impact of soil compaction on the long term productivity of Peidmont and Atlantic Coastal Plain forest soils. Unpublished Report, USDA Forest Service, SE Experiment Station, Charleston, SC.

Miller, J.H. and D.L. Sirois. 1986. Soil disturbance by skyline yarding vs. skidding in a loamy hill forest. Soil Sci. Soc. Amer. J., 50:1579-1583.

Mitsch, W.J. and J.G. Gosselink. 1986. Wetlands. Van Nostrand Reinhold, New York, NY.

Moehring, D.M and I.W. Rawls. 1970. Detrimental effects of wet weather logging. Journal of Forestry, 68:166-167.

Morris, L.A. 1981. Redistribution and mobilizaiton of nutrients as a result of site preparation of a pine flatwoods forest. Ph.D. Thesis, University of Florida. University Microfilms. Ann Arbor, MI.

Morris, L.A. and W.L. Pritchett. 1982. Effects of site preparation on Pinus Elliotti-P. Palustris flatwoods forest soils properties. In R. Ballard and S.P. Gessel, ed.s. IUFRO. Symposium on Forest Site and Continuous Productivity. USDA Forest Service, General Technical Report PNW-163.

Morris, L.A., W.L. Pritchett and B.F. Swindell. 1983. Displacement of nutrients into windrows diring site preparation of a flatwood forest. Soil Sci. Soc. Am. J., 47:591-594.

Mosier, A.D. 1990. Gas flux measurement techniques with special reference to techniques suitable for measurements over large ecologically uniform areas. In A.F. Bouwman, ed., Soils and the Greenhouse Effect. John Wiley and Sons, NY.

Mosier, A., D. Schimel, D. Valentine, K. Bronson, and W. Parton. 1991. Methane and nitrous oxide fluxes in native, fertilized and cultivated grasslands. Nature, 350:330-332.

Nakane, K. and T. Kira. 1978. Dynamics of soil organic matter in a beech/fir forest on Mt. Odaigehara and other climax forest. Proc. Ann. Meet. Ecol. Soc. Japan, 25:25M.

Nakane, K. 1980. Comparative studies of cycling of soil organic carbon in three primeval moist forests. Jpn. J. Ecol., 30:155-172.

Nakane, K., M. Yamamoto and H. Tsubota. 1983. Estimation of root respiration rate in a mature forest ecosystem. Jpn. J. Ecol., 33:397-408.

Neilson, J.W. and I.L. Pepper. 1990. Soil respiration as an index of soil aeration. Soil Sci. Soc. Amer. J., 54:428-432.

Odum, H.T. and C.F. Jordan. 1970. Metabolism and evapotranspiration of the lower forest in a giant plastic cylinder. In H.T. Odum, ed., A Tropical Rain Forest: A Study of Irradiation and Ecology at El Verde, Puerto Rico. U.S. Atomic Energy Commission, Washington, DC, pp. 1-189.

Oremland, R.S. 1988. Biogeochemistry of methanotrophic bacteria. In A.J.B. Zehnder, ed., Biology of Anaerobic Microorganisms, pp. 641-705. John Wiley and Sons, New York, NY.

Patrick, W.H. and R. Wyatt. 1964. Soil nitrogen loss as a result of altenate submergence and drying. Soil Sci. Soc. Am. Proc., 28:647-653.

Paul, E.A. and F.E. Clark. 1989. Soil Microbiology and Biochemistry. Academic Press Inc., San Diego, CA.

Payne, W.J. 1985. Diversity of denitrifiers and their enzymes. In H.L. Golterman, ed., Denitrification in the Nitrogen Cycle. Plenum Press, NY.

Persson, T., E. Bååth, M. Clarholm, H. Lundkvist, B.E. Söderström, and B. Sohlenius. 1980. Trophic structure, biomass dynamics and carbon metabolism of soil organisms in a scots pine forest. In Structure and Function of Northern Coniferous Forests - An Ecosystem Study, Ecological Bulletin (Stockholm) 32: 419-459.

Perry, T.O. 1964. Soil compaction and loblolly pine growth. U.S.D.A. Forest Service, Tree Planters' Notes: 67, p.9.

Pomery, K.B. 1949. The germination and establishment of loblolly pine under various surface soil conditions. Journal of Forestry, 47:541-543.

