

NUTRIENT LOSSES AND RELATED PROCESSES
IN A SEASONALLY-OPERATED SEPTIC BED SOIL
UNDER FAVOURABLE CONDITIONS

By

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ABSTRACT

This study assesses the processes of nutrient loss to ground-water and adjacent soils from a seasonally-operated septic system operating under favourable conditions.

In published studies, highly variable results precluded generalizations about nutrient losses from septic systems. For these reasons, a detailed source-to-sink study of a single septic bed was undertaken to (a) estimate nutrient losses from the bed and (b) identify the intensity and spatial distribution of effluent processes within a septic bed soil.

A research site 40 km northeast of Parry Sound on Whitestone Lake was chosen as typical of cottage waste disposal systems. Sampling surveys using 4 deep soil⁴² inspection pits and 137 coring sites in the septic bed soil and nearby field and woodland soils were undertaken during the summers of 1974, 1975 and 1976. Spatial mapping of soil properties, multivariate analysis and modelling of soil solution and ions were used to interpret the measurements. Soil solutes, including nutrients, and environmental soil conditions in the septic bed soil and site area were mapped. Nutrient and soil water fluxes in the septic bed and reference sites were estimated from a mathematical model.

The reference sites are characterized by plant cycling of soil solutes, combined with small vertical leaching losses following rainfall events. The septic bed soil, when in equilibrium with large

effluent input, experiences continuous large vertical and lateral leaching losses. As the septic bed soil begins to receive effluent, leaching accelerates and nutrient fluxes increase, becoming more uniform. The septic bed becomes anaerobic (oxygen diffusion rate (ODR) $< 20 \mu\text{g cm}^{-2} \text{min}^{-1}$) from its centre outward at 30-60 cm depth, and within 7 days, the soil surface and lateral sides of the bed are anaerobic. Precipitation of Ca and PO_4 and lowering of PO_4 and Ca levels in solution occurs after two weeks of operation. A marked drop in ionic levels occurs less than two weeks after effluent input ceases. Plant uptake of nutrients becomes larger than leaching losses as leaching by periodic rainfall cycles becomes apparent.

The distribution of nutrient fluxes and decomposition processes varies with depth and from place to place in the septic bed. For example, large lateral effluent fluxes out of the septic bed are associated with small nitrate fluxes ($0.659 \times 10^{-5} \text{ meq cm}^{-3} \text{ d}^{-1}$), and anaerobiosis ($\text{ODR} < 18 \mu\text{g cm}^{-2} \text{min}^{-1}$). Conversely, large nitrate fluxes ($0.150 \times 10^{-5} \text{ meq cm}^{-3} \text{ d}^{-1}$) are associated with moderate effluent fluxes and aerobic soil ($\text{ODR} > 22 \mu\text{g cm}^{-2} \text{min}^{-1}$). The spatial pattern of effluent fluxes also varies markedly within the septic bed, from small net inflows of groundwater (0.016 cm/d), to large effluent outflows, up to 49 cm/d .

Mathematical modelling indicates losses of NO_3 and PO_4 of 0.234×10^{-5} and $0.131 \times 10^{-4} \text{ meq cm}^{-3} \text{ d}^{-1}$, respectively, 10^3 and 10^2 greater than in the reference sites. Increased losses of NH_4 , at $0.915 \times 10^{-7} \text{ meq cm}^{-3} \text{ d}^{-1}$ are less (10^1 to 10^2), and represent less than 12% of the nitrogen losses from the septic bed. This indicates that serious

environmental pollution can occur from a septic system operating under favourable conditions.

Large leaching losses of sodium, up to $0.609 \times 10^{-4} \text{ meq cm}^{-3} \text{ d}^{-1}$, from the septic bed soil C horizon indicate long-term changes in soil fabric are occurring, which will probably accelerate effluent flux losses from the septic bed soil, thereby increasing surface water and groundwater pollution. It is concluded that similar estimates of nutrient fluxes must be gathered from many sites before this environmental pollution source can be accurately estimated or effectively controlled.

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CHAPTER I

INTRODUCTION

This thesis examines the mechanisms of and the potential for pollution from a typical cottage soil septic system. Levels and fluxes of soil pollutants, including nitrogen and phosphorus ions, are measured then compared with those levels existing in nearby cropped field and forest soils. Some of the processes controlling the levels, the transport and the transformation of pollutants are identified and quantified as precisely as possible.

A septic system is traditionally used to dispose of domestic wastes on the same lot where a house is located. It consists of a septic tank and septic tile bed. The septic tank is a rectangular or cylindrical container through which waste slowly flows, allowing solid particles to settle and decompose on the bottom of the container. The effluent released from the septic tank contains water and partially-decomposed waste with both dissolved material and transported small solid particles in suspension. The septic bed consists of drainage pipes or tiles placed below the soil surface and connected to the septic tank outlet. Perforated pipes or clay tiles separated at the joints allow effluent to seep into the soil. Around the tile, sand and gravel is usually added to increase effluent percolation and aeration. Most of the water in the effluent flows through the septic bed and out

of the sides and bottom of the bed. Wastes are altered and removed as they are carried through the bed by the water in the effluent. If this alteration and removal is inadequate, water flowing out of the bed will still contain waste products potentially harmful to humans, animals or the environment. Most of these substances leaking out of the septic bed reach and disperse in the groundwater or reach surface water.

Although stability of substances in groundwater varies, the rates of alteration and removal of waste products are reduced in groundwater and these substances may reach surface streams or lakes intact. One of the objectives of this thesis is to evaluate these assumptions with a detailed assessment of the changes in effluent properties and processes spatially and with time as the effluent flows through the septic bed and into the adjacent field.

Harmful effects from waste products include health hazards to humans or animals and eutrophication of nearby lakes and streams. Eutrophication is an abundance of biological growth in lakes and streams and is usually accompanied by low oxygen levels. Pollutant substances from septic systems include pathogenic organisms, toxic chemicals and agents causing eutrophication. Eutrophication in lakes and streams can be caused by decreasing the oxygen levels or increasing the quantity of nitrogen and phosphorus in water. When oxygen is removed from lakes or streams, decomposition becomes slow, permitting organic material to accumulate. This organic accumulation supports an increase in eutrophication. Since the lack of nitrogen and phosphorus is usually limiting to biological growth, an increase in N and P levels in streams and lakes will also cause eutrophication, as is well known

(Ryden et al., 1973).

Septic systems are used where distances between houses are great, and where sewers are prohibitively expensive to install. In most rural and agricultural environments, septic drainage beds are dispersed and their use has not caused major visible problems. Under such favourable conditions, adverse effects are limited to one local but possibly contentious possibility--eutrophication from losses of nitrogen and phosphorus in soluble ionic form. Pathogenic organisms die rapidly outside the human body since the soil is a hostile environment for them (Ziebell et al., 1975; Bouma, 1975). Thus only in poorly-designed septic systems do these organisms reach groundwater lakes or streams. Heavy metals and the complex molecules in detergents, such as surfactants, cannot be safely processed in soil, and are best removed in the manufacturing process and excluded from septic systems as far as possible. Organic material in the household waste consumes oxygen as it decomposes, lowering oxygen levels and causing eutrophication; however, the recommended installation of septic systems is usually far enough from lakes and streams to prevent such effects of inhibition and oxidation. Since most soil and groundwater contains some oxygen, substances which consume oxygen are slowly decomposed as escaping effluent flows from the septic bed to nearby lakes and streams. Thus, by the time effluent reaches lakes and streams, it has a very low organic content and oxygen demand. As previously noted, small increases in nitrogen and phosphorus levels in lakes and streams can also cause eutrophication. The inorganic or ionic forms of nitrogen, NO_3^- and NH_4^+ , and phosphorus, H_2PO_4^- , HPO_4^{2-} and PO_4^{3-} , are more mobile and

available to micro-organisms in soil and water than the organic forms of nitrogen and phosphorus (Ryden et al., 1973). Unlike other causes of eutrophication, nitrogen and phosphorus ions tend to be stable in groundwater and decompose little. Thus, some authors contend that the most prevalent and serious environmental threat from most septic systems is the loss of nitrogen and phosphorus in ionic form (Bolt and Bruggenwert, 1976; Deamish, 1972; Robinson, 1971; personal communication).

Conversely, others claim that this is not a serious problem, most notably Bouma (1975), who asserts that "the conventional soil adsorption system for on-site disposal of septic tank effluent has worked well for many years in many different soils..." and "Generally, pathogenic bacteria and viruses should be completely removed within a specified soil volume and the BOD, suspended solids and N and P concentrations should be lowered to concentrations that would allow human consumption and addition to surface waters without creating adverse effects."

This divergence of interpretation arises from limitations in the state of the art of studies on septic pollution. Most field studies of septic pollution survey several septic beds for pollutant concentrations and pollutant-sensitive properties from several sites in and around each septic bed. Most laboratory studies of septic pollution use several large rigid columns packed with soil representative of the soil materials in the septic bed. Septic tank effluent introduced at the top of such a column percolates down the column, eventually draining out the bottom, not the sides. Although field and laboratory approaches have comparative advantages and disadvantages, both share assumptions which reflect present knowledge, which

assumptions are as follows:

1. Since measurements of several sites over a limited time period are used to represent the system - mainly to save time and money in routine monitoring - it is assumed that the septic bed can be studied as a homogeneous system. This implies that average levels are more important than variations from place to place and time to time, which should be or are assumed to be small. Using soil column studies in the laboratory has enabled limited determinations of the degree to which pollutant levels and soil processes vary over time. In beds used seasonally, there is some concern whether nutrients are stored over the winter period in septic beds and then released in the spring. The release of nutrients from manure and fertilizers during the spring thaw has been studied (Chichester, 1977; Klausner et al., 1976; Burwell et al., 1976; Duffy et al., 1975; Gambrell et al., 1975b; Baker et al., 1975), but not from septic systems. Also, the initial response of the septic bed soil to initial use or cessation of use has not been studied in actual field samples.

2. All significant septic pollution can be estimated and detected from concentration measurements. Previous research (Campbell, 1974), using many sites, indicated that it was difficult to relate measured concentrations of pollutants in soil and water sinks to nearby septic system sources. In the literature, the variations in pollutant levels associated with septic beds are large and attempts at estimating the extent of septic pollution over large areas or in dissimilar fields have

resulted in widely divergent values.¹ This lack of consensus questions the usefulness of concentration measurements. Some flux measurements have been made using soil columns in the laboratory and through the methods of Bower (1961, 1962) and Bouma et al. (1971) and Bouma and Denning (1972) in the field. These, however, are exceptions. Both field and laboratory techniques have only been used to produce estimates of vertical effluent flow for a single point location. Lateral flow and spatial variability are not assessed. Also, the field techniques developed by Bower, Bouma and Denning are laborious and disturb the site sampled, making them unsuitable for continuous flow measurement. This available information, therefore, most of which is expressed as concentration measurements, has led many authors to conclude that nutrient losses from most septic systems are low and do not pollute or produce negligible pollution.

3. In restricting measurements to pollutants and a few soil characteristics known to be sensitive to effluent, implicit assumptions have been made. It has been assumed that only soil gases and the composition of the soil solution vary; with other soil properties, especially the soil solid components, being unaffected. If the solid components do not change, a septic system's life expectancy is almost infinite and thus the system need not be monitored. In most temperate agricultural and natural soils, soil processes change the soil solid components only

¹This is discussed in detail in Chapter II in reviewing published pollutant levels in septic systems.

very slowly. Thus the assumption is made that these processes are operating at similarly slow rates in septic bed soils.

In the studies mentioned above, an initial decision was made to study more than one septic bed so that the results could be generalized. Studying several beds limits the detail of the study, such as the properties and processes which can be measured and their spatial and temporal variation. To study more than one bed, simplifying assumptions are necessary to bring the volume of work within the constraints of available time and funds. Additional constraints of technology and the researcher's knowledge may also exist. The measurement technology is such that certain pollutants can be easily determined (e.g., NO_3^- and NH_4^+) whereas others are difficult to measure (e.g., H_2PO_4^- , HPO_4^{2-} and PO_4^{3-}). Since satisfactory direct methods of measuring fluxes have not been developed, fluxes are computed from a series of measurements. This may explain in part why few flux measurements have been made. In perspective, much of this initial extensive research has focussed on high levels of septic pollution from septic systems operating under unfavourable conditions. In contrast, this study addresses more subtle moderate pollution from a septic system operating under favourable conditions.

At the risk of being accused of producing a study which is site specific, it was felt that the only way to test the validity of the above assumptions and evaluate a serious, if contentious, pollution threat would be to study in detail a single septic bed soil operating under favourable conditions. As this system is typical of a seasonally-

used septic system, it is hypothesized that moderate losses of nitrogen and phosphorus ions have gone undetected from many septic systems for many years. The hypotheses or assumptions on which this study is based are opposite to the assumptions on which most extensive studies of septic pollution are based. These assumptions are:

1. The septic bed soil is a heterogeneous system which cannot be represented by average levels. Spatial and temporal variations in soil properties are large and significant. The response of the septic bed soil when use begins or ceases is also considerable and significant.
2. Estimates of fluxes are necessary to quantify septic pollution, in addition to any measurements of concentration of components. Nutrient losses of nitrogen and phosphorus in ionic form could be significant, even under favourable conditions, and can only be detected by estimates of nutrient flux.
3. The solid components of the soil may be changing. Processes may be operating which have not previously been linked with soils in septic beds; also, the intensity of soil processes may not be the same in both natural and septic bed soils.

To test these three assumptions, specific measurements are required. For the first assumption, soil properties in the various parts of the septic bed must be studied, as well as around or adjacent to it. The variation in these properties, as the use of the bed begins

and ceases, must also be determined. The first assumption can then be evaluated by comparing the variation in soil properties within the studied septic bed to that reported in the literature as existing between specific septic beds. If the variation within the studied septic bed exceeds that between septic beds, it suggests that using average levels to characterize septic beds is invalid and that this and further detailed site-specific research is necessary to determine how to characterize septic bed soils.

The second assumption requires estimates of the fluxes of water and of dissolved soil ions. From the literature², mathematical algorithms to estimate these fluxes from soil properties were translated into Fortran computer programs. The third assumption is an indication of the broad scope of soil properties measured. In addition to known pollutants and pollutant-sensitive soil properties, soil properties indicating soil processes not previously associated with septic effluent are measured. For example, the relationships between sodium ion levels, soil acidity and leaching intensity, and soil calcium ion levels and plant recycling are not usually considered in soil pollution research, but are included in this study because they give a more complete characterization of the soil environments. This facilitated the study of known pollutant processes such as nitrification, and, in addition, revealed previously unreported changes in soil fabric and soil ionic levels.

²The mathematical algorithms are discussed in section 3.4 and their programming is discussed in Appendix C.

In the following five chapters, these measurements are described, analyzed and the results tested against these assumptions and a few other measurements from the literature. Chapter II includes a brief review of the literature pertaining to the relevant soil processes and a summary of the work on nutrient levels in septic systems. This summary illustrates the limited usefulness of studies based on average levels of pollutants. In Chapter III, the research design is discussed, including the choice of a research site and of the soil properties measured. The methods of measuring these properties, both spatially and temporally, are explained, as are the methods of modelling the system. In Chapter IV, the variation in soil properties is used to determine the nature and duration of the soil environments and pollution in the bed and adjacent field. For example, it was evident where much of the nitrate from the effluent was being nitrified and denitrified or lost in seepage to the adjacent field. The extent to which nitrate could be stored over winter in a seasonally-used bed was also evident. In Chapter V, the calculated estimates of nutrient and water fluxes are discussed. This includes vertical and lateral fluxes in the centre and at the edge of the septic bed. These are compared with vertical fluxes in the field and forest soils. The response in flux magnitude after use of the bed commences is also discussed. The conclusions and a summary of the research are presented in Chapter VI. This includes assessment of the validity of the assumptions on which the study is based and an estimate of the amount of soil pollution occurring. The appendices include outlines of analytical methodology where these are different from standard procedures and also the computer programs are presented.

CHAPTER II

SOIL PROCESSES AND POLLUTANT LEVELS IN SEPTIC SYSTEMS

This chapter outlines the soil processes which occur in septic beds, including pollutant adsorption, degradation and transport. Also reviewed are the practical techniques for altering these processes. From a review of reported levels of pollutants in septic beds, it is evident that a new approach is needed to study septic bed soils.

2.1 Soil Processes

Septic tank effluent leaves the septic tile and flows through the septic bed soil, eventually reaching groundwater after a variable 'residence' time period. In flowing through the septic bed, the effluent usually (e.g., Walker *et al.*, 1973a; Bouma, 1974) passes through four soil environments as portrayed schematically in Fig. 2.1:

1) aerobic sand and gravel, 2) anaerobic sand and gravel, 3) adjacent aerobic soil and, at depth, 4) anaerobic soil. As effluent flows through these environments, decomposition processes act on the nitrogen and carbon substances within it (Table 2.1). The adsorption and precipitation processes controlling phosphorus losses are apparently independent of the degree of anaerobiosis (de Haan, 1975) (Table 2.1). The nitrogen in the effluent entering the septic bed soil is in the form of organic nitrogen and ammonia (Otis *et al.*, 1975). In the aerobic sand

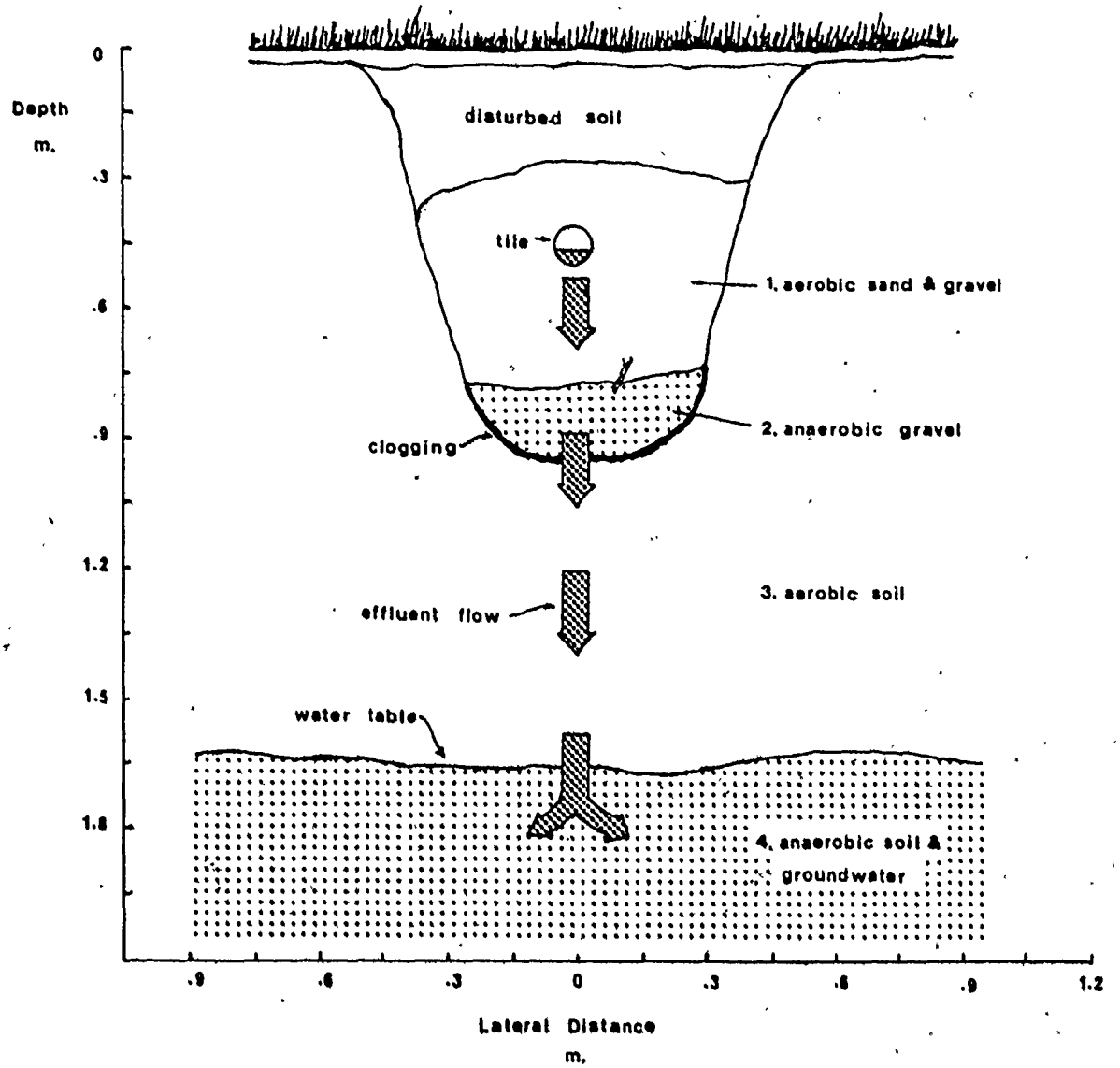


FIG. 2.1 'Typical septic bed soil environments

TABLE 2.1 Processes Removing Nitrogen, Phosphorus and Carbon in Septic Bed Soils

Substance	Form of substance from septic tank	Soil environments and processes				Form of pollutant reaching groundwater
		1. aerobic	sand and gravel	adjacent soil	4. anaerobic	
nitrogen	organic N and NH_4	1. org. N \rightarrow NH_4	$\text{NO}_3 \rightarrow \text{N}_2$ denitrification if carbon available	1. no denitrification 2. NO_3 transport	$\text{NO}_3 \rightarrow \text{N}_2$ denitrification if carbon available	NO_3
		2. $\text{NH}_4 \rightarrow \text{NO}_3$				
		3. NH_4 adsorbed				
carbon	organic carbon	org. C \rightarrow CO_2 fast biological decomposition	1. org. C \rightarrow CH_4 slow biological decomposition	org. C \rightarrow CO_2 fast biological decomposition	1. org. C \rightarrow CH_4 slow biological decomposition	soluble organic carbon
			2. org. C and CH_4 \rightarrow assimilated during denitrification		2. org. C and CH_4 \rightarrow assimilated during denitrification	
phosphorus	organic P and inorganic P	1. org. P \rightarrow inorg. P	1. org. P \rightarrow inorg. P	1. org. P \rightarrow inorg. P	1. org. P \rightarrow inorg. P	inorganic P
		2. adsorption-precipitation on Ca, Mg, Al, Fe compounds	2. adsorption-precipitation on Ca, Mg, Al, Fe compounds	2. adsorption-precipitation on Ca, Mg, Al, Fe compounds	2. adsorption-precipitation on Ca, Mg, Al, Fe compounds	

and gravel, this is rapidly adsorbed on the soil exchange complex and converted to nitrate. As the nitrate enters the anaerobic sand and gravel, it is denitrified to nitrogen gas (N_2), providing carbon is present. Any nitrate which is not thus denitrified is leached through the underlying aerobic soil until it reaches a stream or lake. Such nitrate loss is most likely if the groundwater has a low soluble carbon content and/or is high in dissolved oxygen. Organic carbon in the tile effluent is rapidly decomposed to carbon dioxide in aerobic sand or gravel. Any carbon reaching the anaerobic sand and gravel beneath 2) is slowly decomposed to hydrocarbons, e.g., methane. These compounds are assimilated during denitrification in anaerobic environments and respired as carbon dioxide. If oxygen or nitrate is limiting, some carbon will be lost in pollutant form as dissolved carbon and move to 3) with the effluent flow.

Phosphorus enters the septic bed soil in both organic and inorganic forms (Swahney and Hill, 1975). Organic phosphorus (Taylor and Ashcroft, 1972) is rapidly adsorbed and converted to inorganic phosphorus in the aerobic and anaerobic sand and gravel. If calcium, aluminum or iron compounds are present, inorganic phosphorus is rapidly adsorbed and co-precipitated irrespective of the degree of anaerobiosis (Bower, 1971). This adsorption is more complete in the adjacent clay or loam soils with a higher particle or specific surface area and more cation adsorption sites than sand and gravel (Buckman and Brady, 1960; Banin and Amiel, 1970). If calcium, aluminum or iron compounds are lacking, inorganic phosphorus is carried through the system, producing significant phosphorus losses. Three processes, therefore, control the

losses of phosphorus and nitrate in ionic form from the septic bed: precipitation-adsorption reactions controlling phosphate levels; denitrification controlling nitrate levels; and the rate of effluent transport through the septic bed soil.

A number of studies on the adsorption of nitrogen and phosphorus have been carried out. Lance (1972) reported that cation exchange was important in holding NH_4 in loamy sands with a cation exchange capacity 2-2.5 meq./100g. until it could be nitrified. He used columns 10 cm x 2.5 m of disturbed soil that were flooded with secondary sewage effluent for 2, 9, 16 and 23 days followed by 5 dry days between inundations. Engelstad (1970) reported that clay soils were able to 'hold' nitrates against considerable infiltration of rain water. Hill (1972) reported a bonding affinity or adsorption sequence of $\text{NO}_3 = \text{Cl}_2 > \text{PO}_4 > \text{SO}_4$ and the desorption sequence of $\text{PO}_4 < \text{SO}_4 < \text{NO}_3 = \text{Cl}_2$, suggesting that phosphate and sulfate soil ions would be leached or lost more readily than nitrate ions. Bower (1971), Beek and Frissel (1973) and Magdoff et al. (1974) used equilibrium theory to compute the ammonia adsorbed. Magdoff et al. (1974) predicted ammonia adsorption in a septic bed soil. In this study, adsorption of cations was calculated, using a similar approach to that of Beek and Frissel (1973).

The recent work of Shah et al. (1975), Novak et al. (1975), Novak and Adriano (1975), Swahney and Hill (1975), and that reported in Bolt and Bruggenwert (1976) indicates that phosphorus adsorption is a two-step process. The first step is adsorption of phosphorus ions onto clay and organic matter. The second is rapid precipitation of adsorbed phosphorus followed by a regeneration of the precipitation

sites. This precipitation is irreversible and functions as a phosphorus sink. Little is known about regeneration rates or the factors which control it. In the first step, clay and organic matter act as catalysts, bringing phosphorus ions into contact with other substances to facilitate the precipitation reaction. Organic phosphorus seems to precipitate directly, without preliminary adsorption on clay surfaces (Taylor and Ashcroft, 1972) and most strongly in the presence of iron oxides or humic and fulvic acid complexes. Phosphorus ions precipitate in a number of ways. They can be irreversibly adsorbed on the surfaces of iron and aluminum oxides and calcium and magnesium compounds (Bower, 1971), or they can replace ions within the matrices of iron and aluminum oxides and compounds (Engelstad, 1970) and aluminosilicates (Hill, 1972; Ryden et al., 1973). Iron and aluminum oxides have the greatest capacity as precipitation sites, followed by calcium and magnesium. The net effect, termed chemical sieving by Swahney and Hill (1975), is phosphorus removal from effluent before it reaches groundwater. Almost complete chemical sieving is normal for natural systems, especially forests (Ryden et al., 1973; Bolt and Bruggenwert, 1976), and for most septic systems. For example, Swahney and Hill found that in one septic system, phosphorus transport was one meter in fifteen years, logically giving an infinite life expectancy for phosphorus precipitation. Phosphorus adsorption in the septic bed can be increased by adding iron, aluminum, calcium and magnesium compounds, as reported by Deamish (1972) and Laak et al. (1975).

While phosphorus adsorption and precipitation have been quantified and modelled (Shah et al., 1975; Novak et al., 1975; Novak and

Adriano, 1975), regeneration has not, thus precluding long term models of the process. In addition, the long term implications of this increasing phosphorus sink are unknown (Larsen, 1974).

The decomposition of nitrogen has been studied in the following manner. Walter (1974) has summarized the chemical equations and optimum conditions for nitrification and denitrification (Table 2.2). Of the various forms of soil nitrogen in septic bed soils, nitrate is the most serious potential pollutant. Organic nitrogen is relatively immobile, ammonia volatilizes and is rapidly adsorbed, and of the other inorganic nitrogen forms, only nitrate is stable in the soil. Denitrification accounts for most soil nitrate reduction (Lance and Whisler, 1974). For these reasons, estimates of denitrification were made for this study.

Denitrifying bacteria use gaseous oxygen preferentially (Skinner, 1968). Therefore, for denitrification to occur, the oxygen content must be zero (Skinner, 1968; Broadbent, 1971). Cady and Bartholomew (1961) found that the denitrification rate was determined by the oxygen concentration or oxygen diffusion rate (ODR) and the biological oxygen demand exerted by the microbial population. Anomine (1962) found that denitrification occurred only in soils with both oxidized and reduced layers. He also found that the drop in redox potential (Eh) after waterlogging depended on the extent of prior drying of soils and upon the quantity of readily decomposable organic matter. Kimble et al. (1972) found that the potential for denitrification decreased geometrically with depth due to a lack of energy sources for microbial activity.

The identification of the aerobic-anaerobic interface which divides denitrification from nitrification would be most interesting

TABLE 2.2 Chemical Equations and Optimum Conditions for Nitrification and Denitrification*

Process	Equation	Optimum conditions		
		Temp. °C.	Moisture content % vol.	pH Oxygen
nitrification	$\text{NH}_4 + \text{O}_2 \longrightarrow \text{NO}_2 + 4\text{H} + \text{energy}$	30-35	50-65	7.0-9.0 aerobic
denitrification	$\text{C}_6\text{H}_{12}\text{O}_6 + 4\text{NO}_3 \longrightarrow 6\text{CO}_2 + 6\text{H}_2\text{O} + 2\text{N}_2$	60-65	100	4.8-8.0 anaerobic

* Walter (1974).

in this regard (Table 2.3). Brandt (1964) identified the interface at an ODR of $20 \mu\text{g cm}^{-2}\text{sec}^{-1}$. Anomine (1962), Turner and Patrick (1968) and Rich (1973) found that the shift from anaerobic to aerobic occurred at an Eh of 330 to 350 mv³. Anomine and Rich also identified the Eh range for reduction of NO_3 at an Eh of 350 to 100 mv. Lähde (1969) found that the aerobic-anaerobic shift occurred at a respiration quotient (RQ)⁴ of 0.80 and a gaseous oxygen content of 10% in the macropores.

TABLE 2.3

Aerobic-Anaerobic Interface

<u>Measure</u>	<u>Interface Value</u>
RQ	0.80
ODR	$20 \mu\text{g cm}^{-2}\text{min}^{-1}$
O_2 pct. macropores	10 pct.
O_2 pct. micropores	0.0 pct.
Eh	350 mv

In this study, ODR was measured and Brandt's figure of an ODR of $20 \mu\text{g cm}^{-2}\text{min}^{-1}$ was used to identify environments where denitrification would be possible. It should also be noted that chemical decomposition or denitrification can occur (Bremner and Nelson, 1968; Buresh et al., 1976).

³All values are standardized to Eh₇ or pH = 7.0.

⁴The ratio of oxygen consumed to carbon dioxide produced (Bell and Coombe et al., 1967; Lähde, 1969).

All the methods of measurement of denitrification determine the process indirectly and are not very satisfactory.. Pratt et al. (1972), using transit time equations, measured denitrification as a residual after N input, plant uptake of N and leaching losses had been assessed. Mann et al. (1972) used denitrification constants. Kimble et al. (1972) used nitrate to chloride ratios to distinguish leaching from denitrification, for when this ratio is decreased, denitrification is indicated. In such an approach, care must be taken to account for plant uptake of chloride when it occurs. In the present study, nitrate to chloride ratios were so used to estimate denitrification.

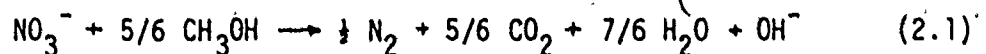
There are five ways by which the rate of denitrification can be influenced in septic bed soils: (i) by soil 'clogging', (ii) by resting (not using) the system or controlling dosing cycles, (iii) additions of organic substrate or carbon, (iv) artificial moisture barriers and (v) effluent cycling.

Magdoff and Bouma (1975) describe soil clogging as an inhibiting layer at the infiltrative surface. Where septic tile has been laid in gravel trenches, the clogged layer or crust forms at the interface of the gravel and soil, as indicated in Fig. 2.1. The crust is an accumulation of organic matter deposited during continuous effluent flow. The saturated hydraulic conductivity of the crust is much lower than that for the gravel in the tile trench, and markedly uniform at around 8 cm/hr, thus ponding effluent in the bottom of the gravel trench. The ponded effluent becomes anaerobic and denitrification occurs if nitrogen and carbon are present (Walker et al., 1973a).

The manipulation of dosing cycles of land waste disposal

systems has been discussed by many authors, Gilbert et al. (1974), Lance and Whisler (1972, 1974), Converse et al. (1975), Lance et al. (1973, 1976), Bouma et al. (1974a) and Magdoff and Bouma (1975). Such a technique influences both the removal of pathogenic bacteria by controlling the degree of unsaturated flow in the bed and nitrogen removal by controlling the magnitude of effluent flux and crust formation. The flux must be small enough to permit sufficient oxygenation by diffusion from the atmosphere to maintain nitrification in the upper part of the bed and large enough to maintain saturation and anaerobic denitrification in the lower part of the bed. Resting and dosing cycles influence the development and dissipation of such soil crusts. Converse et al. (1975) recommend a once or twice daily dosing regime for septic systems.

Additions of organic carbon substrate to enhance denitrification have been discussed by Lance and Whisler (1974), Whisler et al. (1974), Mann et al. (1972), Sikora and Keeney (1975) and Walter (1974). The amount of carbon required for denitrification of a measured quantity of organic N can be computed from the methane equation (Sikora and Keeney, 1975):



Theoretically, 1.3 mg of carbon are required for every 1 mg of nitrate reduced to nitrogen gas. Gambrell et al. (1975a) applied these principles to field data to determine if carbon was limiting denitrification. Lance and Whisler (1974) reduced nitrogen content by 80-90% in soil columns by additions of soluble carbon as glucose. The resultant C/N

ratio was about 1:1. Sikora and Keeney (1975) used methanol (CH_3OH) at a ratio of 2.6 mg carbon to 1 mg nitrate present in the effluent and compared it with other carbon energy sources. Mann et al. (1972) indicated the feasibility of using sulfur as a substrate for denitrification. Mann et al. suggest: "the addition of sulfur to energy deficient soils may be more practical than additions of exogenous carbon since competition for available carbon among nondenitrifying bacteria is also keen".

Artificial moisture barriers are discussed by Bouma (1974, 1975) for use in innovative septic bed soil systems. Where soil permeability is too high, moisture barriers placed deep in the profile serve to pond the effluent, creating an anaerobic environment where denitrification can occur.

Recycling is expensive, as it requires pumping and may require monitoring. It is discussed by Lance and Whisler (1974), Lance et al. (1976), and Hines and Favreau (1975). In land waste disposal systems, high nitrate peaks can be redistributed or recirculated through the disposal system. Theoretically, this can remove more than 90% of the nitrogen. The system designed by Hines and Favreau recirculated effluent through a sand filter. Although biological oxygen demand and total suspended solids are low at 4 and 5 mg/l respectively, the removal of nitrogen and phosphorus has not been evaluated.

The nitrogen and phosphorus which are not adsorbed, precipitated, or removed by decomposition from the effluent will be lost from the septic bed by water flowing out of the septic bed. These losses can be estimated from the amount of nitrogen and phosphorus in solution and

the rate of movement of the soil water. Most models of moisture flow are based on Darcy flow or analogies to Darcy flow. From Darcy's law, moisture can be described after Taylor and Ashcroft (1972) as:

$$J_w = -K_w \frac{d\psi_h}{ds} \quad \text{or} \quad J_w = \frac{Q_w}{At} \quad (2.2)$$

where J_w is water flux density

K_w is hydraulic conductivity

Q_w is water flux

$\frac{d\psi_h}{ds}$ is the hydraulic potential gradient

ψ_h is the hydraulic potential

A is cross-sectional area

t is time.

As Darcy worked with saturated sand, his law is not strictly valid for all unsaturated flow conditions. However, for most unsaturated flow conditions, Darcy flow provides an adequate description (Kirkham and Powers, 1972). Soil moisture fluxes are manipulated in septic bed soils by: (i) additions of sand and gravel or (ii) the use of moisture barriers as previously discussed (p.22). Sand and gravel were first used in septic bed soils to increase infiltration, maintain unsaturated flow and prevent surface ponding of effluent (Schwiesow, 1975; Bouma, 1974) and hydraulic failure or basement flooding (Beatty and Bouma, 1973). Coincidentally, unsaturated flow and associated aerobic conditions also destroy pathogenic bacteria and enhance nitrification

(previously discussed, pp. 3, 17, 21) and biodegradation in general (Ziebell et al., 1975).

Since effluent loading tends to be constant (see page 27) and represents most of the moisture in the bed, these soil processes and the properties they influence tend to approach an equilibrium after several weeks of continuous operation of the septic system. Bouma (1975) found that the crust or clogged layer, and effluent flow were relatively uniform between different septic beds irrespective of age. For example, moisture tension below the crust varied from 23 to 27 cm and effluent flux 8 to 11 cm/d. The work done by Gilbert et al. (1974) on processing secondary sewage effluent in recharge basins suggests that stable oxygen and nitrogen regimes are developed. Column work with dosing cycles suggests that these oxygen, nitrogen and phosphorus regimes are reproducible (Magdoff et al., 1974; Whisler et al., 1974; Lance, 1972).

2.2 Pollutant Levels in Septic Systems

Septic systems represent such an intense nutrient loading that the septic effluent far outweighs the magnitude of other sources of nutrients and water associated with the septic bed. For example, results from this study indicated that, on average, 85% of the soil moisture in the septic bed and 99.8% of the nitrate and 98.3% of the phosphate flow to groundwater from the bed can be attributed to the septic effluent.

Few measurements of nutrient flux out of septic beds are available. This necessitates reliance on measurements of pollutant

levels which are soil and site specific and sensitive to distance from the septic bed. Table 2.4 is a summary of such reported information. Due to the diversity of sample site locations, measurements have been selected (i) from the septic tank outlet pipe, (ii) close to the edges of the septic bed soil and (iii) from groundwater measurements below the septic bed. The first set of measurements indicates the characteristics of the septic tank effluent; the second set indicates the nature of the effluent leaving the septic bed soil and the third set indicates the degree of groundwater pollution occurring. The variation in measured levels is large and, lacking flux estimates, nitrogen and phosphorus losses cannot be evaluated.

Some patterns are apparent, however. Total suspended solids (TSS) and oxygen demand (BOD, COD) levels in septic tank effluent are high, > 50 , > 100 and 200 mg/l respectively; and are reduced to < 50 , < 100 and 150 mg/l respectively, in effluent losses from the septic bed soil. Phosphate concentrations in septic tank effluent are fairly uniform at 10 to 20 mg/l. Phosphate removal in septic bed soil varies from nil to almost complete removal at 10^{-5} mg/l. In septic tank effluent, most nitrogen occurs as ammonia, whereas in effluent leaving the bed, it is in the form of nitrate. High levels of ammonia from septic tank effluent, from 30 to 90 mg/l, are usually converted to nitrate in the septic bed, producing low ammonia levels and, lacking denitrification, high nitrate levels. Thus, despite low nitrate levels in septic tank effluent, 2.2 to 0.03 mg/l, effluent losses from the septic bed and ground water may contain up to 8 mg/l of nitrate.

Effluent bacterial levels vary markedly from 10^8 to 10^0 organisms per

TABLE 2.4 Pollutant Levels in Septic Systems

Property	(i) Septic tank effluent			(ii) Septic bed soil effluent			(iii) Groundwater nearby		
pH	6.9 ^a	7.3 ^b		4.79			3.79		
TSS	176 ^a	51 ^b	55 ^d	27 ^d	49 ^a				
BOD	200 ^a	158 ^b	134 ^c	90 ^e	130 ^k	0 ^e	12 ^k		
COO	568 ^a	397 ^c	256 ^e	257 ^f	297 ^k				
TOD	73 ^a								
PO ₄	11.6 ^a	11.5 ^a	11.6 ^d						
Org. P	3.1 ^b								
Tot. P	15 ^a	15 ^e	21 ^f	11.8 ^g					
NH ₄	97 ^a	39 ^b	36 ^d	56 ^e	66 ^h	21x10 ⁻⁶ g	1x10 ⁻⁵ g		
NO ₃	0.028 ^a	0.6 ^b	0.51 ^d	1 ^e	2.2 ^k	7 ^k	0.062 ^h	1.0 ^k	0.86 ⁿ
Org. N	16.0 ^b	6 ^c	14 ^b						
Tot. N	3 ^c	52 ^d	43 ^f	49 ^k					
Fe	2.63 ^a								
Cl ₂	53 ^a	105 ^b							
FS	3800 ^b	2.7 ^d	1 ^e	38x10 ³ f	0.73 ^k	0.018 ^e	0.97 ^k		
FC	42x10 ³ b	86 ^d	25 ^e	17x10 ⁴ f	60 ^k	0.05 ^e	2.4 ^k	230 ^m	45x10 ⁵ m
TC	34x10 ⁵ b	370 ^e	54x10 ⁵ f	483 ^k	606x10 ⁴ p	0.54 ^e	9.7 ^k	41x10 ² m	68x10 ⁵ m
PA	10x10 ³ b								
TB	34x10 ³ b	11x10 ⁴ e	24x10 ⁸ f						

LEGEND

Authors:

- a Viraspharam and Karnock (1975)
 b Ziebell et al. (1975)
 c Karikari (1975)
 d Otis et al. (1975)
 e Bouma et al. (1974b)
 f Heydoff et al. (1974)
 g Remess and Pettry (1976)
 h Walker et al. (1973b)
 i Bouma et al. (1975)
 m Bouma et al. (1975)
 n Walker et al. (1973a)
 p Remess and Pettry (1975)

Symbols:

pH	pH	total suspended solids	mg/l	Fe	soluble iron content	mg/l
TSS	800	biological oxygen demand	mg/l	Cl ₂	soluble chloride content	mg/l
BOD	TOD	chemical oxygen demand	mg/l	FS	fecal streptococci	no/100 ml
COO	PO ₄	total organic carbon	mg/l	FC	fecal coliforms	no/100 ml
NO ₃	Org. P	phosphate phosphorus	mg/l	TC	total coliforms	no/100 ml
Tot. P	Tot. P	total phosphorus	mg/l	PA	PS. aeruginosa	no/100 ml
NH ₄	NH ₄	ammonia nitrogen	mg/l	TB	total bacteria	no/100 ml
NO ₃	Org. N	nitrate nitrogen	mg/l			
Tot. N	Tot. N	total nitrogen	mg/l			

100 ml. The septic bed soil may remove bacteria in varying degrees from almost all of the harmful bacteria to almost none. The input of water from the septic tank is more uniform as average per capita water use is 150 to 190 l (40-50 gal) per day (Bennett et al., 1975; Witt et al.,⁵ 1975; Jones, 1975; Otis et al.,⁵ 1975).

A tentative assessment of nitrogen and phosphorus losses from septic systems in various North American studies is presented in Table 2.5. The compilation of these figures is explained in the appendices. The large variation of these estimates of nitrogen and phosphorus losses and levels, as shown in Table 2.4, precludes an evaluation of the efficiency of septic bed soils as processors of nitrogen and phosphorus. More data is required for reliable estimates of these losses.

Other nutrient inputs to, and outputs from, the septic system of a comparatively minor nature are precipitation, volatilization from the septic tank, surface runoff and soil ion release from bedrock. These inputs and outputs are difficult to measure on a small scale and are evaluated by comparing the septic bed soil with adjacent control sites. Thus, in this study, these components were inferred rather than measured.

⁵The figure is not given but can be computed from information given at 47.9 gal/d.

TABLE 2.5 Nitrogen and Phosphorus Losses from Septic Systems in North America

kg/d.	United States			Ontario	
	Porcella and Bishop, 1975	Walker et al., 1973b	Ryden et al., 1973	From Table 2.4	Walker et al., 1973b
Phosphorus	69,760 ¹	---	14,795 to 3,804 ²	8,454 ²	396.3 ² ---
Nitrogen	345,600 ¹	718,902 ³	---	2,114 ²	990.75 ² 39,450 ³

¹Based on estimates of waste input to septic tank.

²Based on estimates of waste output from septic tank.

³Based on measurements of effluent levels in septic-bed soils and cottage water use.

2.3 The Effect of Land Waste Disposal on the Natural Equilibrium of Soil Systems

Natural soil systems tend towards an equilibrium between inputs to the soil and nutrient losses from the soil (Ryden, 1973; Webber, 1973; Bolt and Bruggenwert, 1976; Zwimmerman and de Haan, 1973). In natural systems, nutrients in an available form are usually limiting to biological activity, resulting in slow rates of soil transformation. As a result of these slow processes, natural soils must be studied over a long period of time before such changes in soil properties can be detected (Martel and Paul, 1974). Many natural soil systems extend over large areas and require sampling from many spatial locations, as done by Jenny (1961) with organic nitrogen in North American prairie soils. Land waste disposal increases the nutrient supply and biological activity, thus accelerating soil change, and such changes can be detected over shorter periods of time. Impacts of land waste disposal are also spatially restricted, as septic pollution can be detected over a few acres only (Beatty and Bouma, 1973). These rapid changes over small areas render the intensive study of soil pollution a relatively easier small-scale exercise.

2.4 Conclusions

Reported information, combined with information from this study, can be used to draw some conclusions about the adequacy of the assumptions on which this study is based as well as, by extension, the traditional assumptions. To test the first assumption of this study, i.e., the septic system is a heterogeneous system which cannot be

represented by average levels, levels and fluxes of pollutants measured in this study will be compared with reported levels. As part of the second assumption, it was suggested that significant nitrogen and phosphorus losses occur from septic systems operating under favourable conditions. This cannot be evaluated from existing information as losses vary and average North American losses cannot be estimated. This information supports the assumption that more data, especially estimates of nitrogen and phosphorus fluxes, are needed. In addition, the reported techniques for altering denitrification, phosphate removal and soil moisture flow will be used to suggest ways of reducing the losses of nitrogen and phosphorus detected in this study. Existing information also suggests that land waste disposal accelerates soil processes, changing the solid components of the soil, which is the third assumption made for this study.

The acceleration of soil processes, the increased nutrient and water fluxes and the physical alterations undertaken to manipulate soil processes result in an artificial soil. The properties of this artificial soil and the way in which it functions are in many ways different from agricultural and natural soils. This unique type of soil body requires a new approach to its measurement and study, as described in the following chapter.

CHAPTER III

METHODOLOGY

3.1. Research Site

The site of the investigation is at a cottage and adjacent field and wooded area located on Whitestone Lake, 40 km northeast of Parry Sound, Ontario (Fig. 3.1). Criteria for the choice of this research site included accessibility, a septic bed soil which would be expected to function reasonably well in terms of processing effluent, and a clear delineation of source, sink and background areas. The site is geomorphically simple with relatively uniform sediment or parent materials. Slopes are gentle and uniform in direction.

The site is divided into three distinct soil environments: the septic bed, the field soil and the forest soil (Fig. 3.2). The pollutant source is the septic tank effluent and its tile distribution system. The effluent sink includes the septic bed soil and the margin of the septic bed where effluent leaks out as a contaminant of the adjacent field soil. The background areas include the unaffected field soil and the forest soil.

The septic bed soil is an artificial or humanly-altered soil which could be expected to change over time due to large effluent fluxes. The field soil also represents an altered soil. This area was logged about 100 years ago, then ploughed and cropped. The field

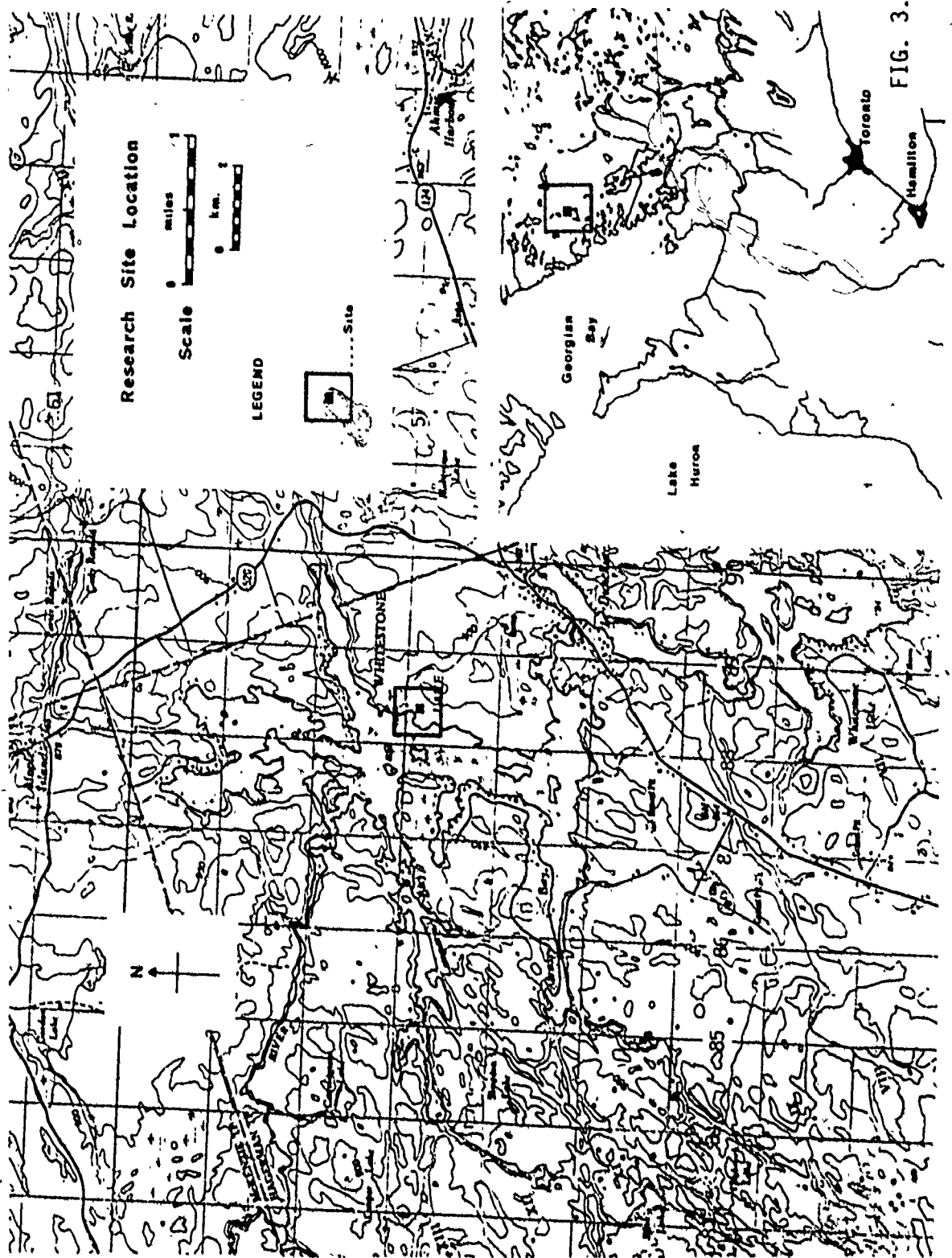


FIG. 3.1

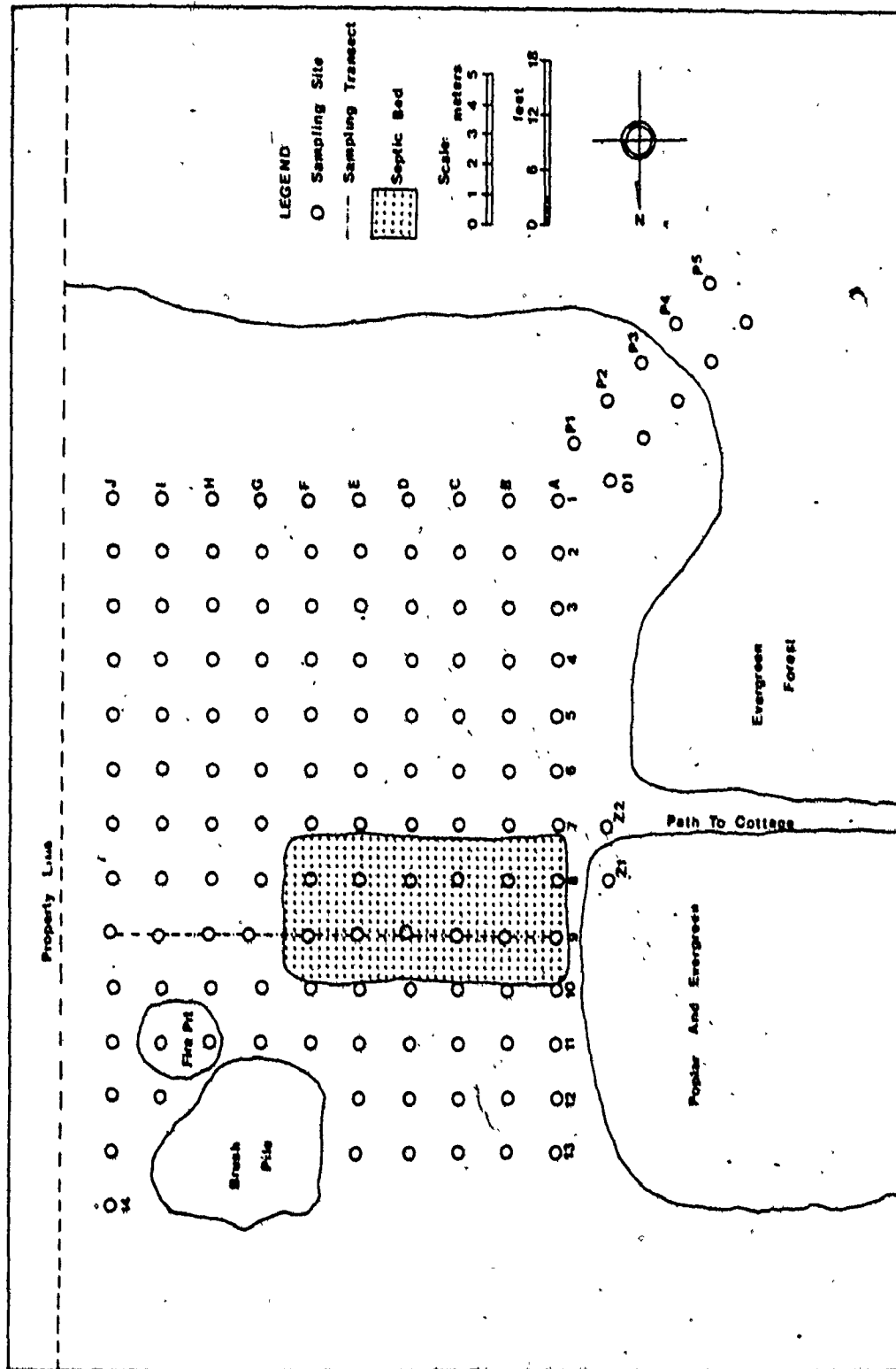


FIG. 3.2 Research site area

soil has not been ploughed for several decades. Hay is cut annually in August. The forest soil represents the undisturbed original natural soil before alteration by settlers. The forest fringe around the field is predominantly coniferous comprising spruce, balsam and hemlock, and further in, deciduous species of poplar, beech and hard maple.

Although the soil at this location is not mapped in the Soil Survey of Parry Sound District (1962), it is apparent that it is closely related to the Magnetewan Series. The parent materials of this Orthic Acid Brown Wooded soil are non-calcareous silt loams deposited in old glacial lakes. The site is near the edge of a former glacial lake, as indicated by nearby sandy glacial lake beach and stream deposits. Due to erosion and variations in bedrock topography, soil thickness varies. In the site, it exceeds 1 m, except in the north-west corner where the Precambrian bedrock extends to within 60 cm of the soil surface. Precambrian bedrock underlies all the soil and occasionally interrupts the undulating topography with small rock islands in the fields.

Figures 3.4 to 3.7 show the septic bed, field and forest soil environments and the soil sampling locations, including the core sampling sites used for spatial mapping and the deep soil inspection pits used for continuous monitoring. Figure 3.3 indicates the location and orientation of the photographs.

3.2/ Measurement of Soil Properties

The research design for the four sampling surveys undertaken is summarized in Table 3.1. Of the four surveys, three are designed to

FIG. 3.3 Locations and orientations of site area photographs



FIG. 3.4 Site area from the northeast

The septic bed soil in the centre of the picture is raised several inches above the rest of the field. The light patches mark spatial sampling locations.



FIG. 3.5 Site area from the north

View taken south from the deep soil inspection pit no. 4 to soil pit no. 1, showing the fringe of the coniferous forest and pits no. 2 and no. 3 located near the piles of earth. Pit no. 4 is in the foreground and pit no. 2 is in the middle of the picture.

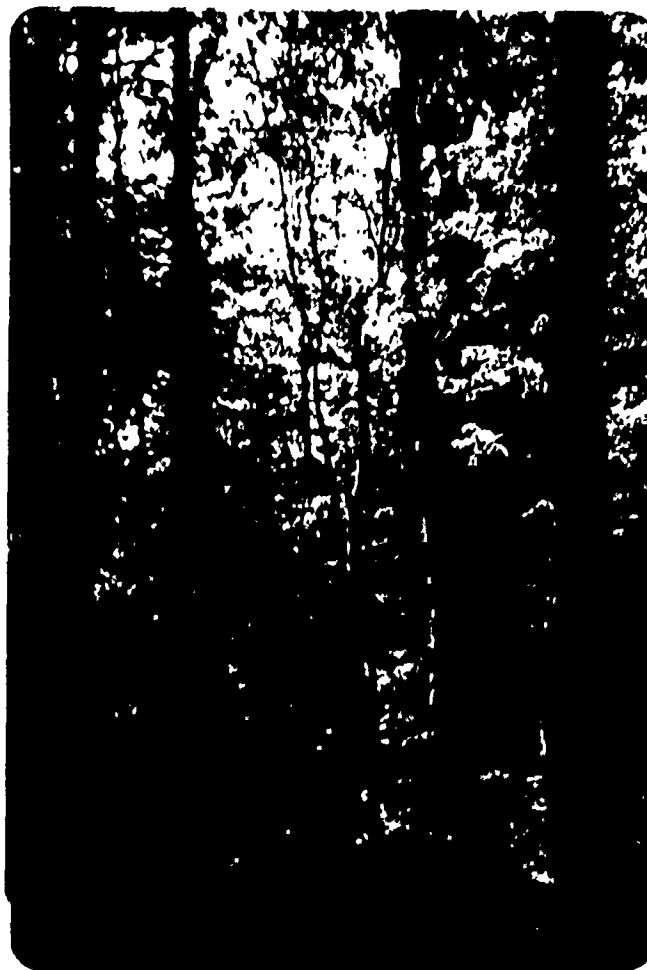


FIG. 3.6 Field and septic bed from the southeast

In the top right corner, the excavated sand pit is visible. This beach deposit marks the glacial lake shoreline.

23

39



6

FIG. 3.7 Wooded area from the west

A view of the deciduous forest near the deep soil inspection pit no. 1, looking east towards the swamp.

TABLE 3.1 Sampling Design

	<u>Stable properties</u>		<u>Transient properties</u>	
Survey	1. average levels	2. spatial variation	3. response to starting septic system	4. response to stopping septic system
Purpose	background levels and equilibrium levels of soils	equilibrium levels, spatial variation and response to effluent flux	non-equilibrium response to commencement of effluent input	non-equilibrium response to cessation of effluent input
Space-time sample	restricted in space and time	spatially distributed, time restricted	time distributed, spatially restricted	13 locations sampled twice
Sampling technique	four deep soil inspection pits	spatial analysis of area soils using grid pattern with 2 m intervals and 137 sites	continuous monitoring of five sites	transect of 13 sites at 2 m intervals through septic bed and field
Type of sample and site	deep soil inspection pit, disturbed and undisturbed samples	coring sites, disturbed samples	coring sites, disturbed samples	coring sites, disturbed samples
Method of interpretation	graphs, tables	spatial mapping, factor analysis	graphs for C horizon, factor analysis, canonical correlation, multiple R and simple r	spatial mapping

measure transient soil properties which change rapidly and over short distances. The remaining survey was designed to measure stable soil properties which change very slowly and are more uniform from place to place. Stable properties are associated with the soil solids whereas transient properties describe soil solution and soil gaseous components.

Measurements of stable properties from the forest soil indicate initial conditions before alteration. Similar measurements from the field soil suggest some of the original properties of the septic bed soil before its alteration during construction. In addition, many of the stable properties of the soil solids influence responses of the soil solution and soil gases to nutrient and moisture inputs. For example, the surface area of the soil, a stable property, influences the soil moisture content, a transient property (Banin and Amiel, 1970; Varazashvili et al., 1976). Thus variations in soil solid properties help to explain differences in soil solution and soil gases in forest, field and septic bed soils.

To study the average levels of stable properties, deep soil inspection pits were used. From these soil pits, the whole profile was also sampled. As previously mentioned, these soils consist of permeable silt loams overlying impermeable clays, which are difficult to penetrate with the core sampler when measuring transient properties. Thus transient properties were also measured when the clay horizons were made accessible during the survey of stable properties. At this depth, transient properties are less variable, making it easier to understand soil processes with limited information. In order that the transient properties would be in equilibrium with effluent input, the soil pits

were sampled three weeks after the septic bed had been in operation.

Referring to Table 3.1, the purpose of surveying stable properties and average levels was primarily to determine background levels of soil solid properties, and, as it was convenient, equilibrium levels of gaseous and liquid properties. The survey is restricted in space and time as it was done once, at four locations, representing the septic bed, field and forest soils and the field-septic bed interface (Fig. 3.8). Soil profiles from the pit face describe the arrangement and organization of soil materials. Disturbed and undisturbed samples from the soil inspection pits were analyzed in the laboratory for organic matter content, texture or particle size distribution, the physical properties of bulk density, particle density and proportion of solids, the soil adsorption properties of cation exchange capacity, hygroscopic moisture, specific surface area, characteristic drying curve and saturated hydraulic conductivity.

As transient soil properties vary in space and time, separate surveys were used to measure spatial and temporal variation. It was assumed, and the literature suggests, that soil properties approach an equilibrium after several weeks of continuous operation of the septic system. It was also assumed that responses of the septic bed soil to initiation and cessation of effluent input would produce the largest variations in soil properties over time. The spatial survey was designed to measure the steady state of the system while the temporal surveys assessed the responses of the septic system to initiation and cessation of effluent input (Table 3.1). The purpose of the spatial survey was to discern patterns of spatial variation.