Poth, M. and D.D. Focht. 1985. $^{15}N$ kinetic analysis of $N_2O$ production by *Nitrosomonas europaea*: an examination of nitrifier denitrification. Appl. and Envr. Microbiol., 49(5):1134-1141.

Prinn, R., D. Cunnold, R. Rasmussen, P. Simmonds, F. Alyea, A. Crawford, P. Frasier, and R. Rosen. 1990. Atmospheric emissions and trends of nitrous oxide deduced from 10 years of ALE-GAGE data. J. Geophys. Res., 95D:18,369-18,385.

Rasmussen, R.A. and M.A.K. Khalil. 1984. Atmospheric methane in recent and ancient atmospheres: Concentrations, trends and interhemispheric gradient. J. Geophys. Res., 89:11,599-11,605.

Reaves, C.A. and A.W. Cooper. 1960. Stress distribution in soils under tractor loads. Agricultrual Engineering 41(1):20-21, 31.

Reichle, D.E. 1976. The role of invertebrates in nutrient cycling. Ecological Bulletin, 25:145-154.

Reddy, K.R. and W.H. Patrick Jr. 1975. Effect of alternate aerobic and anaerobic conditions on redox potential, organic matter decomposition, and nitrogen loss in a flooded soil. Soil Biol. Biochem., 7:87-94.

Rhode, H. 1990. A comparison of the contribution of various gases to the greenhouse effect. Science, 344:1217-1219.

Sahrawat, K.L. and D.R. Keeney. 1986. Nitrous oxide emission from soils. In Advances in Soil Science, Volume 4. Springer-Verlag, NY.

Schlesinger, W.H. 1977. Carbon balance in terrestrial detritus. Annual Review of Ecological Systems, 8:51-81.

Sheppard, J.C., H. Westberg, J.F. Hopper, and K. Ganesan. 1982. Inventory of global methane sources and their production rates. J. Geophys. Res., 87:1305-1312.

Skinner, M.F., G. Murphy, E.D. Robertson, and J.G. Firth. 1989. Deleterious effects on soil properties and the subsequent early growth of second-rotation radiata pine. In W.J. Dyck and C.A. Mees, eds. Research Strategies for Long-Term Site Productivity. Proceedings, IEA/BE A3 Workshop, Seattle, WA, August, 1988. IAE/BE A3 Report No. 8. Forest Research Institute Bulletin 152, Forest Research Institute, New Zealand.

Slemr, F., R. Conrad, and W. Seiler. 1984. Nitrous oxide emissions from fertilized and unfertilized soils in a subtropical region. Journal of Atmospheric Chemistry, 1:159-169.

Smith, C.J., R.D. Delaune, and W.H. Patrick, Jr. 1982. Carbon and nitrogen cycling in a Spartina alterniflora salt marsh. In J.R. Freney and I.E. Galvally, eds., The Cycling of Carbon, Nitrogen, Sulfur, and Phosporus in Terrestrial an Aquatic Ecosystems. Springer-Verlag, New York, pp. 97-104.

Soulides, D.A. and F.E. Allison. 1961. Effect of drying and freezing soils on carbon dioxide production, available mineral nutrients, aggregation, and bacterial population. Soil Sci., 91:291-298.

Stanford, G.J., M.H. Frere, and D.H. Schwaninger. 1973. Temperature coeffecient of soil nitrogen mineralization. Soil Sci., 115:321-323.

Stepniewski, W. and J. Glinski. 1988. Gas exchange and atmospheric properties of flooded soils. In D. Hook, ed. The ecology and management of wetlands volume 1: Ecology of wetlands. Timber Press, Portland, Oregon.

Stevenson, I.L. 1956. Some observations on the microbial activity in remoistened air-dried soils. Pl. Soil, 8:170-182.

Steudler, P.A., R.D. Bowden, J.M. Melillo, and J.D. Aber. 1989. Influence of nitrogen fertilization on methane uptake in temperate forest soils. Nature, 341:314-316.

Technicon Industrial Systems. 1978. Individual/simultaneous determination of nitogen and/or phosphorous in BD acid digests. Ind. Meth. 329-74W/B. Technicon Industrial Systems, Tarrytown, NY. 7p.

Thorud, D.B. and S.S. Frissel, Jr. 1976. Time changes in soil density following compaction under an Oak Forest. Minn. For. Res. Notes, no. 257. University of Minnesota, St. Paul.