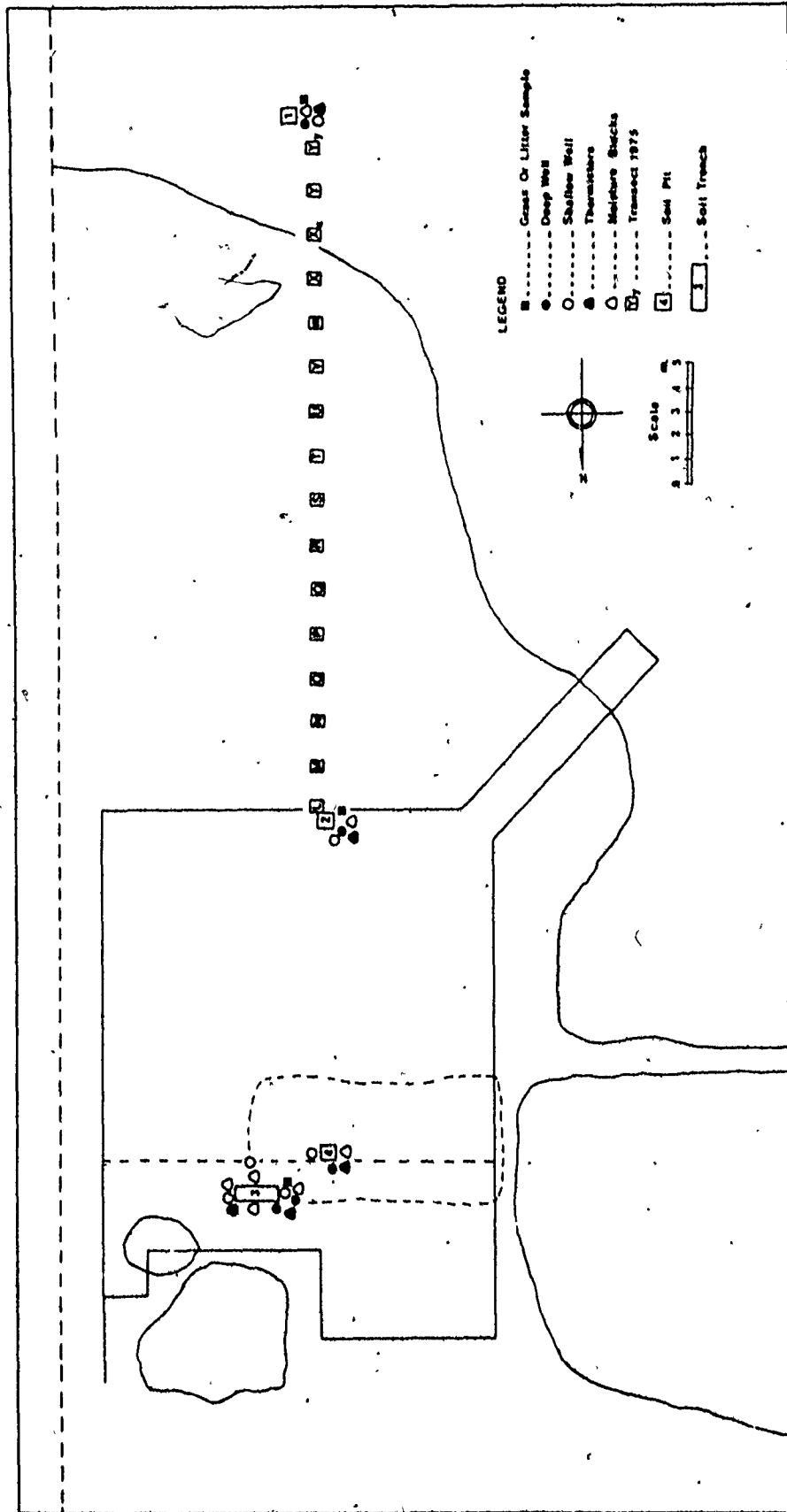



FIG. 3.8 Locations of deep soil inspection pits

within the forest, field and septic bed soils. Also mapped were variations in the intensity of nitrate and effluent decomposition in the septic bed, and the influence of their losses on the adjacent field soil. This required a spatially-distributed sample which could be restricted in time to a single sample event. To accomplish this, 137 sites, distributed over the site area in a grid pattern 2 meters apart (Fig. 3.2), were sampled at 7 cm intervals down to 60 cm or until the hard clay prevented core sampling. The septic bed and its area of influence were not sampled until after three weeks of continuous use to ensure that the soil solution and gases had approached equilibrium with the effluent.

To assess temporal response, two sampling designs were used. The response of the septic bed soil to initiation of effluent input was evaluated by continuous monitoring of five sites adjacent to the soil pits (Fig. 3.8). This provided measurements which are distributed in time and spatially restricted. Core samples were taken and, in addition, solution wells at 100 cm depth were sampled. Sampling bi-weekly for a total of seven sampling events permitted analysis of samples between sampling events with sufficient sensitivity to detect trends in soil responses. To study soil changes after use of the system ceased, a transect through the septic bed and field was sampled 13 and 26 days after use of the bed had ceased.

Previous analysis (Campbell, 1974) indicated the most useful measurements of the soil solution and gases and how to interpret them. These measurements include the pollutant indicators--oxygen diffusion rate, nitrate and chloride levels; pollutant-sensitive properties such



as soil moisture, pH and sodium ion levels, and pollutant-insensitive properties such as calcium ion levels which act as a background or reference. Also measured were other soil ions, soil temperature, rainfall, evapotranspiration and cottage water consumption. Soil moisture content was determined gravimetrically from soil cores and calibrated gypsum moisture blocks. Soil temperature was measured with calibrated thermistors. Using specific ion electrodes and core samples, sodium, calcium, chloride, potassium, nitrate, ammonia and phosphate soil ions were measured. Oxygen status was evaluated with in situ measurement of oxygen diffusion rate. Using small lysimeters and a rain gauge, evapotranspiration and rainfall were measured continuously during the sample period as was cottage water consumption.

3.3 Methods of Interpreting Soil Properties

The interpretations are largely comparisons of information from this study with the previously existing information with the view to a better explanation of what occurs in the field, forest and septic bed soils. In this study, three methods of interpreting measured soil properties are used. Levels of soil properties are grouped, mapped and tabulated for direct interpretation. Statistical multivariate techniques are also used to test known relationships amongst these measurements. From the measured soil properties, estimates of water and nutrient fluxes are calculated and these provide a third way to compare the results of this study with what is known about soil pollution and losses of nitrogen and phosphorus from both affected and natural soils. The first approach involving direct interpretation needs no explanation. A brief

discussion of the employed statistical techniques also follows. Since discussion of the methods of calculating water and nutrient fluxes is more lengthy, it is included in a subsequent and separate section.

Multivariate techniques discern relationships between individual properties and groups of properties. This process could be termed classification. Multivariate analysis was used to classify and group soil ions.

A large number of soil ions were measured, only a few of which have been studied in septic bed soils. In natural soil systems, some of the other ions, for example, calcium, have been studied extensively. For many of the ions measured, little information is available to explain variation in these ionic levels in septic bed soils. Groupings and relationships identified by multivariate analysis permit extrapolation from known behaviour of studied ion species to little studied ion species. For example, the chloride and nitrate leaching processes in septic beds are known. Very little is known about phosphate ion leaching in septic beds. Multivariate analysis could suggest whether phosphate is leached by effluent in a similar fashion to nitrate and chloride. The responses of sodium and calcium to soil processes is known for many natural soil systems. By grouping or not grouping these ions with nitrate and chloride, for example, inferences about their response to effluent flow can be made.

Multivariate analysis can also be used to test or corroborate some of the results and conclusions derived by other analytical approaches. For example, sodium ions, which tend to be grouped with known pollutants such as nitrate and ammonia, show, as does nitrate,

6 large calculated losses from the septic bed in excess of the reference sites.

The multivariate techniques used include multiple regression, canonical correlation and common factoring with oblique rotation. The technique of multiple regression was then used to identify relationships between soil properties and interpret them in the light of existing information on soil properties and processes. Prediction of the levels of properties from regression equations was of secondary concern. Canonical correlation was used only once, as it is uniquely suited to interpreting soil responses such as the initiation of the use of the septic system. Canonical correlation explains one set of properties in terms of another, and hence it could be assumed that soil depth and the time elapsed from starting effluent flow into the septic bed would affect the levels of transient soil properties. This hypothesis can be tested with canonical correlation analysis by using measurements of soil depth and time elapsed to explain the levels of soil properties. In factor analysis, soil properties with common variation are grouped into a smaller number of discrete independent factors. Obliquely rotating the factor solution allows the factors to be correlated or interdependent. Since soil properties are frequently interdependent, this provides a more realistic solution in most cases.

For the limited number of measurements of stable soil properties, direct interpretation was used, as indicated in Table 3.1. With the large numbers of measurements of transient soil properties, direct interpretation was mostly limited to spatial mapping. More appropriate to this quantity of information, multivariate analysis of transient



properties was used extensively.

3.4 Methods of Calculating Mass Balance

Mass balance calculations provide additional information for evaluating the assumptions on which this study is based (Chapter I, p. 8). It was assumed that nutrient losses of nitrogen and phosphorus ions could be significant, even under favourable conditions, and this could only be detected by estimates of nutrient flux; hence, the primary purpose of these calculations was to estimate nutrient losses from the septic bed and compare them with similar losses from the reference soils. In so doing, information was also gained about the variation in fluxes and changes in soil solid components in the septic bed. This would permit further evaluation of the first and third assumptions--the septic bed soil is a heterogeneous system which cannot be represented by average levels, and its solid components may be changing. Comparing the accuracy of concentration measurements and flux estimates as indicators of septic pollution would test the second assumption.

Mass balance calculations were used to provide estimates of (i) adsorbed cations, (ii) fluxes of soil water and soil ions, (iii) denitrification and (iv) plant uptake of soil ions. The mathematical techniques for calculating these estimates from other soil properties are discussed first. These estimates are calculated in three programs using measurements of soil properties made from the various sampling surveys of this study, and this is discussed next. Translation of mathematical formulas into Fortran is described in the

appendices.

Ion exchange was calculated, following Marshall (1964) and Beek and Frissel (1973) as follows:

Part I Calculation of the partial sum of adsorbed ions from ions in solution and cation exchange capacity when all the adsorbed ions are not measured.

$$\sum_{i=1}^n \frac{C}{iA} = \text{CEC} \quad (1)$$

where $\frac{C}{iA}$ is the concentration of an adsorbed soil ion

CEC is the cation exchange capacity

For a monovalent-monovalent system

$$\frac{\text{cation 1 adsorbed}}{\text{cation 2 adsorbed}} = K_m \frac{\text{cation 1 in solution}}{\text{cation 2 in solution}} \quad (2)$$

for sodium (Na) for example, the equation becomes:

$$\frac{C_{KA}}{C_{KS}} = K_m \frac{C_{NaA}}{C_{NaS}} \quad (2a)$$

For a monovalent-divalent system

$$\frac{\text{divalent cation in solution}}{\text{divalent cation adsorbed}} = K_d \left\{ \frac{\text{monovalent cation in solution}}{\text{monovalent cation adsorbed}} \right\}^2 \quad (3)$$

for calcium (Ca) for example, the equation becomes

$$\frac{C_{CaA}}{C_{CaS}} = K_d \left\{ \frac{C_{KA}}{C_{KS}} \right\}^2 \quad (3a)$$

From equation (1)

$$C_{NaA} + C_{KA} + C_{CaA} = CEC - C_{XA} \quad (4)$$

where C_{XA} is the sum of all those ions not measured and absorbed on the exchange complex.

From (2a) and (4), C_{KA} and C_{NaA} can be expressed as a function of C_{KS} and C_{NaS} . Hence

$$(C_{NaA} + C_{KA} + C_{CaA}) = f(C_{NaS}, C_{KS}, C_{CaS}, CEC) \quad (5)$$

Assuming these relationships are independent and additive, then,

$$C_{NaA} + C_{KA} + C_{CaA} = a + b_1 C_{NaS} + b_2 C_{KS} + b_3 C_{CaS} + b_4 CEC \quad (6)$$

The coefficients can then be obtained by multiple regression.

Part II Calculation of the concentration of individual absorbed ions

When the coefficients have been obtained, it will be possible to predict the sum of absorbed cations of Ca, Na and K from the concentration of these ions in solutions, C_{CaS} , C_{NaS} and C_{KS} , and cation exchange capacity, CEC. This calculated sum is termed Z as shown in (7).

$$C_{NaA} + C_{KA} + C_{CaA} = Z \quad (7)$$

From (2a)

$$\frac{C_{NaA}}{C_{KA}} = K_m^{-1} \frac{C_{NaS}}{C_{KS}} \quad (2b)$$

$$C_{NaA} = \frac{C_{KA} C_{NaS}}{K_m C_{KS}} \quad (2c)$$

Rearranging (3a)

$$C_{CaA} = C_{CaS} K_d \left(\frac{C_{KA}}{C_{KS}} \right)^2 \quad (3b)$$

Substituting (3b) and (2c) into (7)

$$\frac{C_{KA} C_{NaS}}{K_m C_{KS}} + C_{KA} + C_{CaS} K_d \left(\frac{C_{KA}}{C_{KS}} \right)^2 = Z \quad (8)$$

$$\frac{C_{KA} C_{NaS}}{K_m C_{KS}} + C_{KA} + \frac{K_d C_{CaS} C_{KA}^2}{C_{KS}^2} = Z \quad (9)$$

$$C_{KA}^2 \left\{ \frac{K_d C_{CaS}}{C_{KS}^2} \right\} + C_{KA} \left\{ 1 + \frac{C_{NaS}}{K_m C_{KS}} \right\} - Z = 0 \quad (10)$$

which is of the form

$$a_1 C_{KA}^2 + b_1 C_{KA} - C = 0 \quad (11)$$

This is a quadratic equation where

$$C_{Ka} = \frac{-1 - \frac{C_{NaS}}{K_m C_{KS}} \pm \sqrt{\left\{ 1 + \frac{C_{NaS}}{C_{KS} K_m} \right\}^2 + 4 \left\{ \frac{K_d C_{CaS}}{C_{KS}^2} \right\} (-Z)}}{2 \left\{ \frac{K_d C_{CaS}}{C_{KS}^2} \right\}} \quad (12)$$

Since soil ion concentrations, i.e., C_{KA} , must be positive, the positive root of (12) is used.

When C_{KA} has been calculated from (12) and since K_m , C_{NaS} and C_{KS} are known, equation (2c) can be solved for C_{NaA} .

C_{CaA} can be calculated from equation (3b) or (7). Equation (7) was chosen as the simpler equation. C_{CaA} can be calculated from equation (7) because Z is obtained from multiple regression.

An empirical approach is also possible:

From (11) and (5), it can be seen that

$$C_{KA} = f(C_{KS}, C_{CaS}, C_{NaS}, K_d, K_m, CEC) \quad (13)$$

Equation (13) can be evaluated using multiple regression. Using data where all the values are known, (11) and (13) can be solved and compared for their predictive ability.

Water and nutrient fluxes were calculated from Taylor and Ashcroft (1972) and Beek and Frissel (1973) as follows:
soil ion transport is calculated as the product of the ions in solution and the transport of soil water

$$J_k = J_w C_{ks} \quad (14)$$

These estimates of solute flux assume that solute and water flow at the same rate. Dispersion and diffusion of soil ions in water cause differences in flow rates. It is known that water in large pores moves faster than water in small pores. Similarly, water close to soil surfaces moves more slowly than water farther from soil surfaces. Cations are attracted to soil surfaces and thus become concentrated in the more slowly-moving soil solution in small pores close to soil surfaces. Anions are repelled from soil surfaces and concentrated in the faster-moving soil solution in large pores away from soil surfaces.

This process is termed ion dispersion and causes cations to move more slowly and anions faster than the average water flux. As it ignores dispersion, equation (14) overestimates cation fluxes and underestimates anion fluxes. Diffusion occurs as ions flow from areas of high concentration to areas of low concentration and is usually very small. The effect of dispersion and diffusion decreases as flow velocity increases and Bolt and Bruggenwert (1976) report that, for moderate and low flux rates, these are usually less than 10% of the total solute flux. In the septic bed soil with large fluxes, this means ammonia losses are probably slightly overestimated while phosphate and nitrate losses are slightly underestimated.

Soil moisture transport is calculated as the product of the hydraulic conductivity and hydraulic potential gradient as explained in Chapter II.

$$J_w = -K_w \frac{d\psi_h}{ds} \quad (15)$$

The hydraulic gradient is computed differently for vertical and lateral fluxes. By computing the vertical hydraulic gradient over a distance of 1 cm, it is equal to the sum of the matric potential gradient, ψ_w , and gravity potential gradient, ψ_z , which is 1.

$$\psi_h = \psi_w + \psi_z \quad (16)$$

$$\text{Assuming } \psi_w = \psi_m \quad (17)$$

$$\psi_h = \psi_m + \psi_z \quad (18)$$

combining (15) and (18) yields (19)

$$J_w = -K_w \frac{d\psi_m}{ds} + \frac{d\psi_z}{ds} \quad (19)$$

Computing ψ_z over 1 cm,

$$\frac{d\psi_z}{ds} = 1$$

(19) reduces to (20)

$$J_w = -K_w \frac{d\psi_m}{ds} + 1 \quad (20)$$

For lateral moisture fluxes, the gravitational potential is zero and equation (15) simplifies to (21):

$$J_{wL} = -K_w \frac{d\psi_m}{ds} \quad (21)$$

Since the flux is lateral, $\psi_z = 0$ and $\psi_h = \psi_m$.

$$J_{wL} = -K_w \frac{d\psi_m}{ds} \quad (22)$$

Matric potential is computed from the soil moisture content and the regression equation between the log of soil moisture and the log of matric potential

$$\psi_m = 10 \left\{ ax (\log \theta_v + b) \right\} \quad (23)$$

The remaining unexplained term in equation (15) is conductivity.

Saturated conductivity is measured. Unsaturated conductivity is calculated as follows (G. Campbell, 1974):

$$K_{UNSAT} = K_{SAT} \left\{ \frac{\theta_v}{\theta_{vSAT}} \right\}^{(2b+3)}$$

Denitrification and plant uptake are calculated as follows:

denitrification is estimated from the Cl_2/NO_3 ratios in the reference areas as follows:

$$R = \frac{C_{\text{Cl}_2}}{C_{\text{NO}_3}} \quad (25)$$

$$D = D_{\text{NO}_3} - \left\{ \frac{C_{\text{Cl}_2}}{R} \right\} \quad (26)$$

estimates of plant uptake of soil ions are calculated from evapo-transpiration and the concentration of soil ions in solution.

For example, for potassium uptake

$$J_{\text{KE}} = \frac{C_{\text{K}}}{1000} E_t \quad (27)$$

Many conversions from one form of measurement to another were necessary and these are explained in the appendices.

The sources of measurements used to estimate these components, and the programs in which they were used, are indicated in Table 3.2. The Fortran programs BALAN1 and BALAN3 were used to calculate fluxes when the septic bed soil was in equilibrium with effluent input. The program BALAN2 calculates these fluxes as the septic bed begins to receive effluent. Fluxes in five sites located in the forest, field and septic bed soils were computed by the programs BALAN1 and BALAN2 using measurements from the survey of stable soil properties, and the continuous survey of transient properties respectively. Detailed lateral fluxes and denitrification around the edge of the septic bed were computed with a third program BALAN3 using data from the spatial survey. The translation of the mathematical equations into Fortran

TABLE 3.2 Soil Properties and Surveys Used for Mass Balance Calculations

<u>Program where component used</u>			<u>Mass balance component</u>	<u>Survey source</u>	<u>Soil property</u>
BALAN1	BALAN2	BALAN3			
X	X	X	(C) volumetric conversions	average levels	particle density, bulk density
X	X	--	(F) prediction of adsorbed cations	average levels	adsorbed cations, cations in solution
lateral vertical	lateral vertical	lateral	(F) water and ionic fluxes	average levels	characteristic curve, saturated conductivity
X	X	--	(M) evapotranspiration and ionic uptake	response to starting septic system	lysimeter measurements
X	X	X	(F) denitrification	average levels and response to starting septic system	Cl ₂ /NO ₃ ratios

LEGEND: M = measured flux.

F = computed flux where the flux is computed from a mathematical relationship or function between measured values

C = conversion of a value from one form of measurement to another

and the organization of these three programs are discussed in appendix C pages 151 to 198.

CHAPTER IV

OBSERVED SOIL PROPERTIES

4.1 Stable Soil Properties

Profiles of forest, field and septic bed soils are shown in Figs. 4.1 and 4.2. The profile descriptions are included in Appendix B. The soils consist of permeable silt loams and sandy loams in the A and B horizons or solum overlying impermeable clays in the C horizon. The loams permit percolation of moisture through the solum, whereas the clays act as a barrier. If the soil receives large inputs of moisture, lateral soil moisture flow occurs at the top of the clay horizons.

The septic bed profiles indicate recent disturbance and alteration, including additions of sand in the A and B horizons and gravel in the C horizon (IIIc). The latter horizon appears black due to organic deposits on the surfaces of gravel fragments. Overlying this, the Ap horizon contains masses of red and white clay from the original IIc1 and IIc2 horizons. The septic bed soil lacks the firm platy-structured white clay horizon IIc1 present in the field and forest soils.

There are small differences between field and forest soils, probably due to a higher intensity of leaching in the field soil than in the forest soil. In the forest soil, the A1 and Ae horizon is about 6 cm thick. The IIc1 horizon is continuous, starts at 23 cm

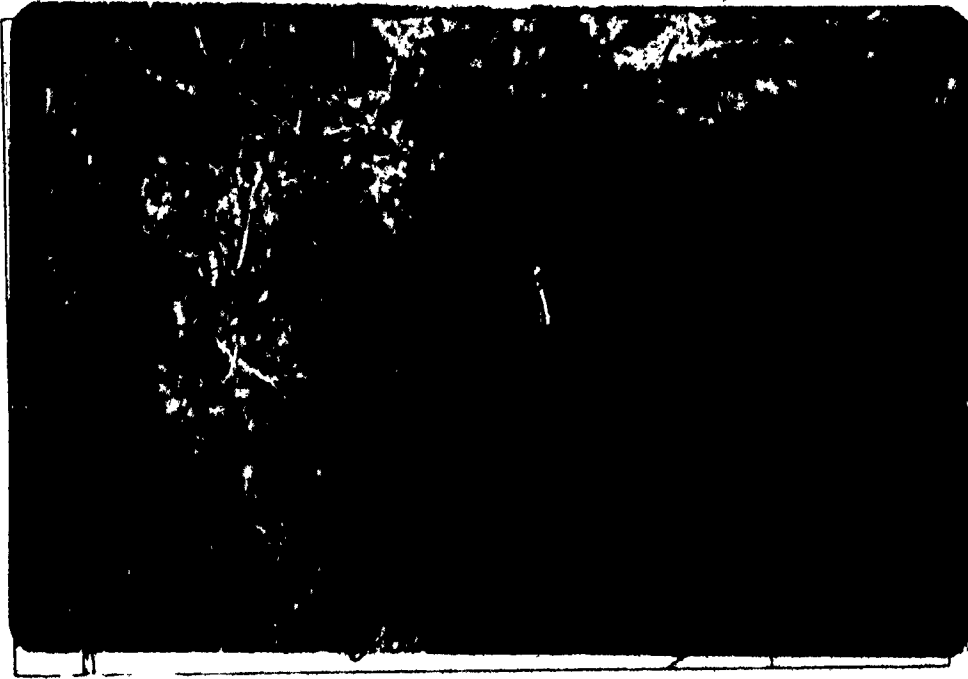


Profile of field soil

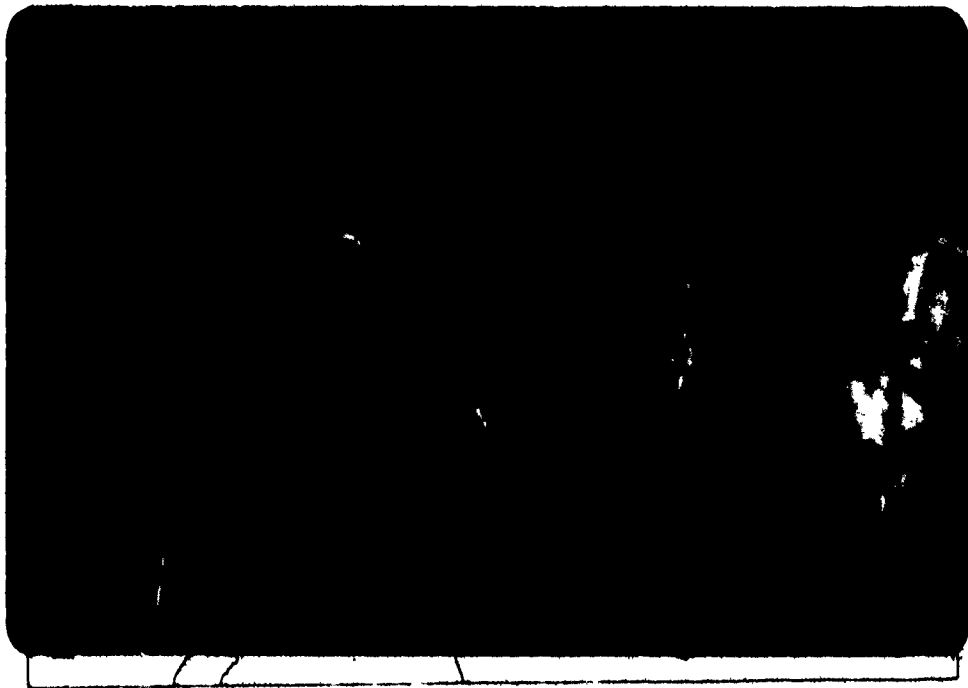


Profile of forest soil

FIG. 4.1 Photographs and overlays of soil profiles in the reference soil's



Profile of edge of septic bed



Profile of centre of septic bed

FIG. 4.2 Photographs and overlays of soil profiles in the septic bed soil

depth and is 20 cm thick. In the field soil, the A1 and Ae horizons are about 2.5 cm and 15 cm thick respectively. The continuous part of the IIC1 horizon starts at 30 cm depth and is 5 to 10 cm thick. Underneath this thin IIC1 is a degraded IIC1 interspersed with inclusions of IIC2. It appears that more intensive leaching in the field soil since deforestation has reduced the depth of the A1 horizon by accelerated organic matter decomposition, increased the depth of the Ae horizon and altered portions of the IIC1 horizon along cracks until it is discontinuous and intermittent. Thus the profiles indicate that the reference soils are essentially similar to one another and dissimilar to the septic bed soil.

The stratigraphy of loams overlying clays and the effect of alteration through construction and operation of the septic bed are evident in the physical properties listed in Table 4.1 and shown in Fig. 4.3. In the clay C horizons, solids occupy about 65% of the soil volume and the bulk density is around 1.6 g/cm^3 . In the silt and sandy loam A and B horizons, solids occupy from 35 to 45% of the soil volume and the bulk density is between 0.7 and 1.5 g/cm^3 . During construction of the septic bed, lumps of clay from the IIC1 horizon have been incorporated into the A and B horizons, increasing the solids content to 50-55% and bulk density to 1.14 - 1.16 g/cm^3 . Adding sand in the tile trench lowered the solids content, bulk density and adsorptive capability, compared to the field and forest soils. The measures of the ability of soil to adsorb water, specific surface area and hygroscopic moisture content (Banin and Amiel, 1973; Varazashvili *et al.*, 1976), are 43 and 40% lower in the sand and gravel than in

TABLE 4.1 Properties of Soil Solids

Profile and horizon	Depth cm	Dry. C %	CEC meq/100g	SSA cm ² /g	MTC % by wt.	PO g/cm ³	BD g/cm ³	Solid % volume	Liquid ^a % volume	Air ^{***} % volume	Particle size distribution ^{***}		
											Sand >50 μ m	Silt 50-2 μ m	Clay <2 μ m
Pit 1 Forest	A1	3.6	7.74	56.54	2.35	2.03	0.72	33.45	8.85	55.68	36.32	44.84	18.8
	A2	2.03	1.97	55.58	2.62	2.24	0.85	37.95	11.29	50.76	45.4	38.6	16.0
	B	2.5	6.14	53.64	2.41	2.20	0.96	44.55	0.71	54.74	45.7	36.8	17.5
	1C	0.16	5.15	30.35	1.29	2.48	1.64	66.13	13.83	20.04	36.0	43.7	20.3
	11C1	0.12	0.89	44.47	1.46	2.51	--	--	--	--	--	--	--
Pit 2 Field	11C2	0.07	3.90	44.86	0.89	2.53	1.39	54.94	9.81	32.25	9.3	59.7	30.9
	A1	6.1	1.93	42.40	2.27	1.36	1.12	46.67	--	--	42.0	42.0	16.0
	A2	2.36	2.68	59.02	1.78	2.40	1.03	42.92	6.70	50.38	24.2	50.8	25.0
	B	1.43	3.64	75.06	1.14	2.44	1.05	43.03	14.57	42.40	42.9	41.9	9.8
	1C	0.91	1.70	71.09	0.40	2.45	1.01	41.22	10.12	48.64	35.1	44.9	20.0
Pit 3 Septic (East)	11C	0.11	3.74	45.08	0.98	2.62	1.69	64.50	--	--	11.0	48.5	40.5
	11C	0.07	3.01	43.93	1.10	2.65	1.69	63.77	--	--	24.1	50.7	25.3
	A1	6.25	9.31	73.15	2.73	1.77	--	--	--	--	45.9	43.7	10.4
	A2	3.40	7.62	55.39	1.77	2.43	1.14	46.91	11.78	41.31	35.3	51.8	12.9
	B1	2.25	6.79	47.94	1.70	2.44	1.13	43.31	5.26	51.43	37.7	47.3	15.1
Pit 3 Septic (West)	B2	1.4	5.57	35.35	1.44	2.50	0.97	38.80	16.33	44.87	32.7	50.6	16.7
	C	2.9	4.29	47.75	1.82	2.37	1.57	56.24	25.15	8.61	28.8	50.0	21.2
	A1	1.70	0.79	17.57	0.86	2.66	1.20	45.11	--	--	58.5	22.5	9.0
	A2	0.68	5.57	49.28	1.56	2.52	1.36	54.76	*16.83	28.41	59.0	20.6	20.4
	B1	1.42	6.24	54.83	2.09	2.23	1.16	50.66	20.95	28.39	54.6	42.7	4.7
Pit 4 Septic (Centre)	B2	1.2	4.45	42.78	2.13	2.34	0.92	39.32	34.29	25.39	64.5	32.1	3.5
	C1	0.26	5.12	30.35	1.27	2.64	1.79	67.80	32.17	0.03	35.86	44.3	19.8
	C2	0.11	4.05	40.11	1.69	2.64	1.62	61.36	21.06	17.58	12.5	52.9	34.6
	A1	4.0	7.51	38.77	1.61	2.50	0.97	38.80	25.64	35.56	45.7	35.7	17.6
	B1	2.0	8.22	47.56	2.23	2.41	0.96	39.83	28.64	37.53	26.9	49.3	23.8
Pit 4 Septic (Centre)	B2	2.1	4.95	42.28	2.05	2.45	0.86	35.10	26.31	38.59	28.1	49.6	22.2
	1C	0.14	2.15	51.57	1.26	2.57	1.43	55.64	24.83	19.53	35.1	41.1	23.2
	11C	0.26	0.43	29.82	0.65	2.72	1.66	61.03	--	0.0	30.8	8.9	10.3

^a Soil moisture content at time of sampling^{**} Air filled porosity at time of sampling^{***} % wt.

LEGEND

Org. C organic carbon
CEC cation exchange capacity
SSA specific surface area
MTC hygroscopic moisture content
PO particle density
BD bulk density

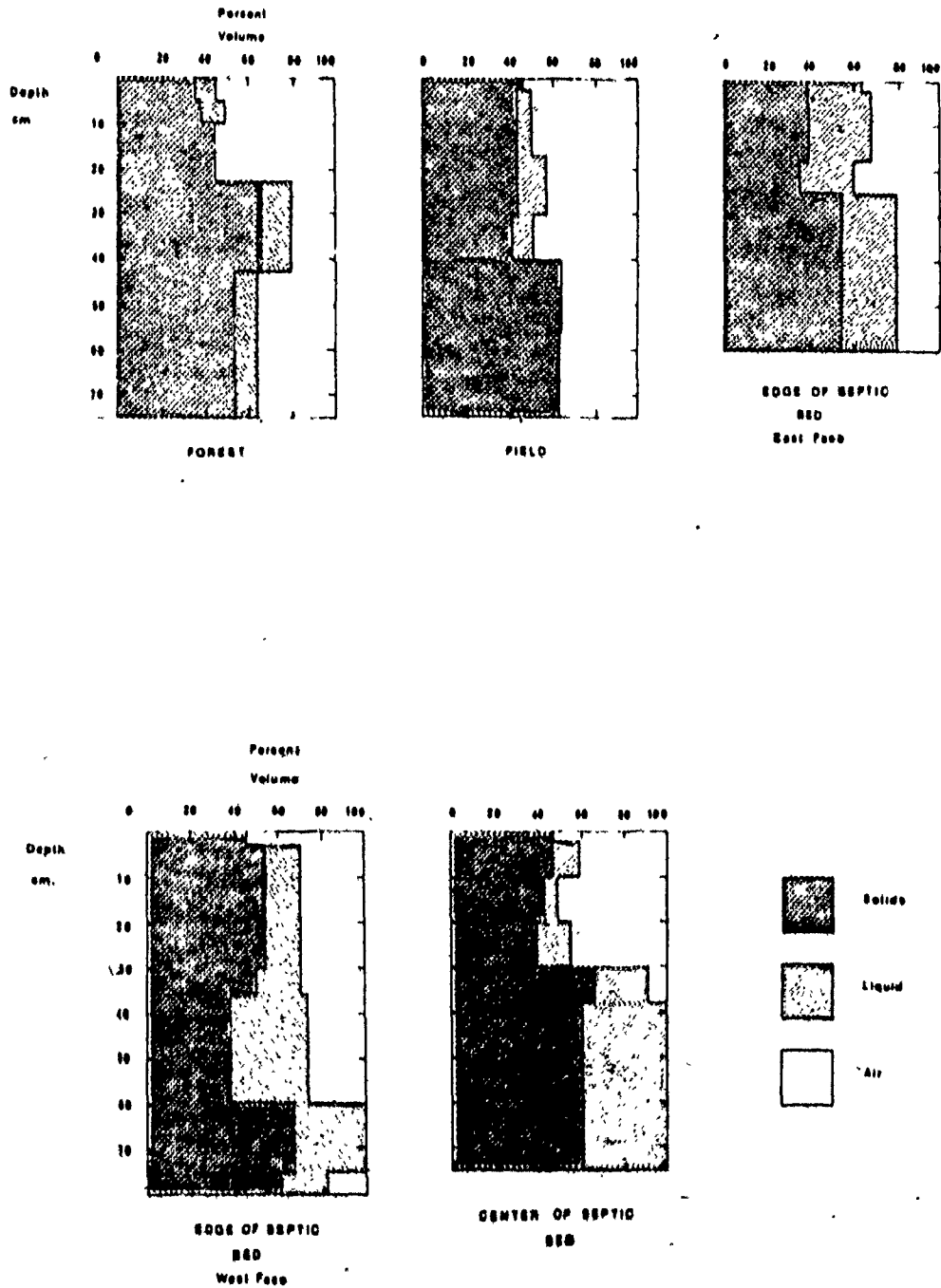


FIG. 4.3 Soil phase composition

the loam and clay.

Cation exchange capacity in the study area is less than 10 meq/100 g soil, indicating low soil fertility and adsorption capacity. Organic carbon levels vary from moderate (6.2%) to very low (0.07%). Both CEC and % C show higher values in the loam than in the clay, decreasing levels with increasing soil depth and very low values in the added sand and gravel. The first two trends reflect natural processes and the third is the result of alteration during construction of the septic bed. The decrease of cation exchange capacity and organic carbon with depth is more pronounced in the field and forest soils. This trend, typical of most natural well-drained mineral soils in temperate areas, is the result of organic carbon cycling by grasses near the soil surface. Very low levels of organic carbon (less than 1%) and a cation exchange capacity of less than 1 meq/100g in the added sand and gravel emphasize their very limited adsorptive capacity.

In summary, the stable soil properties shed some light on the validity of the assumptions on which this study is based. The first two assumptions refer to the transient or soil solution and gaseous properties. Measured stable properties suggest alteration to the septic bed will affect transient properties. The lowering of adsorptive capacity could reduce ammonia adsorption and phosphate precipitation. The destruction of the impermeable white clay layer (C1) could increase vertical effluent losses to groundwater; nevertheless, the largest effluent flows should be lateral across the top of the clay and out of the septic bed. The clay would also act as a moisture barrier, as discussed in Chapter II, enhancing denitrification. The third assumption

refers to the stability of the solid soil components in natural and septic bed soils. The measured properties of these solid components suggest that leaching is more intense in the field than in the forest. The marked alteration of the septic bed during its construction has obscured any possible changes from soil processes. Thus, although changes in the field have occurred and presumably are occurring due to human intervention, it cannot be determined yet if changes in the septic bed are occurring.

4.2 Transient Soil Properties

a. Response to Continuous Effluent Flow

The stable nature of soil solids, which respond to long term processes, contrasts with the transient nature of soil gases and the soil solution, which respond to short term processes and effluent flow in the septic bed. Measurements from the soil pits, listed in Table 4.1, can be used to characterize the soil environments of the area in a general way as follows. The A and B horizons or solum in the septic bed contain from 17 to 34% soil moisture by volume and the C horizon is saturated at 30 to 40%, as indicated in Fig. 4.3. The field and forest soils are drier, especially in the lower C horizons, containing less than 15% moisture. The wet septic bed soils, containing less air-filled porosity, are predominantly anaerobic, especially in the C horizons; whereas the drier field and forest soils are predominantly aerobic.

This information may be used to describe the septic bed soil environment during continuous effluent flow. As indicated in Fig. 4.4,

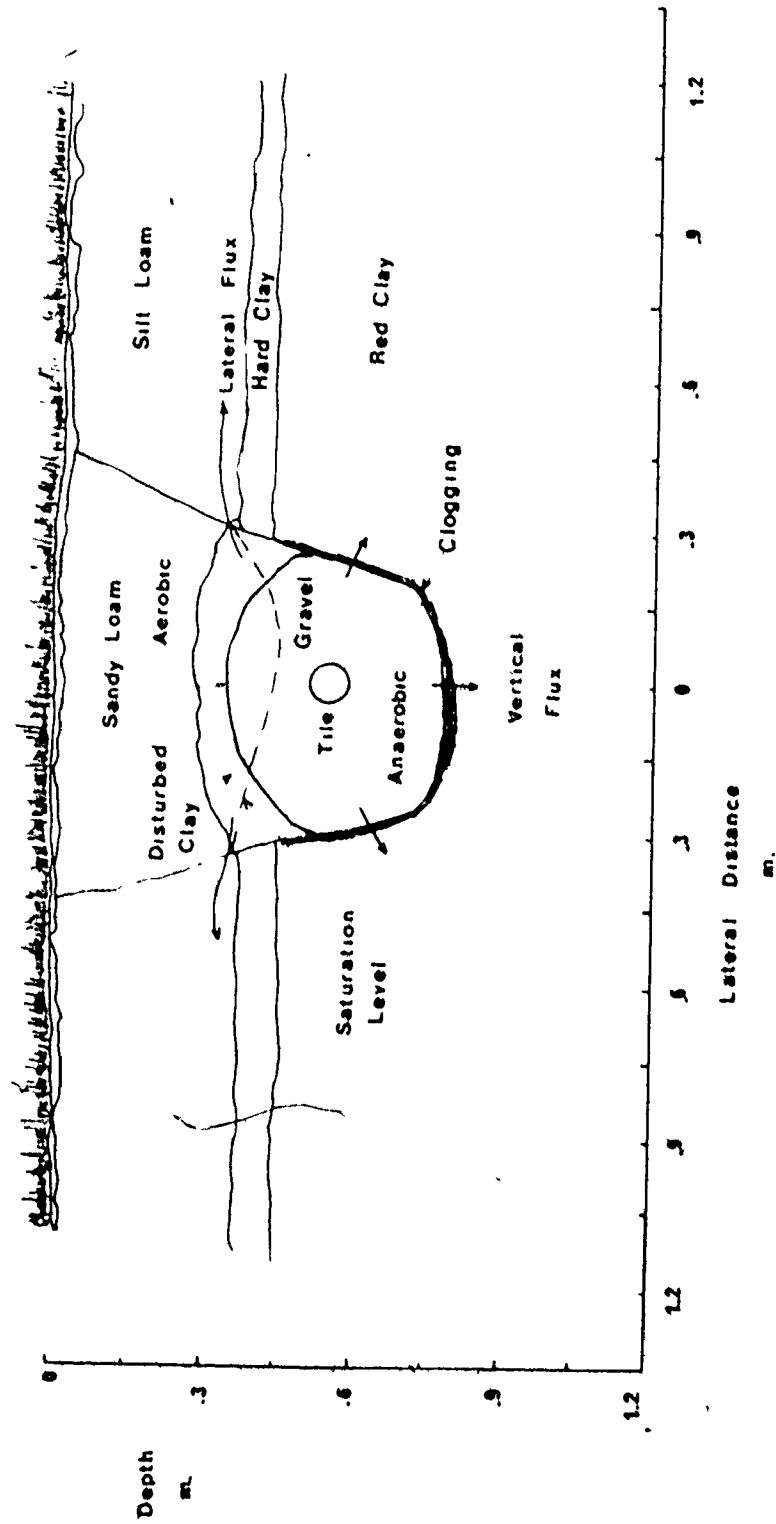


FIG. 4.4 Cross-section of the septic bed in this study showing effluent flow and soil environments

the physically disturbed and added soil materials in the tile trench are surrounded by undisturbed soil. In the tile trench, sandy loams in the A and B horizons are underlain by a thin layer of disturbed clay and gravel at the base of the trench. The sandy loams are predominantly aerobic and the gravel, being saturated with effluent, is anaerobic. Clogging occurs along the edges of the trench due to organic deposits. In the undisturbed soil, silt loam overlies hard white and red clay. Small vertical fluxes of soil solution pass through the clogged organic layer, whereas the predominant flow is lateral over the surface of the clay and out of the trench. Compared to the average system depicted in Fig. 2.1, the effluent flows more slowly and through less aerobic soil and gravel. This could limit conversion of ammonia to nitrate, resulting in significant ammonia losses. Denitrification would be more intense, limiting nitrate losses.

Due to sampling difficulties, information on soil ionic levels in the C horizons is limited to that from the deep soil inspection pits. As ionic levels in these C horizons should be less variable than in the overlying solum, these measurements are sufficient to represent average soil ionic levels at depth. Thus these measurements, which are listed in Table 4.2, permit an initial generalized interpretation of soil ionic levels and related processes in the whole profile. The ions are grouped into nitrogen ions--including nitrate and ammonia, cations--including calcium, magnesium, potassium and sodium, and soil acidity, measured as soil pH.

Ammonia and nitrate tend to accumulate in the surface soil layers. In the forest and septic bed, soil surface concentrations

TABLE 4.2a Soil Ions in Solution at Time of Sampling for Adsorbed Cations

Profile and horizon	Depth cm	Cations					Anions		Sums			
		K ⁺	Ca ²⁺	Mg ²⁺	Na ⁺	NH ₃ ⁺	pH ⁺	NO ₃ ⁻	Cl ₂ ⁻	+	-ve	all ⁺
Pit 1 Forest	A1	.000116	<10 ⁻⁶	.0003796	.0002376	.0008733	4.979	.0077392	.0003869	.0015957	.0081251	.0098218
	A2	.000258	.0000773	.0004516	.0001547	.0008899	4.926	.0006107	.0007763	.0018515	.0008870	.0027395
	B	<10 ⁻⁶	.0000276	.0009949	.0008843	.0004753	4.329	.0005527	.0001336	.0023821	.0006853	.0033674
	IC	.0000027	.0002349	.0009811	.00056378	.00031703	4.127	.0003824	.0004974	.0022768	.0008898	.0081166
	IIIC	<10 ⁻⁶	<10 ⁻⁷	.0001657	.000185	.0003315	5.503	.0012712	.000116	.0006657	.0013672	.0028529
Pit 2 Field	Ae	.0002155	.0005941	.0003455	.0107644	.0002094	5.166	.0019345	.0005306	.0121289	.0024651	.0145940
	B	<10 ⁻⁶	.0000224	.0008267	.0001685	.0005057	5.057	.0009534	.0002901	.0015033	.0012435	.0027458
	IC	<10 ⁻⁶	<10 ⁻⁷	.0006531	.0009037	.0002901	5.016	.0005665	.0002763	.0018569	.0008428	.0026997
Pit 3 Septic (East)	A2	<10 ⁻⁶	.0001271	.0003703	.0000884	.0004919	5.135	.0091201	.0003454	.0010777	.0094655	.0105432
	B1	<10 ⁻⁶	.0001188	.0005444	.0000606	.0004775	4.975	.0012989	.0001713	.0011965	.0014702	.0026667
	B2	.000276	.0000746	.0002570	.0000082	.0003869	5.527	.001216	.0000525	.0007543	.0012685	.0020278
	C	.0000055	.0000002	.0004309	.0000055	.0003344	6.102	.0024873	.0002901	.0007765	.0027774	.0035539
Pit 3 Septic (West)	Ap	<10 ⁻⁶	<10 ⁻⁷	.000525	.0000304	.0002901	4.787	.0048037	.0001492	.0009455	.0059529	.0067994
	B1	<10 ⁻⁶	.0004223	.0000706	.0000248	.0005582	4.970	.0014923	.0005941	.0043371	.0020864	.0064235
	B2	<10 ⁻⁶	.0003454	.0012022	.0000010	.000467	6.091	.000912	.0011054	.0020156	.0020174	.0040330
	C1	<10 ⁻⁶	.0000027	.0007434	.0001547	.0002818	6.149	.0004919	.0006218	.0011826	.0011137	.0022963
Pit 4 Septic (Centre)	C2	<10 ⁻⁶	.0000002	.0004696	.0000027	.0004598	4.717	.0003454	.0002763	.0009423	.0006217	.0015640
	A1	.0004836	.0000773	.0000056	.0000082	.0008208	5.377	.0017134	.0003178	.0013955	.0020312	.0034267
	B1	.0002763	.0000016	.0000757	.0000635	.0004781	4.999	.0011607	.0001105	.0008952	.0012712	.0021664
	B2	.000276	.0000746	.0004504	.0000193	.0008125	5.750	.0008014	.000456	.0013944	.0012574	.0026418
Pit 4 Septic (Centre)	IC	<10 ⁻⁶	.0000055	.0004505	.0000497	.0005389	5.672	.0003869	.0003869	.0010446	.0007738	.0018184

⁺ moles l⁻¹ cm⁻³

⁺⁺ pH units

⁺⁺⁺ dimensionless



TABLE 4.2b Adsorbed Cations

Profile and horizon	Depth cm	Adsorbed cations			Summaries		
		Ca meq/100 g	Na meq/100 g	K meq/100 g	Σ meq/100 g	CEC - Σ meq/100 g	Base saturation %
Pit 1 Forest	A1 0-4	.065184	.02367	.04583	.13468	7.605	1.74
	A2 4-10	.07793	.01709	.04524	.14026	1.830	7.11
	B 10-23	.05980	.02501	.04417	.12898	6.011	2.10
	IC 23-43	.11857	.01212	.04014	.17083	5.019	3.29
	IIIC 43-51	.08352	.019645	.048117	.15128	0.739	17.08
Pit 2 Field	A1 0-2.5	.13076	.01266	.04697	.19040	1.740	9.84
	Ae 2.5-18	.12888	.08914	.04135	.26538	2.415	9.89
	B 18-31	.12234	.1467	.04600	.31506	3.325	8.65
	IC 31-38	.13361	.16995	.05408	.35765	1.342	21.00
	I-IIIC 38-56	.11209	.01461	.04007	.16678	7.573	2.16
Pit 3 Septic (East)	IIIC 56-76	.13346	.00906	.04728	.18982	2.820	6.37
	A1 0-2.5	.12253	.01231	.04611	.18094	9.129	7.94
	A2 2.5-11	.12004	.00886	.03494	.16384	7.456	2.15
	B1 11-18	.10063	.01405	.04205	.15673	6.633	2.31
	B2 18-31	.11249	.00814	.04847	.16909	5.401	3.03
Pit 3 Septic (West)	C 31-46	.12168	.00885	.04536	.17590	4.114	4.10
	A1 0-2.5	.018518	.00575	.00523	.0295	0.761	3.73
	Ae 2.5-25	.187005	.02787	.05582	.27069	5.299	4.87
	B1 25-38	.16849	.0358	.04454	.24895	5.991	3.99
	B2 38-61	.14390	.02186	.067065	.23283	4.217	5.24
Pit 4 Septic (Centre)	C1 61-68	.17207	.01726	.048187	.23750	4.883	4.64
	C2 68-76	.168092	.01418	.04370	.22597	3.824	5.58
	A1 0-4	.06200	.017185	.04442	.14362	7.366	1.92
	B1 4-18	.12611	.00959	.03255	.21733	8.003	2.64
	B2 18-25	.06640	.018898	.04261	.12794	4.822	2.59
IIIC 25-38	IC 25-38	.0999	.017963	.04427	.16213	1.988	2.59
	IIIC 38-61	.15936	.01326	.04906	.22168	0.208	51.63

are x2 to x12 as large as underlying horizons. Ammonia varies from 0.31×10^{-3} to 0.89×10^{-3} moles/l, which is less than for other ions in solution. Nitrate levels in the septic bed are high in the A and B horizons and low in the C horizon. The forest soil has low levels of less than 10^{-3} moles/l in the rooting zone, between 4 to 50 cm, and higher levels at depths > 50 cm. In the field area, surface nitrate accumulation, and its decrease with depth, are less marked. As indicated by process 1 in Fig. 4.5, large quantities of ammonia are released to the septic bed in septic tank effluent. Most ammonia is nitrified to nitrate which in turn is denitrified, below 25 cm, in the saturated anaerobic C horizon (process 2). The remaining nitrate and ammonia is adsorbed and cycled by plants in the solum (process 3a).

Plant cycling of soil ions is more important in the reference soils. In the forest, plant cycling of nitrate by trees results in high nitrate levels in the litter layer of coniferous needles, low amounts in the rooting zone, due to root extraction (3c), and greater amounts below this, where nitrate is being lost in drainage (4).

Calcium and potassium ions in solution both show high levels in the solum, between 10^{-4} to 10^{-5} moles/l, and low levels in the C horizons, between 10^{-6} and 10^{-7} moles/l. Potassium levels, usually less than 10^{-6} moles/l, are lower than other measured ions and decrease as soil depth increases. The septic bed receives sodium, calcium and potassium cations which are released in effluent (1) and from parent materials. Most of the calcium and potassium released from parent materials is in the solum (2). Plant cycling of calcium and potassium is also evident (3). Similar cation processes occur in the field and

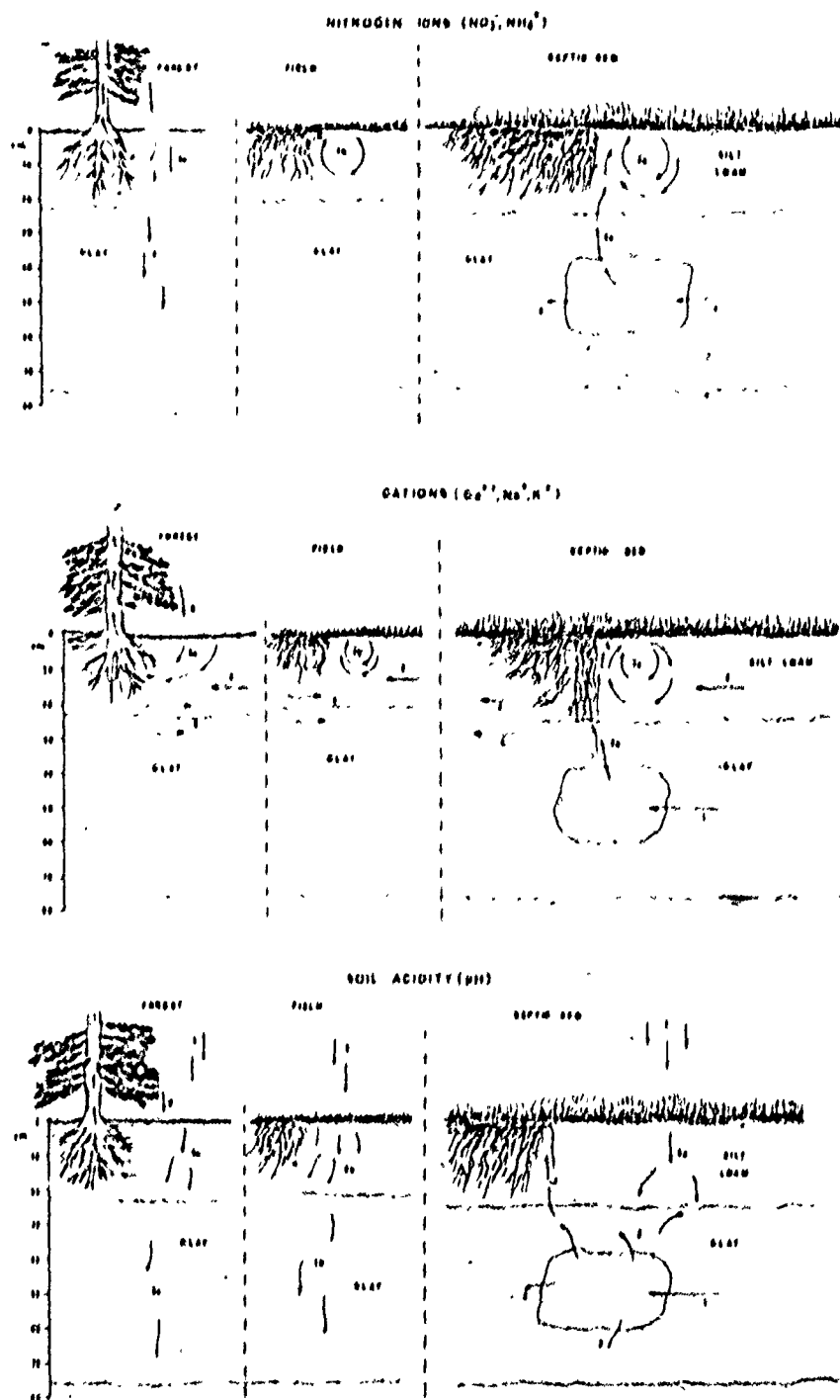


FIG. 4.5 Dominant processes influencing soil ionic levels
see text pages 70, 72 for explanation of numbers

forest soils, except that release and cycling of calcium and potassium near the soil surface involves lower levels (3c).

In the field and forest, the white clay horizon (5) is associated with high levels of sodium and magnesium, up to 10^{-3} moles/l, in solution as well as that adsorbed on the exchange complex. Leaching by effluent and the absence of a white clay layer result in lower sodium and magnesium levels in the septic bed (4).

As indicated in Table 4.2, the field and forest are acidic-- pH ranging from 4.1 to 5.5--while the septic bed tends to be less acidic with a pH from 4.8 to 6.1. Acidity tends to decrease as depth increases in both reference and septic bed soils but for different reasons. Acid rainwater forms acid leachates percolating down the profile producing acidic sola (3). In the septic bed, this leachate is buffered by near neutral effluent, as effluent contributes about 85% of the moisture in the bed (2). In the reference soils, the leachate is weakly buffered by the slightly calcareous clay subsoil (5b, 5c). In the forest soil, this leachate is combined with acidic decomposition products from conifer needles. It is therefore clear that, during continuous effluent flow in the septic bed, most ions are derived from effluent and their movement is controlled by the effluent flow. Due to buffering by effluent, soil acidity does not accurately reflect leaching intensity. Rather, the loss of sodium ions in the lower part of the solum suggests intensive leaching. In the reference soils, plant cycling and rainfall control solute movement, with most ions being released from parent materials at depth, or derived from organic matter at the surface.

This generalized discussion of the whole soil profile is followed and refined by detailed discussion and analysis of the more active solum. The spatial maps of transient soil properties (shown in Appendix E) were studied, especially at the clay-loam interface, at approximately 30 cm depth, where decomposition and lateral effluent flow rates are largest. Spatial maps of polluting and pollutant-sensitive soil properties indicate the extent of the different soil environments within the site area, the spatial extent of some soil processes and the spatial extent of effluent influence.

In natural soil systems, pH and sodium ion levels suggest the intensity of leaching (Hesse, 1971). In the septic bed soil, where the pH is buffered by effluent, free sodium levels are a better indicator of leaching intensity. Using both properties, the leaching intensity in the site area was assessed and this is depicted in Fig. 4.6. The reference areas are minimally leached. The septic bed is intensively leached, especially in the northeast, where the leaching influence has extended into the field.

The relative magnitude of lateral water flow was extrapolated from soil moisture levels at the clay-loam interface. As shown in Fig. 4.7, large lateral water flow occurs in the northeast and southwest. Some groundwater flows into the west end of the bed, while the rest of the bed is characterized by smaller outflows. Lateral flow in the reference soils is very small.

The oxygen diffusion rate, as shown in Fig. 4.8, suggests that the septic bed is anaerobic at depth, becoming aerobic towards the surface with occasional anaerobic pockets extending to the surface.

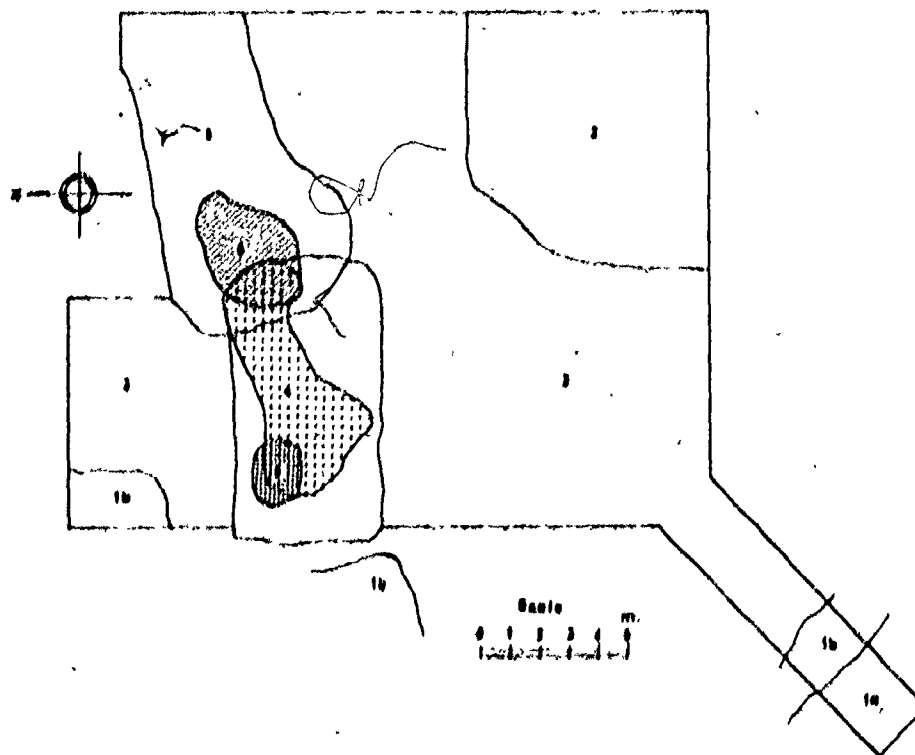


FIG. 4.6 Leaching environments in the site area

(1) Minimum leaching to greatest leaching (6). 1a. forest: low pH = 6.3, high Na 300×10^{-4} moles/l; 1b. forest fringe: lower Na 150×10^{-4} moles/l; 2. wet field: low pH < 6.6, moderate Na 115-150; 3. dry field: neutral pH 6.8-6.9, low Na 75-70; 4. septic bed: very low Na < 65, pH variable; 5. extensive field sink: low pH < 6.8 and Na < 75; 6. intensive field sink: very low pH < 6.6 and Na < 65.

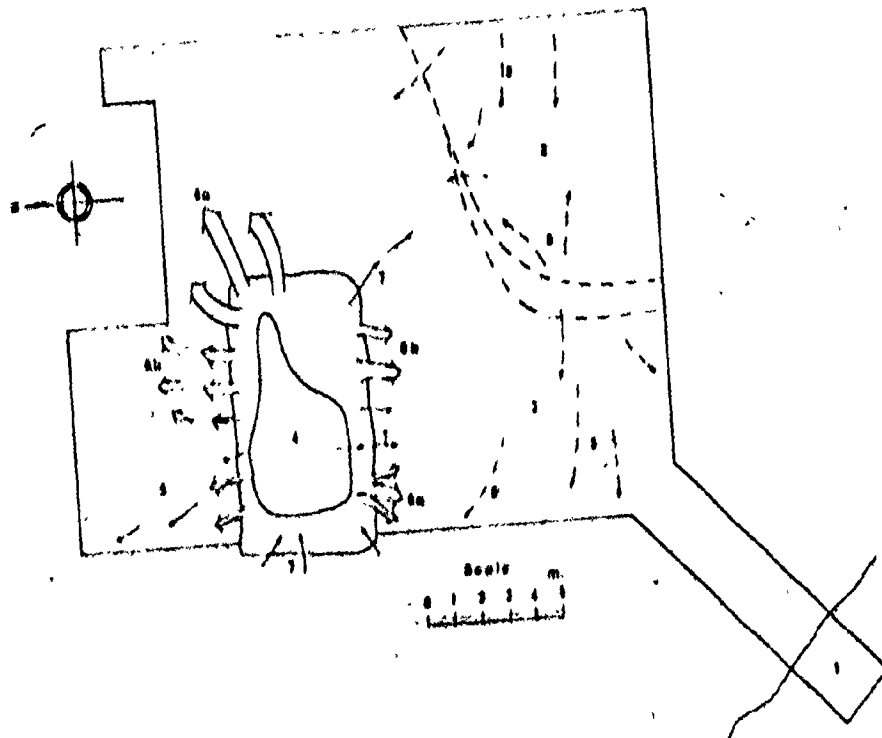


FIG. 4.7 Lateral water fluxes in the site area

1. forest with minimal flow; 2. wet field and 3. dry field with very small flows, and 4. saturated septic bed with very variable flow magnitudes; 5. moderate effluent flux out of bed with a moisture gradient of 2-4%/m; 6a. largest flux out of bed with a gradient of 8-12; 6b. large flux out of bed with a gradient of 4-8; 7. small flow into and out of bed with gradients of <1; 8. very small lateral flows with gradients of <0.5.