Tiedje, J.M. 1982. Denitrification. pp. 1011-1026 in A.L. Page, R.H. Miller, D R. Keeney, eds., Methods of Soil Analysis, Part 2, Chemical and Microbiological Properties. American Society of Agronomy-Soil Science Society of America. Madison WI. 1159 pp.

Tiedje, J.M. 1988. Ecology of denitrification and dissimilatory nitrate reduction to ammonium. In A.J.B. Zehnder, ed., Biology of Anaerobic Microorganisms. John Wiley and Sons, NY.

Tippett, M.D. 1992. Impacts of timber harvesting on soil physical properties in wetlands. Masters thesis. Dept. of Forestry, Virginia Polytechnic Institute, Blacksburg, Virginia.

Torbert, H.A. and C.W. Wood. 1992. Effects of soil compaction and water-filled pore space on soil microbial activity and N losses. Commun. Soil Sci. Plant Anal., 23(11&12): 1321-1331.

Topp, G.C., J.L. Davis, and A.P. Annan. 1980. Electromagnetic determination of soil water content: measurements in coaxial transmission lines. Water Resources Research, 16(3):574-582.

Valiela, I. 1984. Marine Ecological Processes. Springer-Verlag, New York, 546 pp.

Veihmeyer, F.J. and A.H. Hendrickson. 1948. Soil density and root penetration. Soil Sci., 65:487-493.

Vogels, G.D., J.T. Keltjens, and C. Van der Drift. 1988. Biochemistry of methane production. In A.J.B. Zehnder (ed.) Biology of anaerobic microorganisms, pp. 707-770. Wiley and Sons, New York, NY.

Vomocil, J.A. and W.J. Flocker. 1961. Effect of soil compaction on storage and movement os soil air and water. Transactions, ASAE 4(2):242-46.

Wang, Z.P., R.D. Deluane, C.W. Lindau, and W.H. Patrick Jr. 1993. Methane emission as affected by the presence of rice and algae in a flooded soil. In Abstracts: Second Biennial Symposium on Biogeochemistry of Wetlands. Louisiana State Universty, Baton Rouge, LA.

Wert, S. and B.R. Thomas. 1981. Effects of skid roads on diameter, height, and volume growth in Douglas fir. Soil Sci Soc. Amer. J., 45:629-632.

Widdel, F. 1986. Growth of methanogenic bacteria in pure cultrue with 2-propanol and ohter alcohols as hydrogen donor. Appl. Env. Microbiol., 51:1056-1062.

Wijler, J. and C.C. Delwiche. 1954. Investigations on the denitrifying process in soil. Plant and Soil, 5:155-169.

Windsor, J., T.R. Moore, and N.T. Roulet. 1992. Episodic fluxes of methane from subarctic fens. Can. J. Soil. Sci., 72:441-452.

Witkamp, M. and M.L. Frank. 1969. Evolution of $CO_2$ from litter, humus and subsoil of a pine stand. Pedobiologia, 9: 358-365.

Wyckoff, R.D. and H.G. Botset. 1936. The flow of gas-liquid mixtures through unconsolidated sands. Physics, 7:325-345.

Zumdahl, S.S. 1986. Chemistry. D.C. Heath and Co., Lexington, MA.

Zumft, W.G. and P.M.H. Kronek. 1990. Metabolism of nitrous oxide. In N.P. Revsbech and J. Sørenson, ed.s. Denitrification in Soil and Sediment. Plenum Press, NY.

## APPENDIX I

**Typical chronology for a site visit.**
(There were two sites in the study.)

**Day One**

1) Install sixteen chamber bases on one of the two blocks contained in site.
2) Record surface profile data for determination of headspace volume.

**Day Two**

a.m.

1) Measure water-table depths
2) Measure soil temperature and moisture for eight of the sixteen microsites. (microsites randomly selected: 2 per control plot and 3 each per bedded plot)
3) Place chambers over the same microsites. Take initial gas sample.
4) At 30-40 minutes, take a second gas sample.
5) At 120 minutes, take a final sample.

Sample a subset of chambers where rates are likely to be high more intensively to determine if gas evolution is linear over the incubation period.

p.m.

Repeat for the remaining eight microsites.

**Days 3 & 4**

Repeat the above sequence for the second block.

**Day 5**

1) Collect soil samples.
2) Return to Athens.