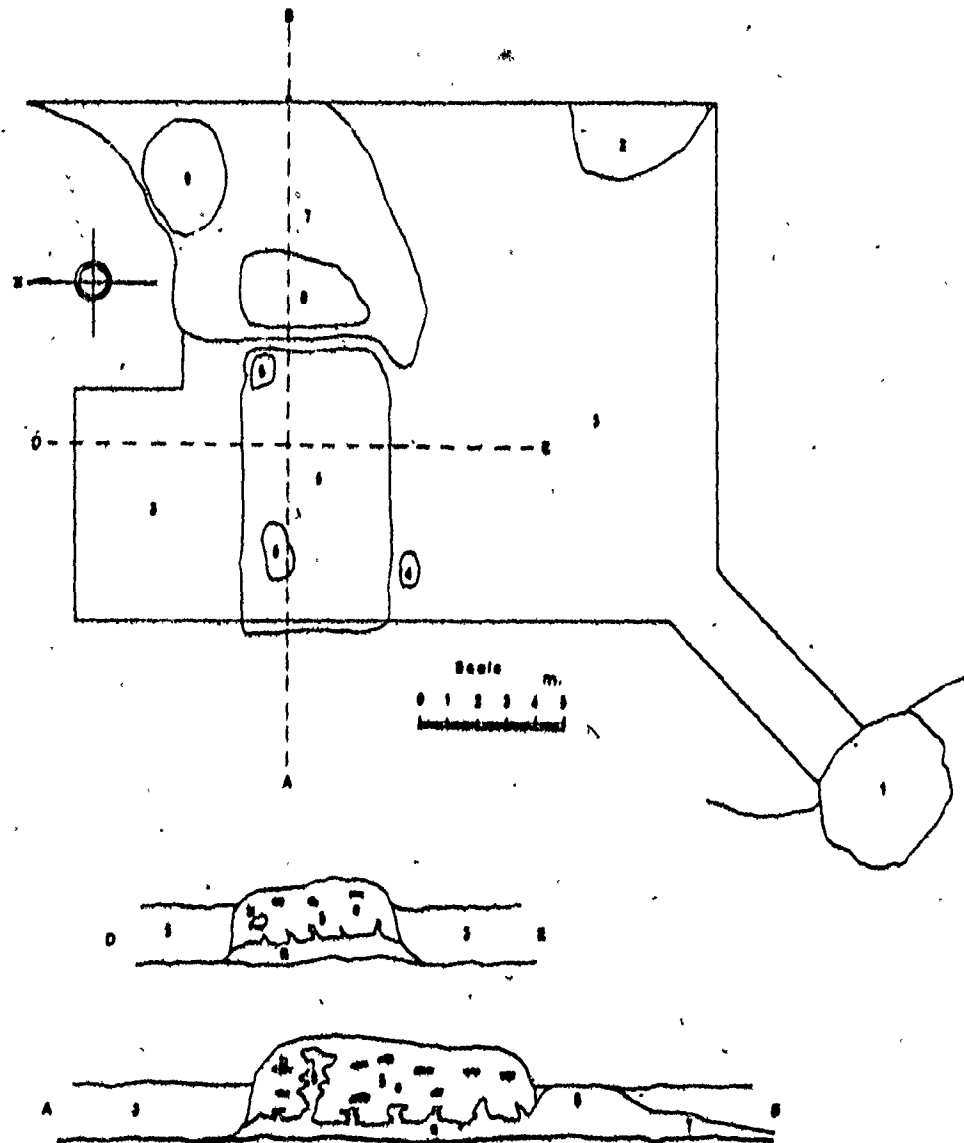


FIG. 4.8 Oxygen status in the site area

1. anaerobic forest pocket: $<18 \text{ ug cm}^{-2} \text{ min}^{-1}$; 2. natural anaerobiosis: <10 at 30 cm depth; 3. natural aerobiosis: 26 at 30 cm to 36 at 7 cm depth; 4. induced anaerobiosis at depth: 6-8 at 30 cm; 5. mixed aerobic-anaerobic bed: 16-22 at 30 cm; 6. anaerobiosis in bed 4-14 at 30 cm; 7. effluent-induced anaerobiosis: <16 at 30 cm; 8. intense induced anaerobiosis at soil surface: 16-20 at 7 cm; 9. aerobic soil surface in fire pit and disturbed ground: 36-48 at 7 cm. Cross-sections A-B and D-E, a. anaerobic effluent-saturated soil; b. anaerobic clay pockets; c. aerobic sand, gravel and silt loam.

Adjacent to the bed in the northeast and southwest, large effluent flows induce intense anaerobic areas. The reference areas tend to be more aerobic.

Using the previous information and studying the nitrate levels, the distribution of nitrate processes was described and is illustrated in Fig. 4.9. Organic nitrogen and ammonia are quickly converted to nitrate in all aerobic areas. This would be most pronounced during the night when the septic bed drains and becomes more aerobic. Denitrification in the mixed aerobic-anaerobic septic bed reduces this large nitrate input to moderate levels. Large effluent outflows in the northeast and southwest are associated with low nitrate levels, whereas moderate lateral outflows along the north and south sides of the bed produce high nitrate levels. This somewhat surprising finding is logical if one considers the oxygen levels. Large effluent flows cause anaerobiosis and contain significant organic carbon, making conditions ideal for denitrification. Conversely, moderate effluent flows do not induce anaerobiosis and the aerobic conditions inhibit denitrification.

A simplified model of the relationships between soil properties during continuous effluent flow is constructed through multivariate analysis. Factor analysis separates soil properties into groups by loading or correlating variables with similar patterns of variations on one factor (Rummell, 1970). The oblique rotation in the analysis indicates some of the relationships between these groups of soil properties. This analysis can be used, for example, to separate soil ionic levels which are influenced by effluent flow from soil ions independent

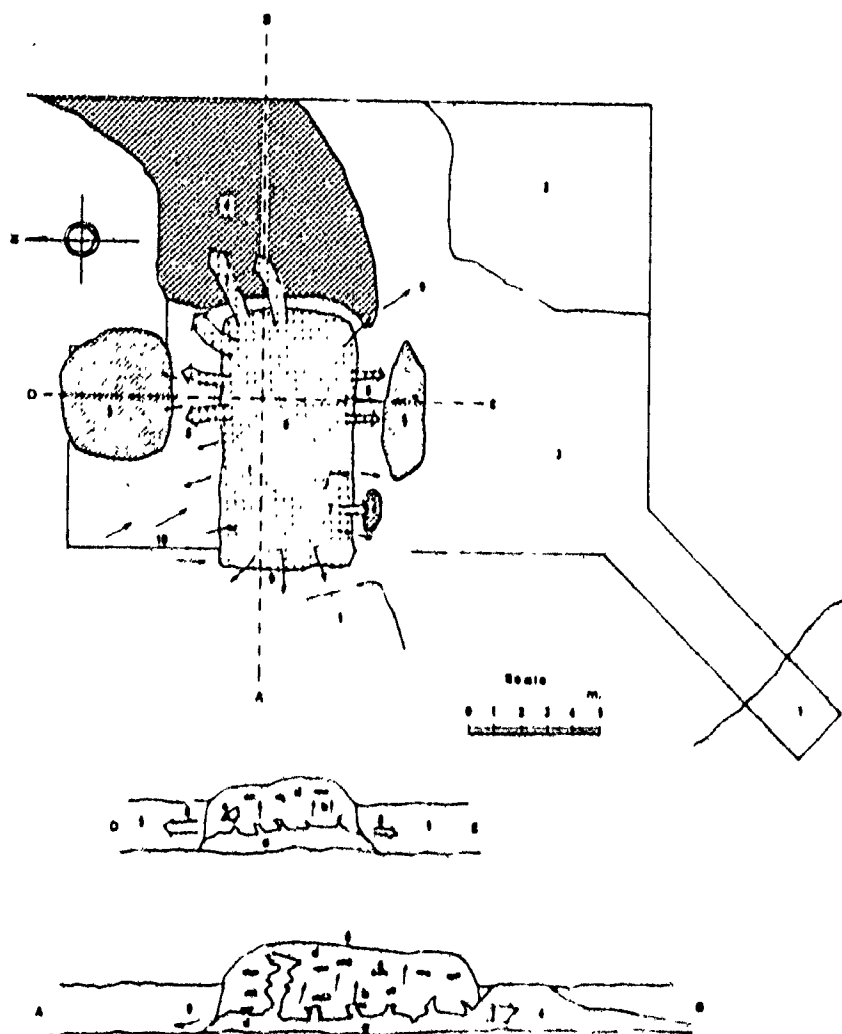


FIG. 4.9 Nitrate processes in the site area

1. natural forest soil: high NO_3 levels, 1000×10^{-5} moles/l, due to limited leaching and N cycling; 2. wet field: low NO_3 , < 525 , limited leaching, low O_2 , $< 24 \text{ ug cm}^{-2} \text{ min}^{-1}$, enhancing denitrification; 3. dry field: moderate NO_3 , 525-625, high O_2 , 24-30, permits nitrification; 4. field sink: moderate NO_3 , 525-600, low O_2 , < 16 , plus carbon in effluent cause denitrification; 5. dry field margin: high NO_3 , 625-725, high O_2 , > 20 , hence no denitrification; 6. septic bed: moderate NO_3 , 525-575, low where $\text{O}_2 < 18$ causes denitrification, higher where O_2 is higher, 22, and denitrification limited; 7. large effluent fluxes containing low NO_3 ; 8. moderate effluent fluxes containing high NO_3 ; 9. minor groundwater flow into bed; 10. small to moderate flow out of bed. Cross-sections A-B and D-E, a. soil saturated with effluent causes denitrification; b. plant uptake of N species; c. anaerobic pockets, denitrification occurs; d. aerobic soil, nitrification occurs.

of this influence. Variations in response of soil ions to effluent flow can also be identified. Table 4.3 summarizes the factors. The relationships between factors are indicated by the factor pattern correlations which are listed in Table 4.4, and by the factor models graphed in Fig. 4.10. The models are constructed by grouping correlated factors.

Oblique factor solutions extract a major soil ionic leaching factor as the initial dominant factor accounting for 43% or more of the variance. The leaching factor consists of soil ions sensitive to leaching, usually sodium, nitrate and pH. Since nitrate levels are also influenced by denitrification and nitrification rates, it is extracted as a separate minor factor in one solution. As expected, this nitrate factor is strongly correlated to the major leaching factor, as indicated in Table 4.4.

A natural soil ionic factor, consisting of calcium and soil moisture, is extracted second and accounts for 31 to 35% of the variance. As the soil profile data and spatial patterns indicate, calcium levels vary less than other measured soil ionic levels and calcium is cycled at the soil surface by plants.

Different minor factors are extracted from the different data matrices. In two instances, soil depth forms a natural factor independent of the influence of effluent. In one instance, the third factor consists of variation of chloride levels. Chloride is sensitive to effluent and to natural soil processes. Soil chloride in solution is rapidly leached and is not adsorbed on the exchange complex nor does it undergo microbial decomposition. Thus, in most natural soils, chloride

TABLE 4.3 Response of Soil Ions to Continuous Effluent Flow as Indicated by Factor Analysis

<u>Factor No.</u>	<u>% Variance</u>	<u>Factor Name</u>	<u>Variables Loaded > ± 0.50</u>
<u>Solum (A and B horizons)*</u>			
1	50.3	soil ionic leaching	PH, NO ₃ , NA
2	33.1	natural soil ionic	CA, H ₂ O ₂
3	16.6	chloride	CL ₂
<u>A horizon**</u>			
1	50.7	soil ionic leaching	PH, NO ₃ , NA
2	34.6	natural soil ionic	CA, H ₂ O ₂
3	14.7	depth	DEPTH
<u>Solum: * data transformed (log₁₀)</u>			
1	43.0	major soil ionic leaching	NA, PH
2	31.4	natural soil ionic	H ₂ O ₂ , CA
3	14.4	natural non-ionic	DEPTH
4	11.5	minor soil ionic leaching	NO ₃

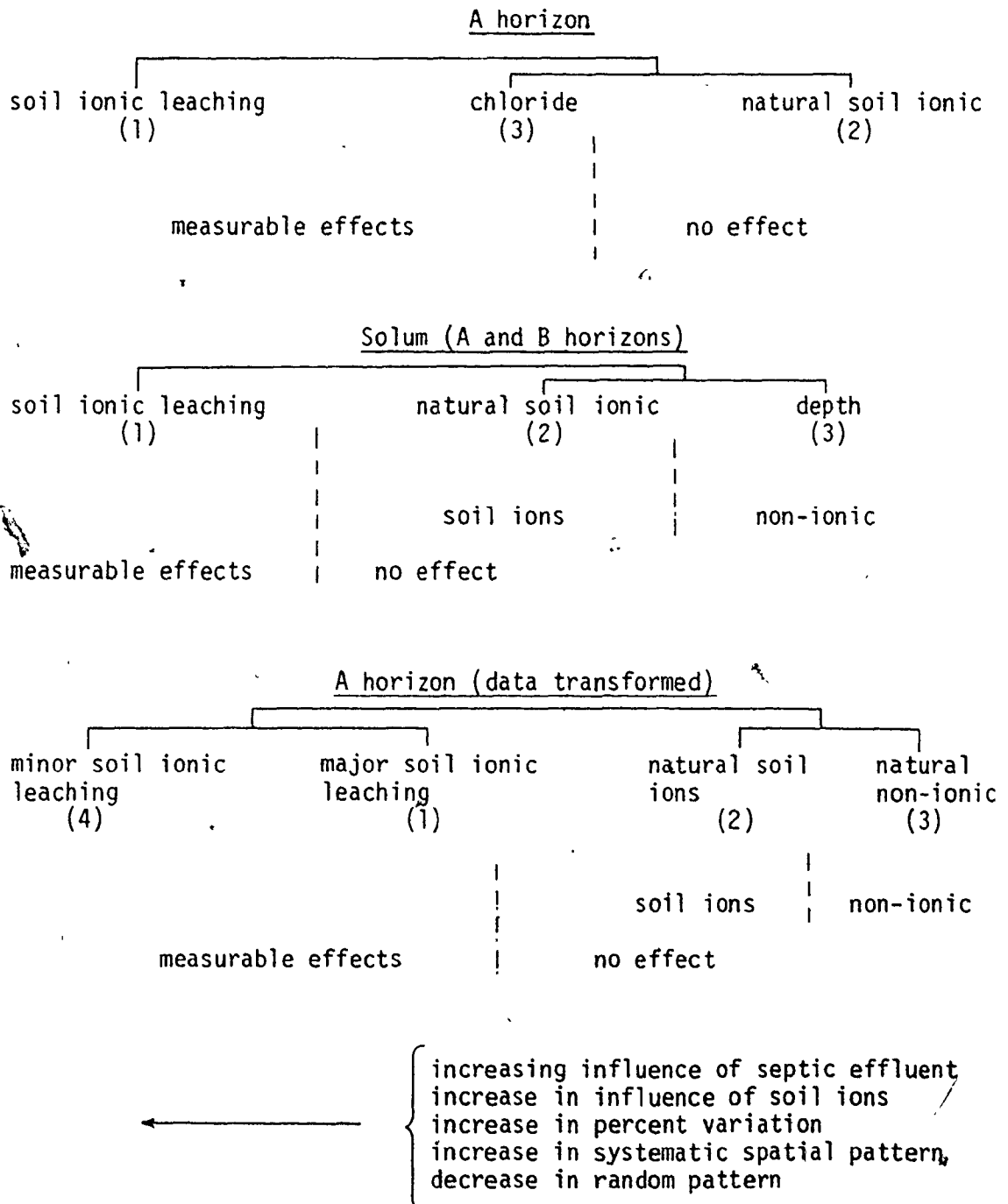
* Includes data down to 60 cm. The matrix consists of 587 cases or samples with values on 7 variables. Oxygen diffusion rate values are not included.

** Includes data down to 30 cm. The matrix consists of 530 cases or samples with values on 8 variables. Oxygen diffusion rate values are included.

TABLE 4.4 The Relationships between Soil Factors for Continuous Effluent Flow as indicated by Factor Pattern Correlations

<u>Solum (A and B horizons)</u>			
	natural soil ionic (2)	chloride (3)	
soil ionic leaching (1)	.06	-.04	
natural soil ionic (2)		-.21	
<u>A horizon</u>			
	natural soil ionic (2)	depth (3)	
soil ionic leaching (1)	.05	-.16	
natural soil ionic (2)		-.31	
<u>Solum: data transformed (\log_{10})</u>			
	natural soil ionic (2)	natural non-ionic (3)	minor soil ionic leaching (4)
major soil ionic (1) leaching	-.05	.20	.23
natural soil ionic (2)		-.21	-.00
natural non-ionic (3)			-.02

FIG. 4.10 Factor models of response of soil ions to continuous effluent flow



levels are very low. High levels of chloride in septic tank effluent produce peaks in the northeast and southwest of the septic bed and in the adjacent field where effluent fluxes are largest. Since nitrate levels in these locations have been reduced by denitrification, chloride is separated from the leaching factor containing nitrate.

The schematic factor models arrange the factors along several environmental gradients. This is most marked for the solution using untransformed measurements from the solum. The soil factors as they are arranged show, from left to right in Fig. 4.10, a shift from spatially heterogeneous natural processes to systematic spatial variation in leaching associated with effluent flow, an increase in the number of soil ionic species loaded and an increase in the percent variance explained by each factor.

The factor analysis thus indicates that pH, nitrate and sodium show relatively similar variations, as they are strongly correlated with and controlled by leaching and effluent flow. Chloride and nitrate show some unique responses to effluent flow and the related processes. Soil calcium levels in solution are clearly independent of effluent flow and the related processes.

b. Response to Commencement of Effluent Flow

At the time of sampling, the soils were drying out, especially at the soil surface where the moisture loss was 10 to 40%, as indicated in Table 4.5. The field and forest soils dried out markedly while the septic bed soil showed little evidence of drying. The related oxygen diffusion rate was measured down to 30 cm depth and these measurements

TABLE 4.5 Soil Moisture Concentrations During Response to Commencement of Effluent Flow

Site	Depth (cm)	Days 0	Soil Moisture Pct. by Weight from commencement of effluent flow			
			3	7	10	13
1 Forest	15	47.5	45.5	31.0	26.0	8.0
	23	36.0	39.0	39.0	36.0	26.0
	36	48.0	59.0	42.5	48.0	29.0
	53	63.0	70.0	58.0	60.0	56.0
	76	52.0	65.0	65.0	26.0	60.0
2 Field	15	47.0	42.5	29.0	29.0	18.0
	28	42.0	18.0	13.5	10.0	5.0
	41	59.0	14.0	29.0	37.0	31.5
	58	45.0	49.0	45.0	45.0	44.5
	76	35.0	36.0	80.0	60.0	74.0
3 Septic Bed East Face	19	40.0	36.0	30.0	30.0	30.0
	30	36.0	35.0	36.0	51.0	55.5
	48	60.0	68.0	65.0	65.0	70.0
	61	75.5	70.0	68.0	62.5	80.0
4 Septic Bed West Face	13	23.0	15.5	15.5	30.0	14.0
	30	11.0	15.5	13.0	15.5	15.5
	48	30.0	37.0	37.0	29.0	38.0
	65	38.5	38.5	46.0	42.0	46.0
5 Centre of Septic Bed	13	42.0	23.5	20.5	23.5	20.5
	28	38.5	65.0	80.0	60.0	38.5
	36	15.5	24.5	21.5	24.5	24.5
	43	32.0	32.0	38.0	35.0	38.0

(in Table 4.6) show distinct temporal patterns. Anaerobiosis starts in the centre of the septic bed at depth and progresses outward towards the edge of the bed. In the centre of the bed at 30 cm depth, the soil is anaerobic in less than 3 days after it begins to receive effluent from the septic tank. In some locations, anaerobiosis extends to the edge of the septic bed in less than 7 days.

Soil ionic levels in the A and B horizons (solum) may be discussed first. Multivariate analysis was used to discern relationships between soil ions as effluent flow started. For canonical analysis, measures of soil properties were used as one set of variates and measures of elapsed time from the commencement of effluent flow comprised a second set of variates. The canonical analysis is summarized in Table 4.7. In the solum, the inclusion of elapsed time from the starting of effluent flow in the initial canonical variates indicates that it is more important than soil depth and influences ionic levels of chloride, nitrate, potassium and ammonia. The response of chloride is restricted to the A horizon as it was only included in the canonical variates for the analysis of this horizon.

Multivariate analysis can also be used to compare the response of the septic bed to continuous effluent flow with the response to the commencement of effluent flow. The factor analysis of the response to continuous effluent flow was presented in Tables 4.3 and 4.4 and in Fig. 4.10 on pages 80, 81 and 82. The factor analysis of the response to commencement of effluent flow is presented in Tables 4.8 and 4.9 and Fig. 4.11. In the analysis of response to continuous flow, the initial soil ionic factor, consisting of sodium, nitrate and pH, accounted for

TABLE 4.6 Oxygen Diffusion Rates During Response to Commencement of Effluent Flow

Site	Depth (cm)	Oxygen Diffusion Rate ($\mu\text{g cm}^{-2} \text{min}^{-1}$)				
		Days from commencement of effluent flow				
		3	7	10	13	17
1 Forest	7	28.7	22.0	23.6	25.3	18.6
	15	30.4	22.0	22.0	25.3	8.4
	21	35.5	30.4	30.4	18.6	16.9
	28	27.0	33.8	20.3	13.5	16.9
2 Field	7	32.1	30.4	13.5	35.5	37.2
	15	16.9	25.3	33.8	32.1	30.4
	21	25.3	27.0	33.8	33.8	32.1
	28	27.0	18.6	13.5	33.8	33.8
3 Septic Bed East Face	7	10.1	33.8	27.0	18.6	18.6
	15	18.6	16.9	28.7	15.2	18.6
	21	20.3	25.3	27.0	20.3	22.0
	28	20.3	25.3	22.0	16.9	22.0
4 Septic Bed West Face	7	41.9	32.1	54.0	33.8	45.6
	15	40.5	50.7	35.5	28.7	31.1
	21	30.4	10.5	8.4	28.7	40.5
	28	35.5	18.6	6.8	16.9	0.0
5 Centre of Septic Bed	7	32.1	47.3	37.2	54.0	43.9
	15	37.2	47.3	33.4	40.5	42.2
	21	30.4	32.1	28.7	30.4	45.6
	28	20.3	30.4	51.5	30.4	11.8

TABLE 4.7 Response of Soil Ions to Commencement of Effluent Flow as indicated by Canonical Correlation Analysis

<u>Canonical</u>	<u>Canonical</u>	<u>Variable Coefficients</u> $\geq \pm .50$	
<u>Variate No.</u>	<u>Correlation</u>	2nd set	1st set
<u>Solum (A and B horizons)</u>			
1	.77549	CL2, NO3	X, XY
2	.70042	CL2, K, NH4	X, X2, XY
3	.66335	NO3, K, NH4	Y, Y2, X, X2
4	.43391	NH4, CL2, P04	Y2, Y, X2, X, XY
<u>A horizon</u>			
1	.71222	NO3	X2, X
2	.48792	NH4, CA, NA	X, XY, Y2

* X is time in days from commencing use of the septic bed
 Y is soil depth in cm
 XY is the time-depth product and represents interaction
 between time and depth
 X2 is time squared
 Y2 is soil depth squared

TABLE 4.8 Response of Soil Ions to Commencement of Effluent Flow as indicated by Factor Analysis

Factor No.	% Variance	Factor Name	Variables Loaded $> \pm .50$
------------	------------	-------------	------------------------------

Solum (A and B horizons)^{*}

1	48.1	major soil ionic	NH ₄ , CA, K, CL ₂
2	21.9	time-soil ionic	NA, TIME
3	16.5	phosphate	PO ₄ , CL ₂
4	13.5	nitrate	NO ₃

A horizon^{**}

1	45.3	major soil ionic	NH ₄ , CL ₂ , CA, K
2	29.0	time-soil ionic	TIME, NO ₃ , NA
3	14.4	depth	DEPTH
4	11.3	phosphate	PO ₄

Profile (A, B and C horizons)^{***}

1	46.7	major cation*soil ionic	NH ₄ , K, CA, CL ₂
2	21.9	time-soil ionic	TIME, NA
3	17.2	anion	PO ₄ , CL ₂
4	14.2	minor soil ionic	H ₂ O (NO ₃) ⁺

* Includes data down to 60 cm. Ions are expressed as moles l⁻¹ ml⁻¹ soil solution. The matrix consists of 187 cases or samples with values on 10 variables. Oxygen diffusion rate values are not included.

** Includes data down to 30 cm. Ions are expressed as moles l⁻¹ ml⁻¹ soil solution. The matrix consists of 100 cases or samples with values on 11 variables. Oxygen diffusion rate values are included.

*** Includes data down to 90 cm. Ions are expressed as moles l⁻¹ ml⁻¹ soil solution. The matrix consists of 200 cases or samples with values on 10 variables. Oxygen diffusion rate values are not included.

⁺ r = -.47

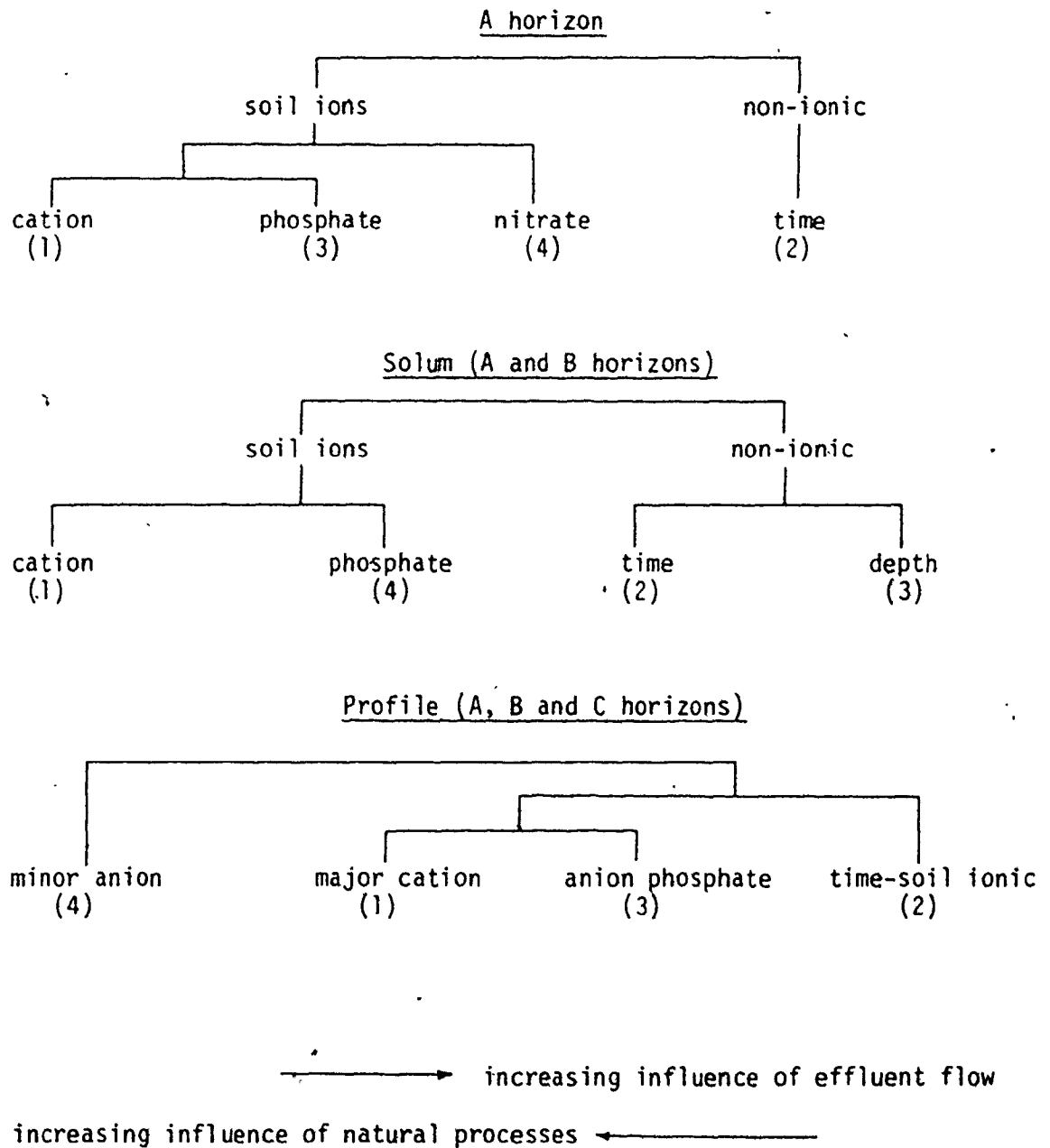
TABLE 4.9 The Relationships between Soil Factors for Commencement of Effluent Flow as indicated by Factor Pattern Correlations

<u>Solum (A and B horizons)</u>			
	time-soil ionic (2)	phosphate (3)	nitrate (4)
major soil ionic (1)	.00	.30	-.21
time-soil ionic (2)		-.02	-.16
phosphate (3)			-.10

<u>A horizon</u>			
	time-soil ionic (2)	depth (3)	phosphate (4)
major soil ionic (1)	.06	-.07	.29
time-soil ionic (2)		.21	-.05
depth (3)			.05

<u>Profile (A, B and C horizons)</u>			
	time-soil ionic (2)	phosphate anion (3)	minor soil ionic (4)
major soil ionic cation (1)	.05	.20	-.05
time-soil ionic (2)		-.11	-.03
phosphate anion (3)			-.03

FIG. 4.11 Factor models of response of soil ions to commencement of effluent flow



up to half of the variance. Minor factors consisting of calcium, chloride, soil moisture and depth were extracted subsequently. The major difference in the factor solutions is a soil ionic and time factor composed of time elapsed from the starting of effluent flow and sodium and nitrate levels.

The schematic factor models show similar gradients as described for the spatial models. There is a gradient from left to right, shown in Fig. 4.11, which is in response to commencement of effluent flow. For example, for the A horizon, the cation factor consisting of ammonia, calcium, potassium and chloride is on the left, while to the right is the soil ionic and time factor consisting of sodium, and of time elapsed from effluent input. The two factors are independent and uncorrelated ($r=.00$). The cation factor consists of ions influenced by natural processes and is not influenced by commencement of effluent flow. Between these two extremes, in Fig. 4.11, the phosphate and nitrate factors are more closely related and correlated with the cation factor ($r=.30$ and $r=-.21$, respectively) and weakly correlated with the soil ionic and time factor ($r=-.02$ and $r=-.16$, respectively).

The limited data on the C horizons was graphed. Ionic levels in runoff from a ditch 15 meters distant from the septic bed soil are also included. Variation in ionic levels after effluent flow was started is shown in Fig. 4.12 and 4.13. Sodium and ammonia levels show clear trends. Ammonia does not accumulate, as it is adsorbed (Taylor and Ashcroft, 1972) and rapidly nitrified to nitrate (Walter, 1974). Increased sodium ion levels could be due to increased release of sodium from the clay, and from sodium contained in cottage waste.

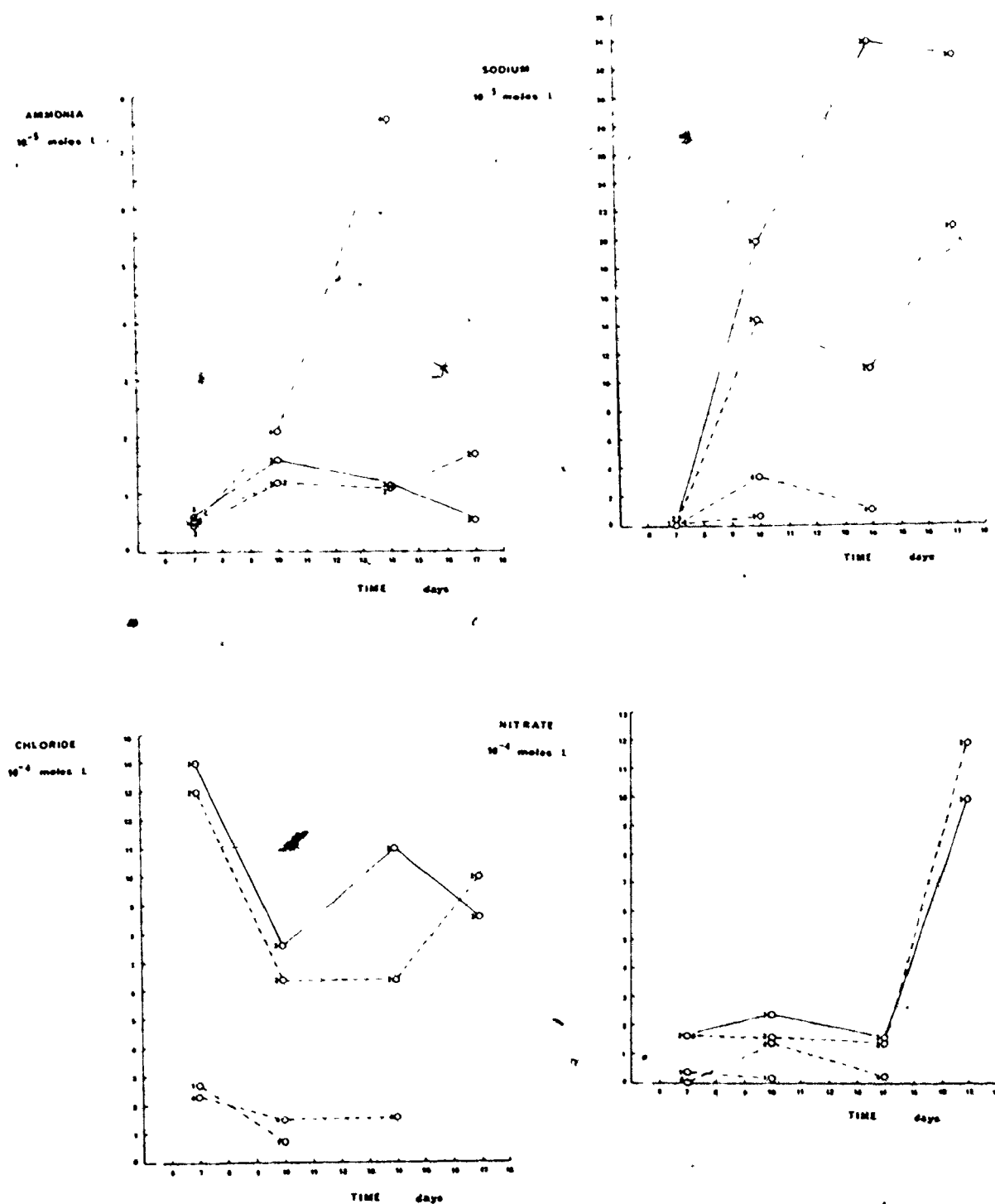


FIG. 4.12 Soil ionic levels of NH_4 , Na , Cl_2 and NO_3 in the C horizons during response to commencement of effluent flow

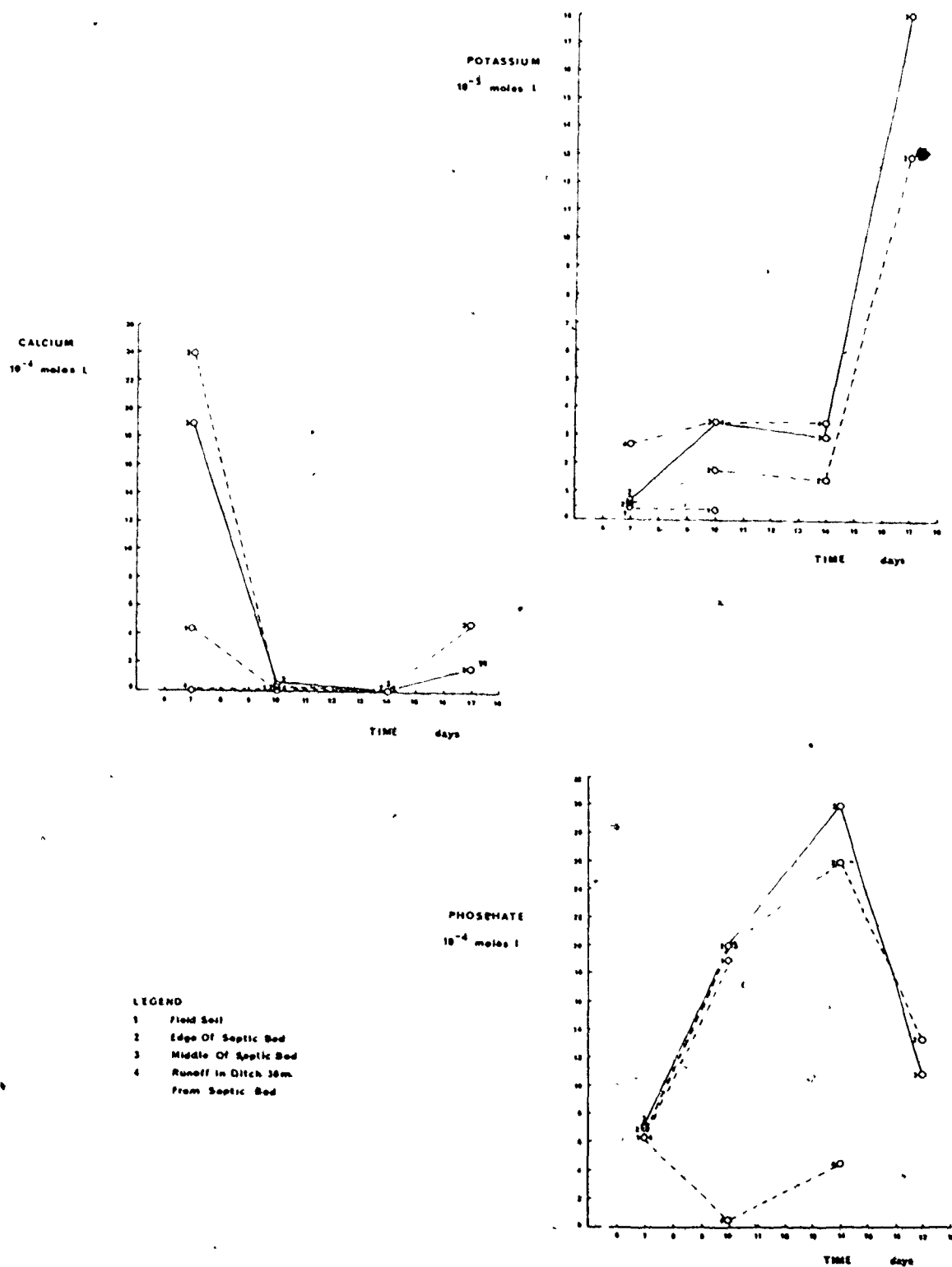


FIG. 4.13 Soil ionic levels of K, Ca and PO₄ in the C horizons during response to commencement of effluent flow

Although chloride levels do not indicate a clear trend, levels are usually higher in the centre of the bed than in the reference sites. Low initial levels of nitrate and potassium were followed by marked increases of nitrate and potassium in the septic bed after two weeks. These increases are probably related to increased effluent input. Calcium and phosphate exhibit markedly high initial levels in the septic bed soil followed by lower levels. Calcium and phosphate are probably being precipitated as calcium phosphates (Swahney and Hill, 1975).

The response of soil ions after commencement of effluent flow in the A, B and C horizons are compared next, and the word 'time' is used to represent such temporal responses. Canonical correlation analysis (Table 4.7) indicated that the response of soil ionic levels to commencement of effluent flow is linear and is more important than soil depth. Also, the influence of time decreases with depth and is greater with anions than for cations. The correlation coefficients computed from the variations in soil ionic levels, after starting effluent flow, are summarized in Table 4.10 and in Fig. 4.14. These correlation coefficients suggest that the strength of the relationships between ion concentrations and time and depth usually decreases as sampling depth increases. Sodium ion levels, which are independent of depth, and calcium and ammonia ion levels, which are independent of time elapsed and depth, are exceptions. The variation in responses with time, of nitrate, phosphate and chloride levels, are restricted to the soil surface. Of these, time influences nitrate more than sodium, and sodium more than phosphate. The multiple regression

TABLE 4.10 The Influence of Time from Commencement of Effluent Flow on Soil Ionic Levels as indicated by Correlation Analysis

(a) Relationships between soil ions

soil horizons	A	A + B	A + B + C
maximum sampling depth (cm)	30	60	90
no. of cases	100	187	200
correlated variables			
NH ₄ , CA	<u>.73</u> *	<u>.64</u>	<u>.50</u>
NH ₄ , CL ₂	<u>.79</u>	<u>.52</u>	<u>.52</u>
NH ₄ , K	<u>.61</u>	<u>.58</u>	<u>.59</u>
CL ₂ , PO ₄	<u>.34</u>	<u>.46</u>	<u>.46</u>
CL ₂ , K	<u>.50</u>	<u>.37</u>	<u>.37</u>
K, DEPTH	-.40	-.23	-.15
K, CA	.43	.42	.33
NA, NO ₃	.46	.21	.22

(b) Relationships of soil ions with time and depth

TIME, CA	-.01	.16	.01
TIME, NH ₄	-.07	-.13	-.15
TIME, CL ₂	-.30	-.10	-.11
TIME, PO ₄	-.43	.02	.02
TIME, NO ₃	<u>.63</u>	.09	.08
TIME, NA	<u>.37</u>	.35	.33
TIME, K	.08	.00	.17
DEPTH, K	-.40	-.23	-.15
DEPTH, NO ₃	-.04	-.05	.02
DEPTH, NH ₄	-.24	-.11	-.15
DEPTH, PO ₄	-.26	-.04	-.02
DEPTH, CL ₂	-.12	-.10	.06
DEPTH, NA	.11	.05	.01
DEPTH, CA	.00	.01	-.06

* Correlation coefficients ≥ 0.50 are underlined.

(c) Multiple Regression Equations*

<u>Dependent variable</u>	<u>Soil horizon</u>	<u>Independent variables</u> ^{**}	<u>Multiple R</u>	<u>Multiple R²</u>
N03	A ^{***}	TIME, NA, CL2, CA, P04	.76	.58
N03	A, B ^x	P04, CL2, NA, CA	.52	.27
N03	A, B, C ^{xx}	H2O, P04, CL2, NA, CA	.53	.29
P04	A	TIME, DEPTH, NA, CL2, CA, N03	.65	.42
P04	A, B	CL2, N03, NA, CA	.62	.38
P04	A, B, C	CL2, N03, NA, CA	.62	.38
NH4	A	CL2, CA, K, TIME	.90	.81
NH4	A, B	CA, K, CL2, TIME	.79	.62
NH4	A, B, C	K, CL2, CA, TIME, NA, P04	.75	.56

* significant at the 5% level

** in order of inclusion in the equation

*** down to 30 cm, 100 cases

^x down to 60 cm, 187 cases

^{xx} down to 90 cm, 200 cases

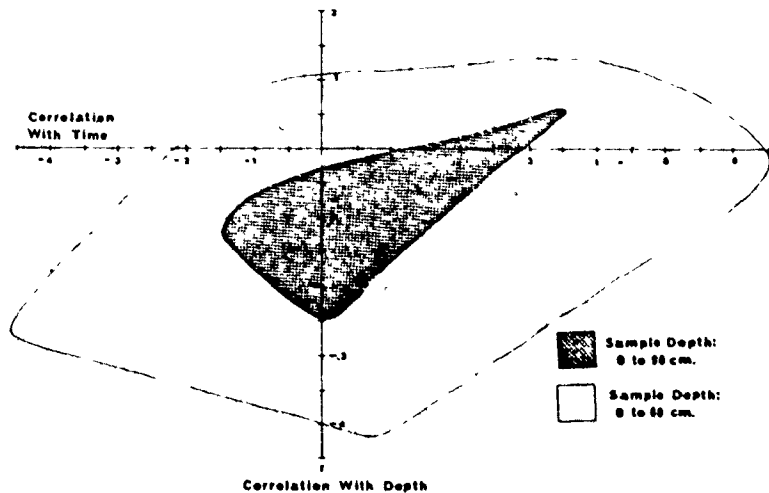
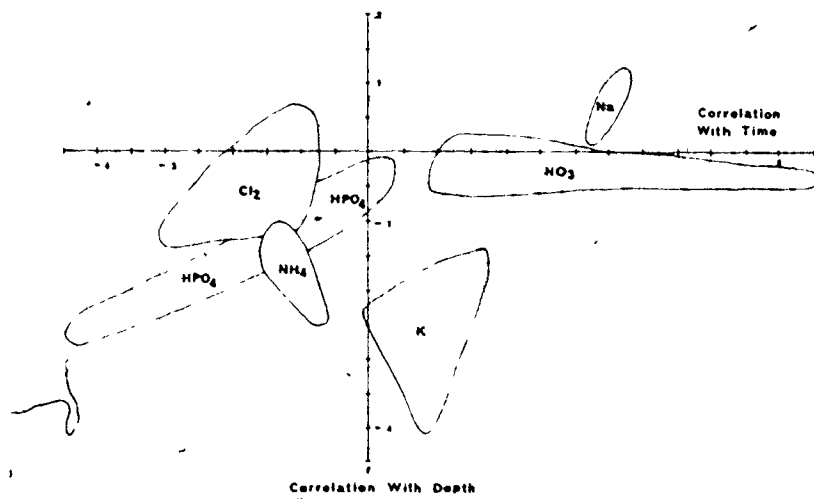
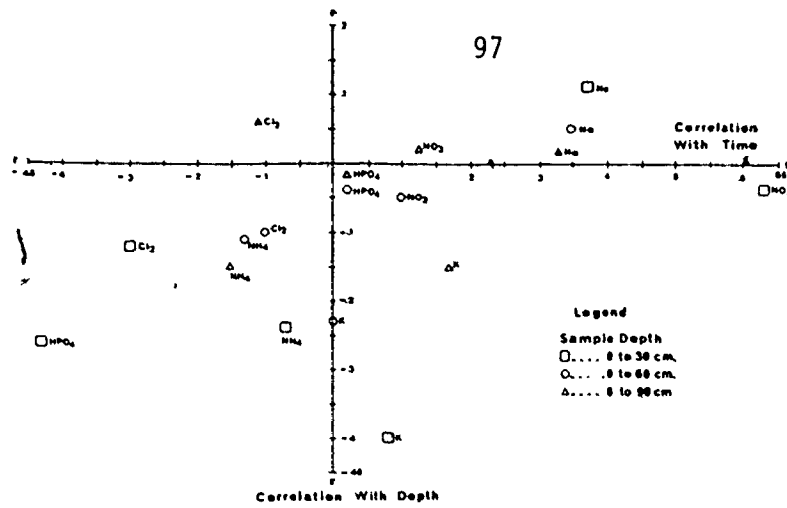


FIG. 4.14 The influence of time from commencement of effluent flow on soil ionic levels as indicated by correlation analysis

equations computed from variations in soil ionic levels, after starting effluent flow, are summarized in Table 4.10.

The multiple regression equations for predicting ammonia indicate that at depth, potassium competes with ammonia for adsorption sites (Taylor and Ashcroft, 1972; Lance, 1972; Magdoff et al., 1974). In the multiple regression equation for nitrate, soil moisture content is more important at depth as higher moisture contents increase anaerobiosis and denitrification and reduce nitrate levels. This is also illustrated by the factor solutions where, at depths greater than 60 cm, nitrate becomes more independent of time and more dependent on soil moisture content.

In summary, at the soil surface or A horizon, soil moisture variations, temperature gradients and plant uptake are most pronounced. Here the soil is most variable, processes are most intense, environmental gradients are steepest and much of the interaction occurs. Thus marked responses over time to the commencement of effluent flow and natural drying or plant cycles occur in the surface soil horizon. In the C horizon, responses to starting effluent flow also occur as nitrate is denitrified, calcium and phosphate ions precipitate and ammonia and potassium ions compete in the exchange complex.

c. Response to the Cessation of Effluent Flow

As described in the methods section (pp. 40,44), measurements of soil ionic levels and soil moisture were made, along a transect, at intervals of 13 and 26 days after cessation of effluent flow. From these measurements, maps of soil moisture and soil ions were produced

by computer and are included in Appendix E. Interpretations from these maps are discussed as follows and are also summarized in Fig. 4.15.

The moisture regime in the septic bed soil changes after the effluent flow ceases. When the septic bed soil is in use, it is saturated between 30 and 60 cm depth. After the cessation of effluent flow, rainfall saturates the 30 to 60 cm portion of the bed periodically for short periods of time, while for the rest of the time, it is relatively dry. The pH level and nitrate content showed marked changes, calcium and sodium levels showed less response. Nitrate is leached in waves by rainfall during infiltration and adsorbed by plant uptake between rainfall events. The pH level drops as acid rainwater replaces near neutral effluent. The low pH and low calcium and sodium levels in the centre of the septic bed suggest intensive leaching after effluent flow ceases.

The sand and gravel in the septic bed loses water rapidly, causing the soil to dry quickly after effluent flow ceases and between rainfall events. This in turn leaches calcium and sodium out of the bed. Pockets of disturbed clay are dispersed randomly throughout the A and B horizons of the bed. The clay pockets retain moisture longer, reducing leaching and moisture fluctuations, resulting in localized higher pH, calcium and sodium levels. The lower acidity and the increased cations and moisture in these clay pockets provide a more favourable environment for plant roots than the sand and gravel. Thus the clay pockets are associated with plant uptake of nitrate after effluent flow ceases.

These responses to the cessation of effluent flow are shown

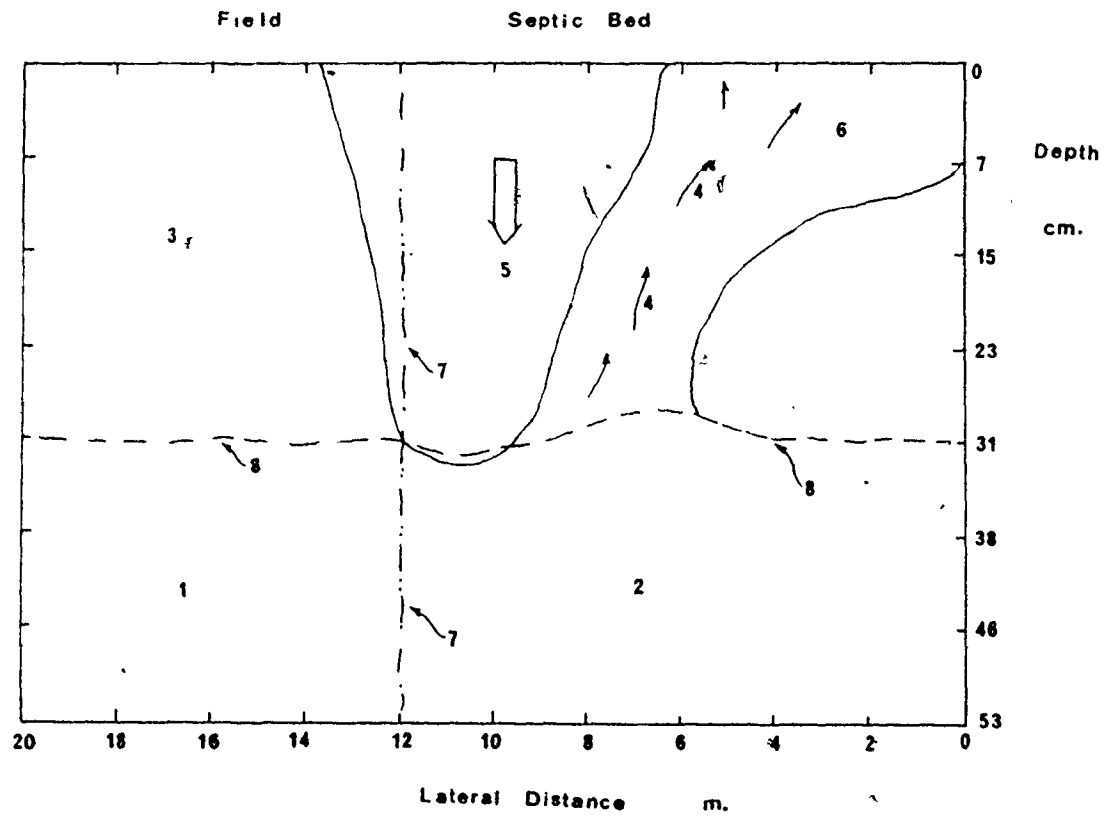


FIG. 4.15 Response of the septic bed soil to cessation of effluent flow

along the transect through the septic bed in Fig. 4.15. The coarse textured material in section no. 5 at the centre and eastern end of the bed is intensively leached, with low calcium and sodium ion levels and a stable acidic level. Soil moisture contents show periodic fluctuations from 25 to 45% soil moisture. The clay pocket, no. 6, in the west end of the septic bed, is associated with plant uptake of nitrate (no. 4) and is consistently wet in excess of 40% soil moisture, with high calcium, sodium and pH levels. The wet septic bed subsoil, no. 2, has high sodium, calcium and pH levels which decrease after effluent flow ceases. In this zone, moisture fluctuations are large, ranging from 20 to 60%. The dry field surface horizons above 30 cm, no. 3, change little, while the subsoil, no. 1, which is at or below the upper limit of saturation, shows similar but less intense responses than the septic bed subsoil (no. 2).

This very rapid recovery from septic input alleviates concern over the storage of nutrients for long periods of time in an unused septic bed, and their release at inopportune times. If nutrients in the septic bed were released slowly after effluent flow stopped, nutrients would be stored over winter and released during the spring thaw. Low winter temperatures would limit the biological and chemical degradation of the effluent and limit moisture losses, as water in snow and ice would be unavailable for transport. Thus large volumes of water containing high nutrient concentrations would be released during the spring thaw and would pose a serious pollution hazard.

4.3 Summary of Observed Soil Properties

The parent or original soil materials consist of permeable loams overlying impermeable clays. Alterations to the septic bed during its construction have also lowered its adsorptive capacity and increased its hydraulic conductivity and porosity. Since the bed consists of disturbed soil and of sand and gravel in tile trenches between undisturbed soil, the bed is a complex and heterogeneous soil body. During continuous effluent flow, factor analysis suggests that leaching influences sodium, nitrate and chloride levels, with the secondary processes of nitrification, denitrification and chemical precipitation influencing ammonia, nitrate and phosphate levels respectively. Spatial analysis indicates that the lower B and C horizons are saturated, with large variations in effluent flow at the edge of the septic bed. Where these lateral flows out of the bed are small, significant losses of nitrate occur. Where the outflows in the north-east and southwest are large, significant losses of sodium and phosphate occur. Despite large amounts of ammonia released from the septic tank, most of it is nitrified, adsorbed and volatilized within the septic bed.

The combination of disturbance during construction and leaching of sodium has removed the white clay IIC1 horizon. This leaching of sodium suggests that changes in soil structure and fabric are occurring, which are increasing both lateral and vertical conductivity and effluent flow out of the septic bed soil. Moderate levels of sodium tend to disperse soil particles and inhibit the formation of stable soil aggregates. When wet, these dispersed soil aggregates are highly impermeable to water or air (Taylor and Ashcroft, 1972). The low

hydraulic conductivity of the white clay layer is probably due to this dispersion and thus losses of sodium should increase aggregation and hydraulic conductivity.

Field and forest soils are drier and more aerobic with no large lateral soil ionic fluxes. Some nitrate is lost to drainage below the rooting zone in the forest. Factor analysis indicates that plant uptake and rainfall cycles control soil ionic levels in these sites.

After the commencement of effluent flow, there was a shift in dominant processes in the septic bed soil from plant uptake and rainfall cycles to leaching by effluent. Also, after commencement of effluent flow, the centre of the bed and the soil below 30 cm depth rapidly became anaerobic. Phosphate precipitation and denitrification in the C horizons followed. The processes controlling ammonia equilibrated almost immediately. Conversely, chloride, nitrate and sodium leaching losses were still increasing 20 days after effluent flow had commenced.

After cessation of effluent flow, the C horizon became aerobic between rainfall events and plant uptake of soil ions exceeded soil ion leaching losses, especially near clay pockets. For example, nitrate was leached in waves by rainfall and adsorbed by plant uptake between rainfall events.

This information confirms the validity of the assumptions on which this study is based. Interpretations from the spatial maps indicate that there are large variations in lateral effluent flux of soil ions, including nitrate and phosphate. Studying the response of the septic bed to commencement and cessation of effluent flow indicates

that different responses occur in different parts of the bed. For example, after cessation of effluent flow, most nitrate in solution is leached out in parts of the bed, while in others, some is retained by plant cycling. These complex spatial and temporal variations confirm the first hypothesis that the septic bed is a heterogeneous system which cannot be represented by average levels.

Results of the spatial analysis, especially of the multivariate analysis, indicate which ions in the soil solution are sensitive to effluent flow and their varying degrees of sensitivity. The results clearly show that calcium ions in solution are not sensitive to effluent flow. Conversely, sodium is readily leached by effluent flow, causing changes in soil structure and increased vertical flow of effluent and nutrients to groundwater. This tends to support the assumption that the solid soil components may be changing because of the impact of this flow. The remaining assumptions, concerning the degree of pollution which is occurring, and the limitations of concentration measurements, are evaluated in the following chapter.

Following this qualitative interpretation of soil properties, a mathematical approach will be used to provide quantitative estimates of these soil processes and solute fluxes.

CHAPTER V

MASS BALANCE CALCULATIONS

5.1 General Water and Nutrient Flux Patterns

Mass balance calculations are used to provide estimates of nutrient and water flow in the forest, field and septic bed soils. Losses of nitrate, ammonia and phosphate to groundwater are calculated. The mathematical techniques for estimation of these fluxes are explained in Chapter III and the translation of these equations into FORTRAN and their organization into computer programs are explained in Appendix C. Comparison of estimates of flux with interpretations from the observed soil properties contained in the previous chapter explains both the processes and the reasons why these losses occur.

Despite large variations in nutrient and water flow, nutrient losses from the base and sides of the septic bed are consistently larger, by at least one order of magnitude, than in the reference sites. These calculations suggest that 80% of the flow is out of the base of the septic bed. Figs. 5.1 and 5.2 show that water flows are much greater than nutrient fluxes. Among the nutrients, the flux of nitrate is greater than those for phosphate and ammonia, which also vary in their relative magnitudes.

The soil horizons influence water flow and show differing levels of soil processes. In the A1 horizon, water is lost by

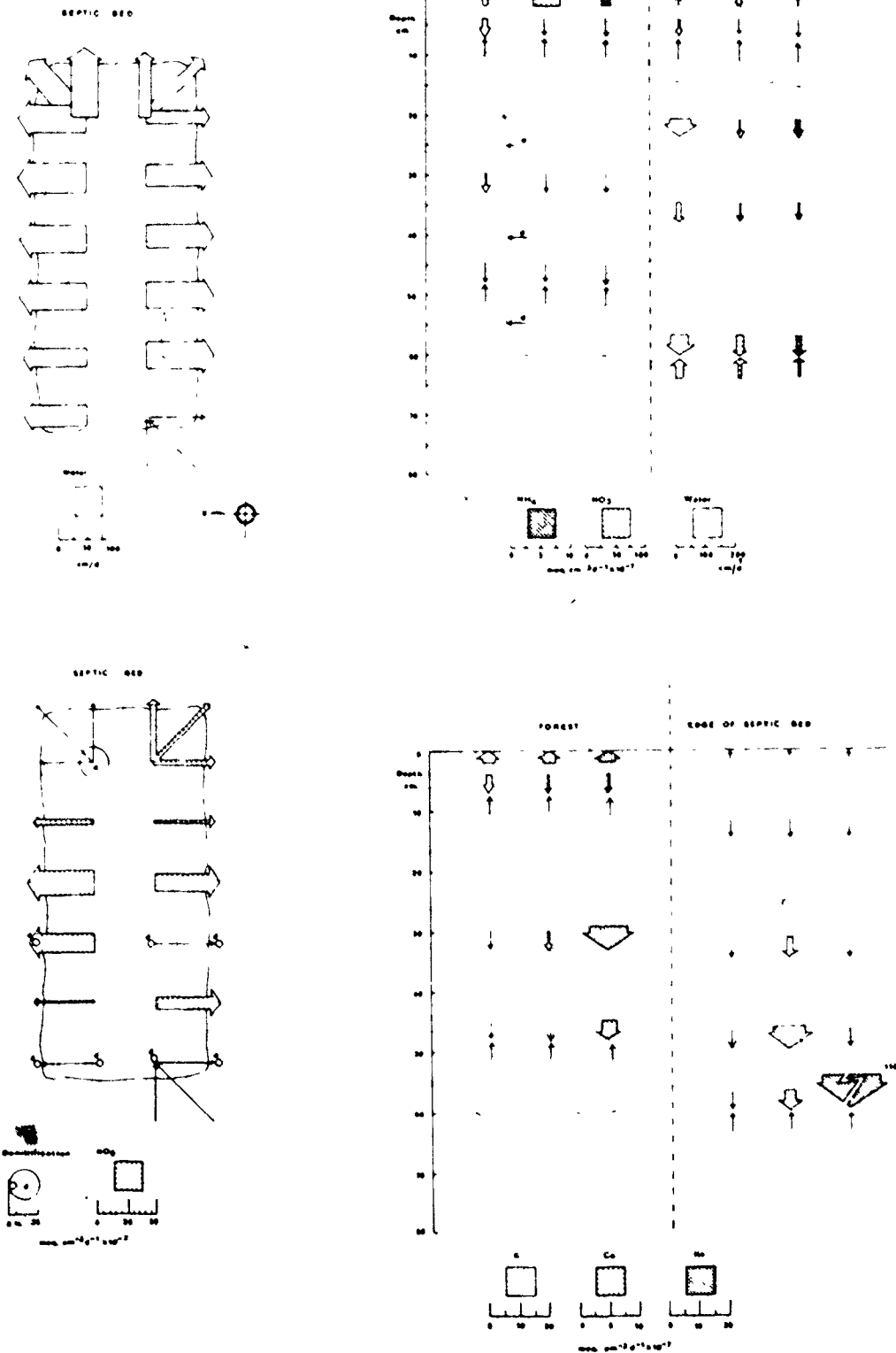


FIG. 5.1 Computed water and nutrient fluxes during continuous effluent flow

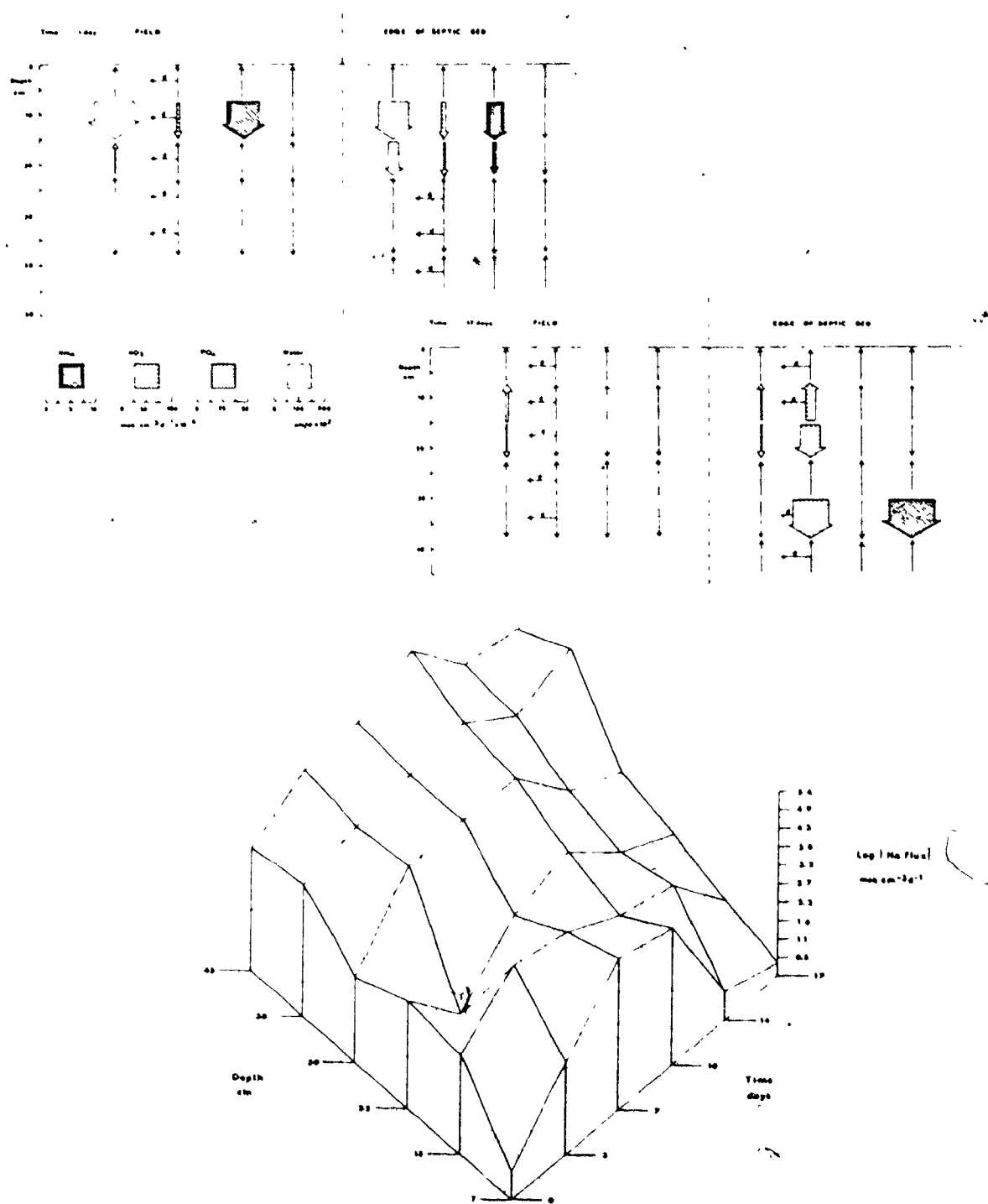


FIG. 5.2 Computed water and nutrient fluxes after commencement of effluent flow

evapotranspiration. In the Ae and Ap horizons, nutrients are alternately leached by moisture flow and adsorbed by plant uptake. This cycling extends through to the upper B horizons. In the lower B horizons, nutrients are leached and plant uptake is not evident, in contrast to the situation within natural soils. Below the B horizon, there is a small but consistent capillary uptake in the clay C horizons. The exception to this pattern is in the upper B horizon in the centre of the septic bed, where plant uptake occurs consistently. The largest fluxes are associated with plant cycling in the Ae and Ap horizons in all sites and leaching in the lower B horizon at the edge of the septic bed.

The pattern of estimates of NH_4 fluxes is similar to the spatial patterns described in the analytical results in the previous chapter. The analysis of measurements of soil ions in solution from the soil in the solum indicated that nutrients are cycled and adsorbed by plant uptake in the solum. The spatial analysis indicated that the leaching intensity varied in the site area soils. These analyses did not, however, indicate the magnitude of these processes or fluxes.

5.2 Water and Nutrient Fluxes in the Septic Bed

The pattern of lateral moisture fluxes derived from spatial mapping of soil moisture is not identical to the calculated pattern. This is partly due to differences in mathematical technique. Spatial mapping averages the moisture gradient over a large area and permits flow at any angle in the horizontal plane. In calculating flow, less data is used and fluxes are portrayed as at right angles to the edge

of the septic bed. Both techniques predict similar lateral moisture flows, except in the west end of the septic bed, where the flows of groundwater into the bed are in different locations. Large lateral moisture outflow from the bed occurs, except in the southeast corner where only moderate flows occur.

Denitrification has produced a much more variable pattern of lateral nitrate fluxes. Denitrification in the northeast, northwest and at one location on the south side of the bed has reduced much of the nitrate in the effluent flow. The chloride-nitrate ratio predicts the occurrence of denitrification but quantitatively underestimates it.

From the interpretation of observed soil properties in Chapter IV, it was hypothesized that most of the water and nutrient losses from the septic bed would occur as lateral flow along the interface between the permeable loams of the solum and the impermeable clay C horizons. The calculations of flow indicated that 54% of the lateral water flow and 60% of nitrate flux occurred between 30 and 60 cm depth, which corresponds to the sand layer on top of the clay. The calculations suggest that these lateral flows range from small groundwater inflows to large outflows of up to 80% of the average vertical flow. The calculated flows indicate that, on average, 87 and 95% of the water and nitrate fluxes in the lower B horizon are vertical.

Immediately below this large vertical flux in the lower B horizon are small flows of capillary uptake in the clay C horizons, as indicated in Fig. 5.1. This means that the water and nutrients in the lower B horizon cannot be leaching through the C horizon to groundwater. They can only be flowing laterally across the top of the clay

in a very thin layer of soil and are not detected by samples or moisture sensors placed in the centre of the soil horizons. The mass balance calculations seriously underestimate total lateral flux in the lower profile because they do not detect the large fluxes at the silt loam-clay interface. In addition to this large lateral moisture and nutrient flux, mapping also accurately predicts spatial relationships between oxygen status, denitrification and moisture and nitrate fluxes.

Ammonia volatilization, adsorption and mineralization, which involves the conversion of ammonia to nitrate in the presence of oxygen, and plant uptake in the solum limit ammonia losses. Ammonia mineralization intensifies as the septic bed reaches equilibrium, as indicated by the decrease in ammonia flux in the solum 20 days after commencement of effluent flow (Fig. 5.2).

Low nitrate flux occurs in the lower B horizon in the septic bed due to intense denitrification. Above and below this, nitrate flux is greater. Denitrification is limited in the C horizon by a lack of organic carbon, and in the upper B and A horizons by a more aerobic soil environment. From the analytical results, it was assumed that nitrate losses in the forest would exceed those in the septic bed; however, calculated fluxes of nitrate reaching the C horizon of the septic bed are much larger than in the forest, as shown in Fig. 5.1.

The interpretation of observed soil properties in Chapter IV suggested that precipitation of calcium and phosphate was occurring in the C horizon of the septic bed two weeks after effluent flow started. Computed phosphate flux reaching the septic bed C horizons from the lower B horizon was $0.449 \times 10^{-3} \text{ meq cm}^{-3} \text{ d}^{-1}$, which is about 10^3

greater than in the reference soils. Assuming that phosphate precipitation occurs as phosphate flows across the top of the clay, phosphate fluxes would be reduced to $0.131 \times 10^{-4} \text{ meq cm}^{-3} \text{ d}^{-1}$. Despite this large reduction in phosphate flux by precipitation, phosphate losses are still 1-2 orders of magnitude greater than in the reference soils.

Neither the interpretation of measured soil properties nor the mass balance approaches alone provide an accurate assessment of water and nutrient fluxes. For example, accelerated nitrate and phosphate losses in the septic bed are not readily apparent from the concentration measurements. Also, the mass balance calculations do not detect large lateral effluent flows which are intuitively obvious from the soil profile or fabric and its physical properties. The most significant contribution of the interpretation of observed soil properties is to explain calculated flows of water and of soil ions in solution.

5.3 Nutrient Loadings

The flows of water and of nitrate, ammonia and phosphate in solution out of the sides and base of the septic bed and down the profile in the reference soils are shown in Tables 5.1 and 5.2. The derivation of these figures is explained in Appendix C.

At 69 cm/d, water flows in the septic bed are 25 to 50 times those in the reference soils. At 0.131×10^{-4} , 0.234×10^{-5} and $0.915 \times 10^{-7} \text{ meq cm}^{-3} \text{ d}^{-1}$, respectively, phosphate, nitrate and ammonia fluxes reaching the C horizon of the septic bed are one to three orders of magnitude larger than those reaching the C horizons of the reference sites.

TABLE 5.1 Computed Water and Nutrient Fluxes from the Septic Bed Soil (C Horizon)

Material	Average lateral flux	Average vertical flux	Total lateral flux	Total vertical flux	Total flux	% Vertical of total
Water (cm/d)	34.25 *	60 *	$.641 \times 10^7$ **	$.432 \times 10^8$ **	$.496 \times 10^8$ **	87
NO ₃	$.442 \times 10^{-6}$ x	$.234 \times 10^{-5}$ x	$.828 \times 10^{-1}$ xx	1.685 xx	1.768 xx	95
NH ₄		$.915 \times 10^{-7}$ x		.06588 xx		
PO ₄		$.131 \times 10^{-4}$ x		9.432 xx		

x meq cm⁻³ d⁻¹

xx meq/d

* 2 cm²

** for all septic bed

TABLE 5.2 Computed Water and Nutrient Fluxes from the C Horizons

Soil	Water ⁻¹ (cm d ⁻¹ cm ⁻²)	NO ₃ (meq cm ⁻³ d ⁻¹)	NH ₄ (meq cm ⁻³ d ⁻¹)	PO ₄ (meq cm ⁻³ d ⁻¹)
Septic bed	69*	.234x10 ⁻⁵ *	.915x10 ⁻⁷	.131x10 ⁻⁴
Field	1.0	.358x10 ⁻⁸	.532x10 ⁻⁹	.222x10 ⁻⁶
Forest	2.7	.902x10 ⁻⁸	.247x10 ⁻⁸	.577x10 ⁻⁶

* These fluxes include vertical and lateral flow. All others are vertical flow only.

5.4 Sodium Fluxes in the Septic Bed Soil

In the literature, attention has been focussed on soil ions considered to be pollution threats, and the cations, potassium, calcium and sodium, are not considered as such. The spatial analysis indicated very low sodium levels in solution and adsorbed in the septic bed soil and these are shown in Figs. 4.4 and 4.6. This was attributed to the low adsorptive properties of the added sand and gravel and intensive leaching by large effluent flows.

The mass balance indicates large fluxes of sodium ions in solution leaving the original septic bed soil and this is shown in Figs. 5.1 and 5.2. Some of this sodium is from the original soil materials and has been mobilized by effluent flow. The rest of the sodium is from the effluent itself⁶. As effluent flows laterally across the top of the white clay layer, alterations to the clay layer increase the rate of effluent flow and groundwater pollution. As was previously noted, the sodium and magnesium concentrations are associated with the white clay layer. The white clay has a lower hydraulic conductivity than other soil materials, as shown in Fig. B.2; Appendix B, and this is probably due to increased levels of sodium and magnesium (Bolt and Bruggenwert, 1976; Taylor and Ashcroft, 1972). As sodium and presumably magnesium are leached out of the white clay layer, the conductivity of this layer increases and vertical and lateral fluxes

⁶ Although not discussed directly, it is fairly obvious from the pH and phosphate characteristics of waste sources using detergent, e.g., washing machine and showers, that most of the sodium in effluent is from detergent (Bennett et al., 1975; Witt et al., 1975).

increase. As these flows increase, saturation or ponding in the septic bed soil diminishes, limiting denitrification and shortening the amount of time that the effluent will remain in the septic bed. Biological and chemical mechanisms have less time to process effluent and a gradual and long-term escalation of nutrient losses occurs from the septic bed.

In conclusion, it can be maintained that these estimates of nutrient and soil ionic fluxes indicate both a serious pollution potential and long-term changes in the soil fabric of the bed which aggravate this problem.

CHAPTER VI

SUMMARY AND CONCLUSIONS

6.1 Summary of Results

The soil properties relevant to this study are of two types-- stable properties, associated with soil solids, controlled by processes which are slow, except when altered by man as in the construction of the septic bed soil; transient soil properties, associated with the soil solution and gases and controlled by the variable influences of rainfall, of plant uptake and the input of septic effluent.

The parent or original stable soil materials consist of permeable silt loams overlying impermeable clays. In the septic bed soil, additions of sand in the solum and gravel in the C horizons, and the removal of the IIc1 horizon have reduced its adsorptive capacity and increased its porosity. For the added gravel, the decrease in specific surface area and hygroscopic moisture content by 40%, and the low organic carbon and cation exchange levels of 1% and 1 meq/100 g, respectively, indicate very low adsorptive or 'processing' capacity.

The reference field and forest soils are very similar, except that there has been more leaching of the field soil since deforestation. The natural fertility of these soils is low with most nutrients, organic carbon and exchange capacity concentrated near the soil surface by plant cycling.

The low ammonia losses confirm that nitrification of ammonia is fairly complete despite measured low oxygen levels in the bed. This finding is best explained by reference to the time of sampling and the diurnal cycle of bed utilization. Oxygen levels were measured during the early afternoon when the septic bed was saturated with effluent. At night, when use of the system stops, the large flow out of the C horizon of the bed (69 cm/d in Table 5.2) indicates that most of the bed would drain in the 8-10 hours elapsed by the following morning. As it drained, the bed would become progressively more aerobic, permitting nitrification of adsorbed ammonia.

This study therefore reveals two substantial groundwater pollution probabilities from phosphate and nitrate losses from the septic bed. Ammonia does not appear to be a serious contributor to groundwater pollution as it represents less than 12% of the nitrogen losses from the septic bed. Ammonia is first adsorbed on the exchange complex, then almost quantitatively converted to NO_3 . Denitrification, where it occurs in the septic bed soil, reduces the nitrate flux close to the level of fluxes in the reference soils, but its distribution is sporadic, permitting large vertical and lateral fluxes out of the septic bed. Plant uptake, the other natural sink, is not sufficient to adsorb the remaining nitrate. In the septic bed soil, there is no evidence of phosphate removal by plant uptake or precipitation before it reaches the C horizon. Although precipitation of phosphate in the C horizon can reduce phosphate flux by 1 to 2 orders of magnitude, losses are still much larger than in the reference soils.

When the septic bed soil is in equilibrium with effluent flow, it is moist to wet in the solum and saturated in the C horizon. Thus air-filled porosity is low in the septic bed soil. The reference soils are drier with more air-filled pores. The septic bed soil environment during continuous effluent flow can be described as follows. A tile trench containing physically-disturbed and added materials is surrounded by undisturbed soil. In the trench, predominantly aerobic sandy loams in the top of the solum are underlain by a thin layer of disturbed clay. Below this is anaerobic, effluent-saturated gravel in the base of the trench. The drainage tile is centred in this gravel. Clogging due to organic deposits occurs along the edges of the tile trench. Small vertical fluxes of soil solution pass through the clogged organic layer from the effluent-saturated gravel. The predominant fluxes are lateral over the surface of the clay and out of the tile trench. Within the solum, anaerobiosis develops in clay pockets which are from the original C horizons.

The average soil ionic levels during continuous effluent flow are controlled by several processes. Plant cycling in the forest and septic bed soil accumulates nitrate in the surface horizons, as its levels are from 2 to 12 times greater than in the underlying horizons. Conversely, ammonia varies only to a slight degree. At depth, nitrate is controlled by other processes. Most of the ammonia in the effluent which flows into the bed is converted to nitrate and this is lost by denitrification in the anaerobic horizons below 25 cm soil depth. Most of the remaining ammonia is adsorbed by the exchange complex. The remaining nitrate is lost by drainage or cycled by plants in the solum.

In the field and forest soils, plant uptake and cycling of ammonia and nitrate are the dominant processes.

Among the cations, calcium and potassium display similar patterns and are influenced by similar soil processes. Both exhibit high levels in the solum, of 10^{-4} to 10^{-5} moles/l, and low levels in the C horizons, of 10^{-6} to 10^{-7} moles/l, with potassium showing the lowest levels. This depth gradient may also be attributed to plant cycling. High levels of sodium and magnesium, up to 10^{-3} moles/l in solution, are associated with the white clay layer in the field and, forest and this with adsorbed sodium and magnesium. Removal of the white clay layer during construction of the septic bed soil and intensive leaching during its use have reduced sodium and magnesium levels in solution by one to two orders of magnitude below the corresponding levels in the reference sites.

Acid rainwater has produced naturally acid forest and field soils. Near neutral effluent buffers this acidity below 30 cm, resulting in large pH gradients in the septic bed soil.

The spatial patterns of pollutant and pollutant-sensitive variables during continuous effluent flow indicate the distribution within the site area of (i) soil environments and gradients, (ii) soil processes and (iii) the impacts of effluent. Using the spatial patterns of soil leaching, of lateral water fluxes, of oxygen levels and of nitrate levels, the nitrogen attenuation and transport in the septic bed is described. Effluent, containing large quantities of organic nitrogen and ammonia, is released from the septic tank. Most of the organic nitrogen and ammonia is bound and converted to nitrate at night,

when the soil is more aerobic, and denitrified to nitrogen gas during the day, when the soil is predominantly anaerobic. This produces moderate nitrate levels within the septic bed. The remaining nitrate is leached laterally out of the septic bed soil, resulting in two types of nitrate flux: (i) large lateral effluent flows, containing high levels of organic carbon, in the northeast and southwest, which induce anaerobiosis, causing most of the nitrate present to be denitrified, and resulting in low nitrate levels; (ii) small lateral effluent fluxes out of the north and south sides of the septic bed soil which promote more aerobic soil conditions, inhibiting denitrification and resulting in high nitrate levels. Small quantities of groundwater, low in nitrate, flow into the septic bed in the northwest.

Simplified models of the relationships between soil properties during continuous effluent flow were constructed from multivariate analyses. Factor analysis separates soil properties into groups by loading or correlating properties with similar patterns of variation on one factor. The oblique rotation of the axes in the analysis indicates some of the relationships between these groups of soil properties. These properties which are sensitive to, and controlled by, soil leaching include sodium, pH and nitrate levels in solution, and are extracted as the initial dominant factor. Since nitrate is also sensitive to denitrification and nitrification, in some factor solutions it is extracted as a separate minor factor. Since soil moisture and calcium respond to natural or non-effluent processes as well, they are extracted as the second factor. Factor models constructed from the factor pattern correlations differentiate the soil properties

along several environmental gradients, as there is a shift from spatially heterogeneous natural processes influencing the natural and non-soil ionic factors to systematic spatial variations in leaching intensity often associated with effluent flow.

After effluent flow commences in the septic bed, there is a shift from plant uptake and rainfall cycles to leaching by effluent flow. After effluent flow ceases, the reverse shift in processes occurs. In the solum, the magnitude of response to effluent flow for soil ions is $\text{Na} > \text{NO}_3 > \text{PO}_4 \gg \text{Ca} \approx \text{K}$, as sodium, nitrate and phosphate fluxes in the septic bed increase. In the C horizons between 30 to 60 cm depth, ammonia is adsorbed and nitrified and does not accumulate, whereas sodium, chloride and nitrate leaching losses are still increasing 20 days after the commencement of effluent flow. After two weeks of operation, calcium phosphate precipitation at this depth lowers calcium and phosphate levels. Also, in the C horizons, after starting effluent flow, the centre of the septic bed, and most of the soil below 25-30 cm, becomes predominantly anaerobic within 3 and 7 days respectively.

After the cessation of effluent flow, there is a rapid response or recovery of the septic bed soil, following the shift from leaching by effluent flows to plant uptake and leaching by rainfall. In the solum, the centre and eastern end of the bed is leached of nitrate and of calcium and sodium by rainfall cycles. At the same depth in the west, plant uptake associated with clay pockets retains moisture, calcium, sodium and nitrate in solution. In the C horizons at 30 to 60 cm soil depth, moderate leaching and anaerobiosis during successive rainfalls reduce soil ion concentrations of sodium and nitrate developed

during continuous effluent flow. This rapid recovery alleviates the danger of long-term storage of nutrients in an unused septic bed and their release during winter and spring when they are more likely to pollute groundwater.

In comparing the response of soil ionic levels in the different horizons to the commencement and cessation of effluent flow, it is apparent that soil moisture variations, temperature gradients and plant uptake are most pronounced in the solum. This is where the soil is most variable, processes are most intense, environmental gradients are steepest and where much of the interaction occurs. Thus, the most marked responses to effluent flow, natural rainfall and plant cycles occur in the solum.

Following this qualitative interpretation of soil properties, quantitative estimates of some of these soil processes and solute fluxes are calculated. The mass balance calculations provide estimates of nutrients and water flows, including losses to groundwater in the forest, field and septic bed soils.

Among the nutrients, the nitrate flux is greater than phosphate and ammonia fluxes. In the A1 horizon, the largest flux is due to evapotranspiration. In the Ae, Ap and upper B sub-horizons, cycles of plant uptake and leaching occur. In the lower B horizon, large leaching losses occur. There is a small capillary uptake in the clay C horizons. The exception to this pattern occurs in the upper B horizon in the centre of the septic bed where plant uptake occurs. The largest fluxes are due to plant cycling in the Ae and Ap horizons of all sites and leaching in the lower B horizons at the edge of the

septic bed.

There is general agreement between the pattern of lateral moisture flows derived from the spatial maps of soil ionic levels and calculated lateral flows in the septic bed soil. Denitrification estimates, although uniformly low, explain the differences between the magnitudes of lateral water and nitrate fluxes. The calculations also underestimate the lateral flows out of the septic bed. However, large vertical flows in the lower B horizon are underlain by small capillary uptake. This means that water and nutrients in the lower B horizon are not leaching through to the C horizon, but are flowing laterally across the top of the clay, as was predicted by the preceding qualitative discussion.

In the septic bed, ammonia fluxes are limited primarily by nitrification and do not increase. Denitrification in the lower B horizon produces the lowest nitrate fluxes. Above and below this, nitrate fluxes are larger, as denitrification is limited by more aerobic soil and a lack of organic carbon, respectively. Neither interpretation of measured soil properties nor modelling approaches alone provide an accurate assessment of water and nutrient fluxes. For example, accelerated nitrate and phosphate losses in the septic bed are not readily apparent from concentration measurements. Also, the mass balance calculations do not detect large lateral effluent fluxes which are intuitively obvious from the soil profile or the soil fabric and its physical properties. The qualitative analysis provides an explanation for the calculated fluxes.

These calculations also permit evaluation of nutrient loadings

to groundwater in the forest, field and septic bed soils. The magnitude of vertical fluxes of water and nutrients in solution reaching the C horizons follow the sequence: $H_2O < PO_4 < NO_3 < NH_4$. At 69 cm/d, water flows in the septic bed soil are 25 to 50 times those in the reference soils. At 0.131×10^{-4} , 0.234×10^{-5} and 0.915×10^{-7} meq $cm^{-3} d^{-1}$, respectively, phosphate, nitrate and ammonia fluxes reaching the C horizons of the septic bed soil are 1 to 3 orders of magnitude larger than the corresponding fluxes in the reference soils.

The nitrate losses from the septic bed soil are primarily due to the sporadic distribution of denitrification in the C horizons, which in turn is due to aerobic conditions at these locations. The phosphate losses from the septic bed are due to incomplete precipitation which is limited by available precipitation sites. Since ammonia represents less than 12% of the nitrogen losses from the septic bed, it is not considered a serious contributor to groundwater pollution.

6.2 Conclusions

In the introduction, it was noted that most studies of septic pollution survey several beds for pollutant concentrations and pollutant-sensitive variables at a few sites in each bed, so that results can be generalized. In so doing, it is assumed that the septic bed is homogeneous with limited temporal and spatial variability. The concentration or level of pollutant substances is used to assess the degree of pollution. In Chapter II, it was pointed out that such generalizations have not been too successful. This study looks at the temporal and spatial variability at many points both within and without one septic

bed. In addition to the levels of substances, the actual losses or movements of these materials out of the septic bed are calculated.

Thus, a set of alternate or opposing assumptions have been established as a basis for this study contrary to the assumptions on which most studies of septic pollution are based. These alternate assumptions may usefully be restated:

1. The septic bed is a heterogeneous system which cannot be represented by average levels. Spatial and temporal variations in soil properties are large and significant. The response of the septic bed soil when use commences or when it ceases is also considerable and significant.
2. Estimates of fluxes are necessary to quantify septic pollution, in addition to the simpler and easier measurements of pollutant concentrations. Nutrient losses of nitrogen and phosphorus as ions in solution will probably be significant, even under favourable conditions, and their real impact can only be detected by estimates of the nutrient flux.
3. The internal solid components may themselves be changing. Processes may be operating which have not previously been associated with septic bed soils and the intensity of soil processes may not be the same in natural and septic bed soils.

A sampling methodology was designed and carried out to provide the measurements needed to evaluate these assumptions, which, in turn,

can be tested against the results of this study. This evaluation of the assumptions may now be stated:

1. The septic bed is not homogeneous and average levels gained from spot checks are meaningless. For example, part of the bed denitrifies nitrate efficiently and part does not. This gives an inverse relationship between water and nitrate flux which is not suggested in the literature. For example, a large effluent flow produces a low nitrate loss from the bed while a moderate effluent flow may produce a high nitrate loss. This raises the question of where to sample effluent losses from the bed, as it is traditionally done where flow is largest (Walker et al., 1973b).

Efficient nitrification and sporadic denitrification in a bed which is predominantly anaerobic during the day and more aerobic at night suggests that, for periods of large effluent flow during the day, the value of aerobic conditions may have been overestimated and the value of anaerobic conditions underestimated. Thus, the ideal system in Fig. 2.1 is not so, and a rethinking of existing septic system designs is necessary.

The temporal response, in the bed, to commencement and cessation of effluent flow, indicated the role of plant uptake in controlling nutrient levels, which has not been reported before. The rapid response in the bed to the cessation of effluent flow alleviates the danger of long-term storage of nutrients in an unused bed and their release at inopportune times.

2. Although widely used, concentration data can be misleading and should be supplemented with estimates of nutrient flux. In this study, nitrate concentrations in the forest in the C horizons exceed those in the septic bed at the same depths; however, due to root cycling and different rates of soil water flow, nitrate losses are largest in the septic bed.

Serious losses of nitrate and phosphate occur from this and probably from most septic systems due to design weaknesses. The work of Bouma (1974, 1975), Laak et al. (1975) and Deamish (1972) suggests that losses could be controlled with moisture barriers below the bed and/or by the addition of calcium and iron compounds, and this capacity for internal processing of nitrogen and phosphorus should be built into most systems.

3. The solid components of the septic bed are being altered by leaching. For example, large losses of sodium from the septic bed in the C horizon indicate ~~that~~ long-term changes in soil fabric are occurring which will probably accelerate effluent flow losses from the septic bed soil. Existing legislation and the literature assume no change. These long-term changes reinforce the need for monitoring, since nutrient losses from a septic system could increase over time with the degradation of the underlying clay at the interface.

APPENDIX A

INDEX OF TERMS

<u>Mathematical</u>	<u>Computer</u>	<u>Description</u>	<u>Dimensions</u>
C_{NaA}	BNA, CBNA	concentration of adsorbed Na	meq/cm ³ soil
C_{KA}	BK, CBK	concentration of adsorbed K	meq/cm ³ soil
C_{CaA}	BCA, CBCA	concentration of adsorbed Ca	meq/cm ³ soil
z	Z, CZ	Σ adsorbed Na, Ca, K	meq/cm ³ soil
C_{NaS}	YNA	concentration of Na in solution	meq/cm ³ soil
C_{KS}	YK	concentration of K in solution	meq/cm ³ soil
C_{CaS}	YCA	concentration of Ca in solution	meq/cm ³ soil
CEC	CEC	cation exchange capacity	meq/cm ³ soil
K_m	XKM1	monovalent equilibrium constant	dimensionless
K_d	XKD1	divalent equilibrium constant	dimensionless
a_1	Y	parameter in equation (11) containing known or measured ion quantities	moles/l
b_1	X	parameter in equation (11) containing known or measured ion quantities	moles/l
K_{SAT}	KSAT	conductivity	cm/h
θ_{vSAT}	H2SAT	soil moisture content at saturation	% volumetric

Mathematical	Computer	Description	Dimensions
---	XB	moisture content where characteristic slope changes	% volumetric
$\frac{d}{ds}$	DS	distance between samples	cm
a	AW	intercept of log-log relationship between θ_v and ψ_w	% volumetric
b	BW	slope of log-log relationship between θ_v and ψ_w	dimensionless
θ_v	H20	volumetric soil moisture content	% volumetric
K_{UNSAT}	KW	unsaturated conductivity	cm/h
$\log \psi_m$	XPH20	log of matric potential	cm
ψ_m	PH20	matric potential	cm
$\frac{d \psi_m}{ds}$	DH20	matric potential gradient	cm/cm
$\frac{d \psi_h}{ds}$	(DH20(I) + 1)	hydraulic gradient	cm/cm
J_w	FH20	water flux density	cm/d
J_k	FK	flux density of K	meq cm ⁻³ d ⁻¹
$\frac{d \psi_{mnl}}{ds}$	DH2L(nl)	lateral water potential gradient	cm/cm
J_{wL}	FLH20	lateral water flux density	cm
J_{CaL}	FCAL	lateral Ca flux density	meq cm ⁻³ d ⁻¹
C_{K1}	XK	K concentration in soil solution	moles/l
C_{K2}	AK	K concentration of adsorbed ions	meq/100 g of soil (air dry)
C_{K3}	AZK	K concentration of adsorbed ions	meq/cm ³ of soil solution

<u>Mathematical</u>	<u>Computer</u>	<u>Description</u>	<u>Dimensions</u>
C_{K4}	ZK	K concentration in soil solution	meq/cm^3 of soil solution
J_{KE}	UK	flux density of K in soil ion uptake at soil surface	$\text{meq cm}^{-3} \text{ d}^{-1}$
E_t	FLUX	evapotranspiration	$\text{ml cm}^{-2} \text{ d}^{-1}$
EW_K	EK	equivalent weight of K	
V	---	valency	
FW	---	formula weight	
AW	---	atomic weight	
P_b	BD	bulk density	g/cm^3
D	DENIT	denitrification of nitrate	$\text{meq cm}^{-3} \text{ d}^{-1}$
R	RATIO, RAT	chloride-nitrate ratio	dimensionless
C_{N1R}	YNO3	nitrate concentration in reference sites	moles/l
C_{N1}	YNO3	nitrate concentration	moles/l
C_{Cl}	YCL2	chloride concentration	moles/l
J_{KL}	FKL	lateral K flux density	$\text{meq cm}^{-3} \text{ d}^{-1}$

APPENDIX B

METHODS OF DATA COLLECTION, FIELD AND LABORATORY RESEARCH

Most of the methods of data collection, field and laboratory research are reviewed in Campbell (1974) and Bunting and Campbell (1976). As was noted in Chapter III, describing methodology, two types of sampling sites were used--deep soil inspection pits and coring sites. Four deep soil inspection pits were used to describe soil profiles of the soils in the area (Tables B.1 to B.7 and Figs. B.1, B.2). Samples from the pits were used to determine stable soil properties, including particle density, grain size distribution, organic carbon, cation exchange capacity, specific surface area and hygroscopic moisture content. A non-standard rapid method described in Bunting and Campbell (1976) and Campbell (1974) was used to determine organic carbon. Particle size distribution was determined according to McKeague (1976). The soil was first oven dried ($105^{\circ}\text{C}.$), ground and sieved to < 2 mm, and pretreated to remove organic matter. Calcimeter analysis (Bunting and Campbell, 1976; Hesse, 1971) indicated that only the added gravel in the septic bed soil (pit 4, IIC) was calcareous (3.2%), so that carbonates were removed from this sample. Sonic sifting (Allen-Bradley Co., Milwaukee, Wisconsin) down to $37\ \mu$ was used to determine the sand distribution. The silt and clay distributions were determined by

/p.141

TABLE B.1 Profile #1

Vegetation: spruce, maple; nearby--elm, basswood, poplar, birch

Litter:	2.5-0 cm	10YR 2/3, organic material, pine needles, fine roots, some woody material, insect pupae
A1	0-4 cm	10YR 3/3 silt loam, fine crumb, few very fine subangular blocky peds; non-sticky when wet; many small and medium pores, many fine roots, gradual merging boundary
Ae	4-10 cm	10YR 4/2 silt loam, crumb; non-sticky when wet, moderate fine roots, few medium roots, many fine pores, few medium vughs, root channels, gradual merging boundary
B	10-23 cm	7.5YR 4/4 silt loam, weak medium subangular blocky, breaking to granular; non-sticky when wet; few medium roots, fine pores, irregular merging boundary
IC	23-43 cm	10YR 6/2 silty clay, massive, some weakly-developed fine platy structure, some iron concretions and mottles; sticky when wet, few fine pores, no roots, irregular abrupt boundary due to tonguing and inclusions
IIC	43-76 cm	7.5YR 4/6 silty clay, coarse subangular blocky; sticky when wet; many fine vughs and vesicles; no roots, mottles or lighter (7.5YR 8/2) concretions, platy structure in places, with planar voids and smooth cutanic coated surfaces; some of the platy structure is between IC and IIC; weak mottling along root channels

Note: The location of these profiles is shown in Figs. 3.2 and 3.3, pp. 33 and 35; the profiles are featured in Figs. 3.4 to 3.7, pp. 36-39.

TABLE B.2 Profile #2

Vegetation: mower-cut grass

Litter: virtually absent

A1	0-2.5 cm	10YR 4/3 silty loam, crumb structure, friable; very slightly sticky when wet; fine to medium thick grass forming a 'mat', fine fibrous roots, distinct boundary
Ae	2.5-18 cm	10YR 4/3 silty loam, weak medium subangular blocky structure breaking to fine to medium crumb; very slightly sticky when wet; few moderate to fine roots, few vughs and vesicles, irregular merging boundary
B	18-28/33 cm	10YR 4/4 to 10YR 5/4 silt loam, weak subangular blocky, breaking to granular; very slightly sticky when wet, moderate fine to medium roots, few vughs and vesicles, irregular distinct boundary, some inclusions of IC in B
IC	28/33-38 cm	10YR 6/2 clay with some silt inclusions, hard massive, very few roots, few fine pores, very few fine vughs and vesicles, channel linings along planar joints (7.5YR 4/4); incidence of roots and of vesicles of mesofaunal origin increase where horizon is thin; roots concentrated along vertical planar cracks, distinct irregular boundary
I-IIC	38-56 cm	10YR 6/2 (from IC), secondary colour 10YR 4/4 (from IIC) clay, hard, massive, some very large angular ped formations along vertical planar cracks; complex horizon consisting of degraded IC intermixed with IIC material in the form of inclusions; few roots along planar cracks, irregular distinct boundary
IIC	56-76 cm	5YR 4/3 clay, hard to medium subangular blocky with weakly-developed platy structure, light coating of cutanic deposits of IC on ped surfaces and joint planes

TABLE B.3 Profile #3 West

Septic Face

Vegetation: mower-cut grass

Litter: virtually absent

A1	0-2.5 cm	7.5YR 4/3 clay, medium to fine subangular blocky structure; very sticky when wet; secondary colour 10YR 7/2; horizon is a disturbed mixture of IC and IIC in the form of inclusions as well as finely mixed material, few medium to fine roots, regular distinct boundary
B1	20/30-38 cm	10YR 3/3 silt loam, weak medium subangular blocky breaking to medium to fine crumb; non-sticky when wet, few medium to fine roots, few vughs and vesicles, few inclusions of B2 in B1, distinct irregular boundary
B2	38-61 cm	7.5YR 4/4 fine sand, single grain, some silt coatings and linings, few medium roots, few fine roots, very few vughs and vesicles, linings of B1 along medium and large root channels, merging boundary
C	61-76 cm	10YR 6/2 clay, weak platy structure, few roots, vughs and vesicles, grading to subangular blocky structure with clay cutans on ped surfaces, more mottled cutans are 7.5YR 4/4

Note: 0-30 cm of profile was very dry (Ap),
 30-61 cm moderately moist (B1 and B2),
 61-66 cm moderately moist (upper C),
 66-76 cm much drier than upper C (lower C),
 moisture is from septic effluent.

TABLE B.4 Profile #3 East

East Face

Vegetation: mower-cut grass

Litter: virtually absent

A1	0-2.5 cm	10YR 6/2 silt loam, single grain, thick root mat of fine and medium roots, irregular distinct boundary
A2	2.5-11 cm	10YR 4/3 silt loam, weak subangular blocky, moderate medium to fine roots, merging boundary
B1	11-18 cm	10YR 4/3 silt loam, weak subangular blocky, few fine roots, distinct irregular boundary
B2	18-30 cm	10YR 5/6 silty sand, single grain, few vughs and vesicles, few medium roots, irregular merging boundary
C	30-46 cm	10YR 5/2 clay, medium subangular blocky, very few roots, cutanic coatings on ped surfaces 7.5YR 4/4

TABLE B.5 Profile #3 North

North Face

Vegetation: mower-cut grass

Litter: virtually absent

(a) West Half

Ap	0-8 cm	10YR 4/3 silt loam, crumb, many fine to medium roots, merging boundary
B1	8-18 cm	10YR 4/3 silt loam, moderate medium subangular blocky, few roots, few vughs and vesicles, little staining, gradual merging boundary
B2	18-38 cm	7.5YR 4/4 fine sand, single grain, few roots, vughs and vesicles, little staining, clear distinct boundary
C	38-64 cm	10YR 5/2 clay, medium subangular blocky, very few roots, cutanic coatings on ped surfaces 7.5YR 4/4, some mottles

(b) East Half

Ap	0-26 cm	7.5YR 4/3 clay, medium to fine subangular blocky, see Ap of west face for more detail, mixture of IC and IIC, clear regular distinct boundary
B	26-66 cm	7.5YR 4/4 fine sand, single grain, few roots, no vughs, vesicles or lining, clear distinct boundary
C	66 cm +	10YR 5/2 clay, medium subangular blocky, very few roots, cutanic coatings on ped surfaces 7.5YR 4/4, some mottles

TABLE B.6 Profile #3 South

South Face

Vegetation: mower-cut grass

Litter: virtually absent

(a) East Half

Ap	0-8 cm	10YR 4/3 silt loam, crumb, many fine to medium roots, merging boundary
B1	8-18 cm	10YR 4/3 silt loam, weak medium subangular blocky breaking to crumb, moderate fine roots, moderate vesicles, vughs and worm channels, organic stainings along large root channels, gradual merging boundary
C	36 cm +	10YR 6/2 clay, hard subangular, no visible cutans or mottles, fairly dry

(b) West Half--the same as the West (Septic) Face--note depths on diagram

TABLE B.7 Profile #4

Vegetation: mower-cut grass

Litter:	0.6-0 cm	Layer 1: cut grass and dead twigs Layer 2: black humified twigs and grass mixed with some coarse sand
A1	0-2.5/5 cm	10YR 3/4 silt loam, very fine crumb, many fine to medium roots; non-sticky when wet; irregular distinct boundary
Ap	2.5/5-18 cm	7.5YR 4/6 sandy silt loam; very fine crumb to single grain, many medium and fine roots; non-sticky when wet; gradual merging boundary
B2	18-26 cm	10YR 3/4 silt loam, very fine crumb, many fine and medium roots; non-sticky when wet; distinct irregular boundary
IC	26-38 cm	10YR 5/2 clay, medium subangular blocky, 7.5YR 4/6 coatings on ped faces; very sticky when wet; few coarse roots, distinct irregular boundary
IIC	38-61 cm	10GY 2/1 fine gravel, 'septic' gravel, septic odour, effluent-saturated, black organic coatings on gravel surfaces, no roots, few vughs, few vesicles, some mesofauna; lower boundary not determined, horizon is intermittent

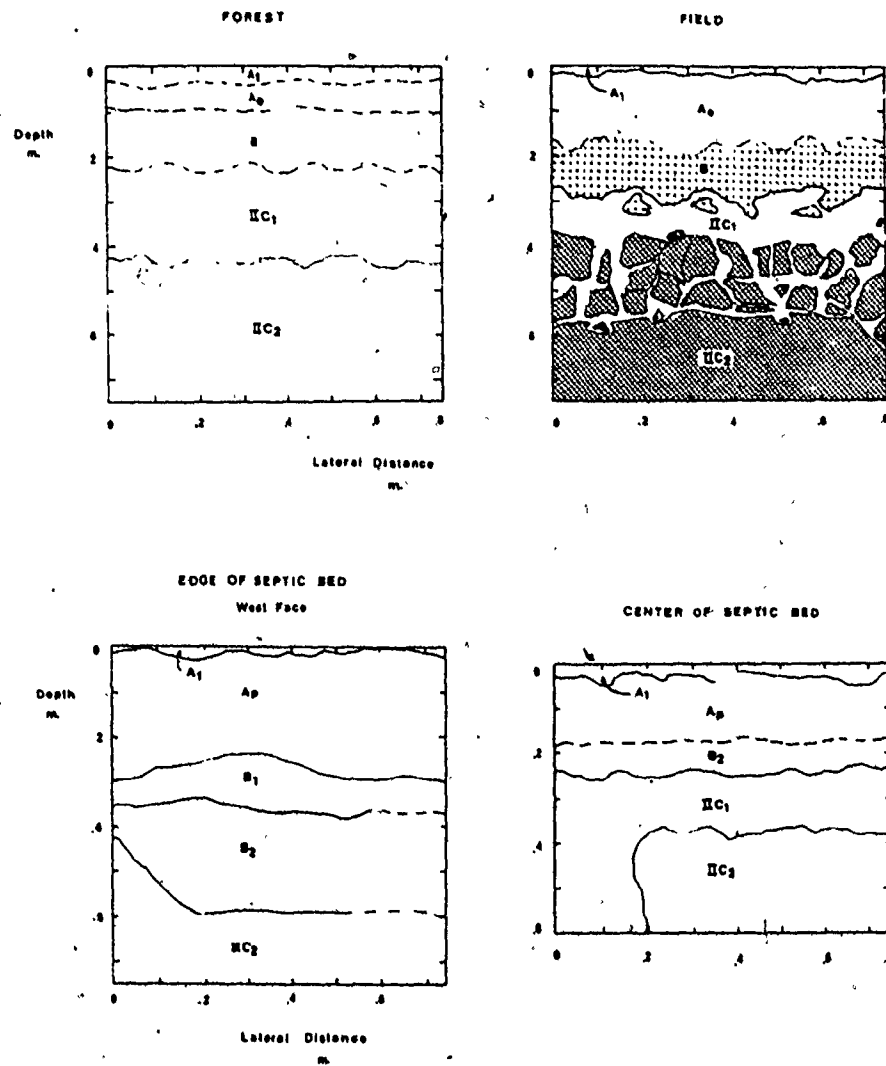


FIG. B.1 Soil horizon boundaries of selected field, forest and septic bed profiles

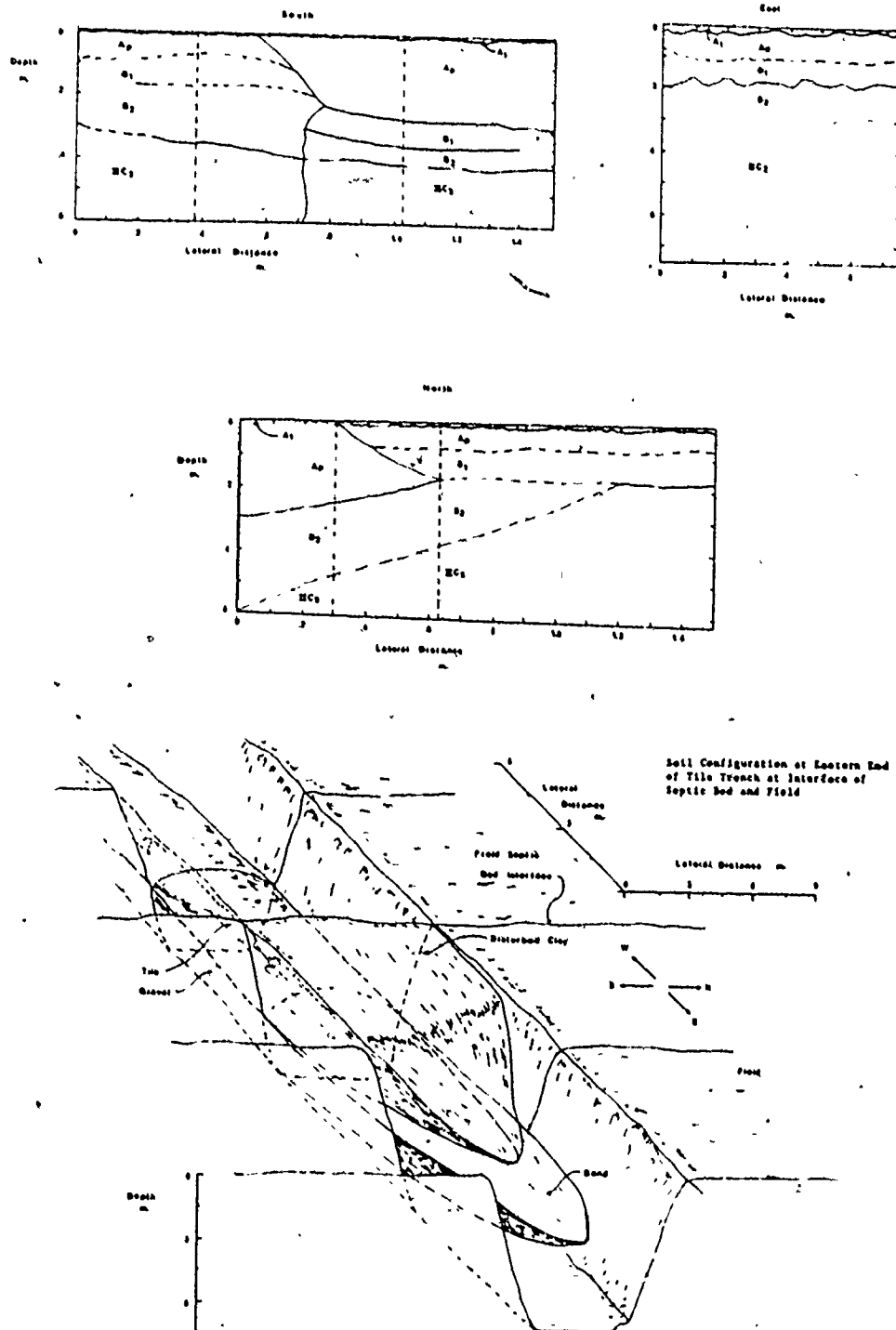


FIG. B.2 Soil horizon boundaries at the margins of the tile trench and field

pipette analysis on separate samples.

Cation exchange capacity was determined according to Bunting and Campbell (1976) and Metson (1961), using the method for non-calcareous soils except for pit 4, IIC, where the method for 0-3% carbonates was used. Specific surface area was determined by glycerol retention, according to Soil Conservation Service, U. S. Dept. of Agriculture (1967). Hygroscopic moisture content was calculated as the difference between air dry and oven dry (105°C.), soil moisture contents are expressed as percentage by weight of oven dry sample (Bunting and Campbell, 1976).

Large, undisturbed, cores (2.5 cm length, 4.8 cm I.D.) from soil pits were used to determine soil phase composition, bulk density, saturated conductivity (McKeague, 1976) and the desorption part of the characteristic curve (Soil Moisture Equipment Corp., Santa Barbara, California), as illustrated in Fig. B.3, for the field soil horizons. From these measurements, unsaturated conductivity was calculated (G. Campbell, 1974), as explained in Chapter III, section 3.4.

New or non-standard techniques were used to measure most transient soil properties. This includes the design of continuous monitoring equipment installed in the deep soil inspection pits and the analysis of coring sites for soil solute ionic levels, oxygen diffusion rate and measurement of evapotranspiration from small lysimeters. Soil moisture was determined from cored samples or by the use of in situ moisture blocks. Cottage water use was also monitored and is graphed in Fig. B.4. The measurement of soil ionic levels and oxygen diffusion rate is reviewed in Bunting and Campbell (1976) and

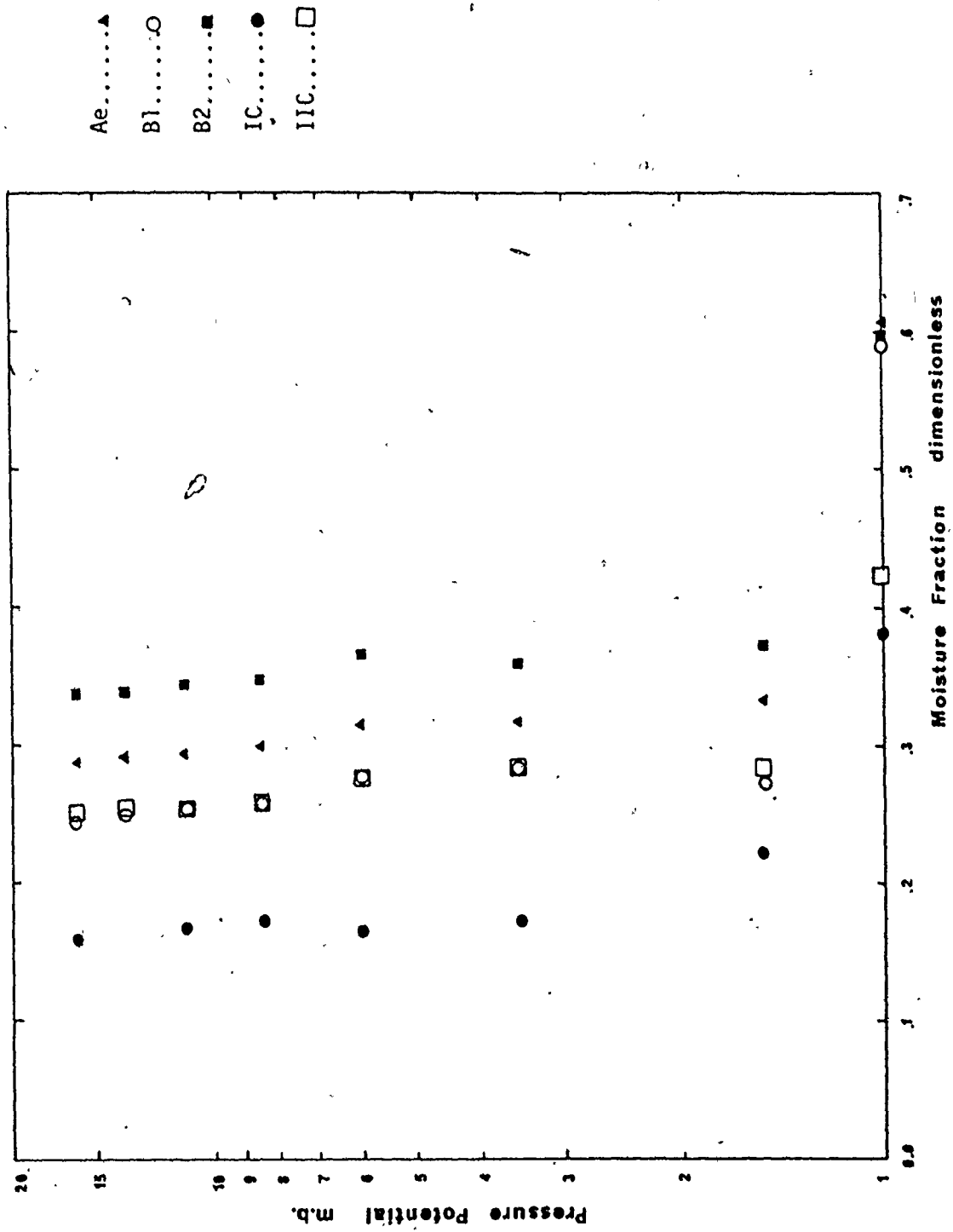


FIG. B.3 Characteristic desorption curves for the field soil

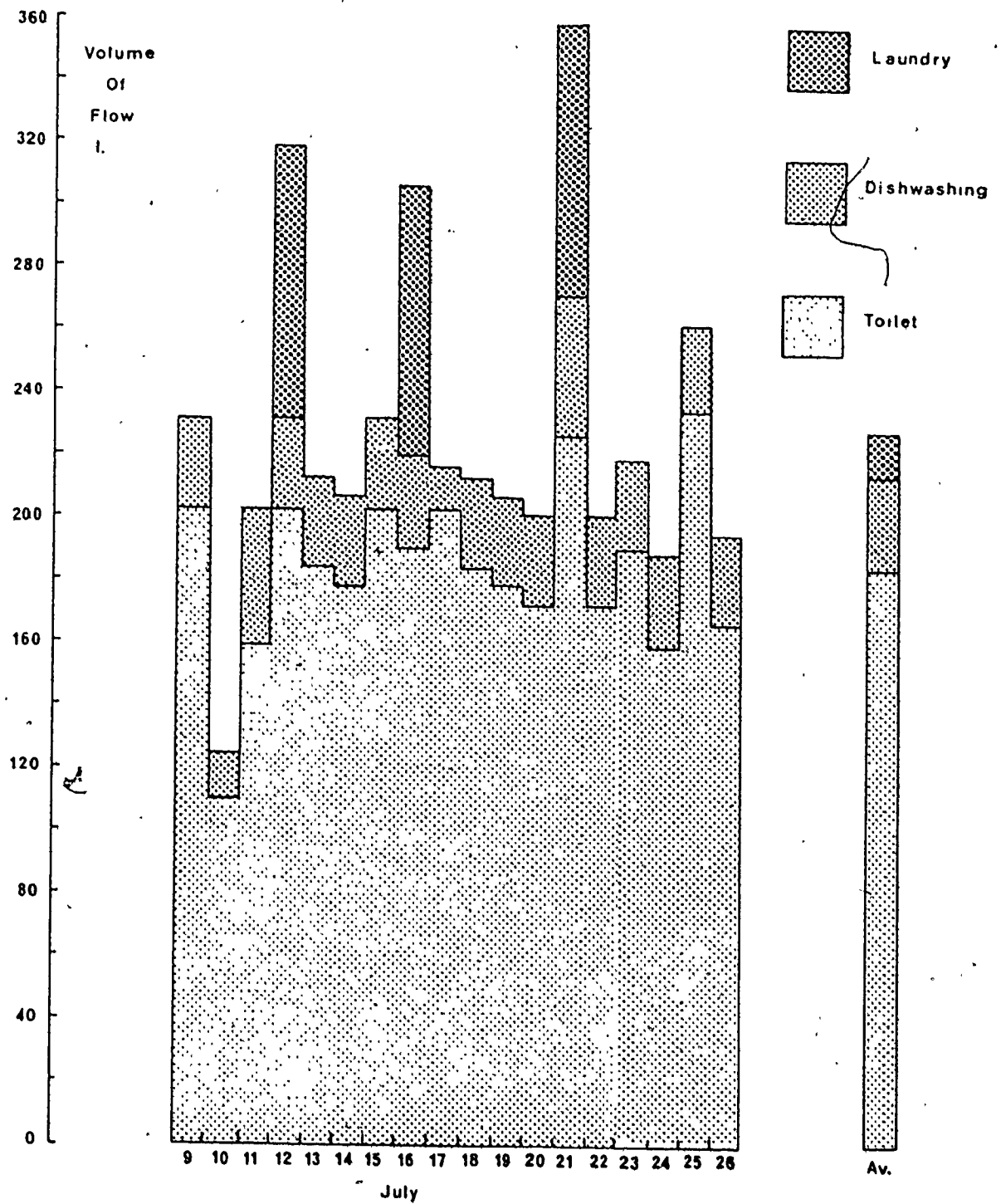


FIG. B.4 Cottage water use during July 1976

6 Campbell (1974), including the application of the specific ion electrode designed by Nagelberg et al. (1969) to soil ionic phosphate analysis.

Included here is a brief review of the design of the continuous monitoring equipment including solution wells, gypsum moisture blocks, temperature sensors and lysimeters. Solution wells consisted of porous ceramic agricultural cups (5 cm length, 2 cm O.D.) cemented by epoxy to polycarbonate tubing 1 m in length. The wells were sampled with a portable battery-operated peristaltic sampling pump (Horizon Ecology Co., Chicago, Illinois). Rectangular gypsum moisture blocks were constructed, 4.0 cm x 2.5 cm x 2.9 cm high, with solder-coated electrodes 2.5 cm apart. Exposed wire was coated with epoxy. To test their in situ longevity, moisture blocks were installed in a wet, effluent-receiving site in the field soil during the summer of 1974 and removed the following summer of 1975. During this time, about 2 mm of gypsum was lost from the block surface by solution. The temperature sensors at each site consisted of thermistors (Phillips, NTC) coated with varnish, arranged at intervals along a polycarbonate tube and epoxied in place. These thermistor posts had a length of one meter and one was installed at each inspection pit. The thermistors were calibrated against a mercury thermometer in the laboratory (Fig. 8.5). Three small lysimeters were constructed from metal tins (10 cm length x 13 cm diameter), containing 1330 cm³ of soil and 600 to 900 g of air dry soil, with a screened drain in the bottom. An undisturbed core of soil, the exact size of the lysimeter, was sampled and gently inserted into it. Lysimeters were installed at inspection pits 2, 3 and 4 and weighed daily on an electronic balance to 0.1 g, during the research

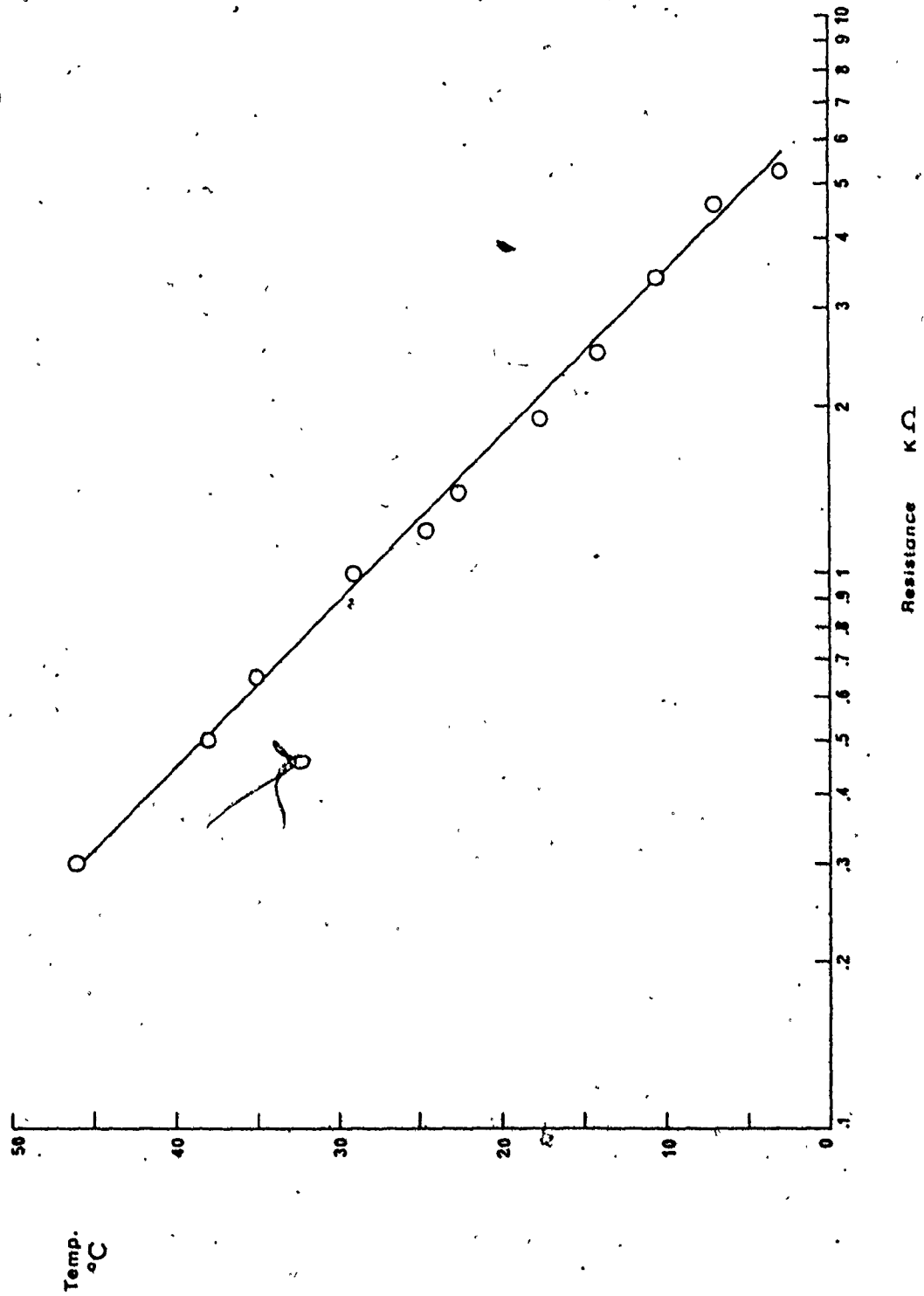


FIG. B.5 Calibration curve for thermistors used for measurement of soil temperature in situ

period of summer 1976. The moisture blocks, solution wells and thermistor posts were installed in the summer of 1975 and monitored the following summer, thus allowing the moisture blocks to equilibrate and the soil to settle. A portable battery-operated multimeter was used for resistance measurements of moisture blocks and thermistors. Moisture blocks were calibrated in situ during the field survey in 1976 by comparing resistance measurements with simultaneous determinations of gravimetric soil moisture from cored samples. 8

The continuous monitoring equipment installed in the deep soil inspection pit in the forest is shown in Fig. B.6. The use of a polarographic electrode to measure oxygen diffusion rate in situ in the field soil during the spatial sampling survey in 1975 is shown in Fig. B.7. The use of the Orion (Orion Research Incorporated) field meter in the cottage laboratory during the same survey to measure soil ionic levels from freshly cored soil samples is illustrated in Fig. B.8.

Multivariate techniques used to interpret measured soil properties included multiple regression, canonical correlation and common factor analysis with oblique rotation from the SPSS program library (Nie et al., 1975). The very large number of soil samples, compared with the small number of measured properties, resulted in a sample to property ratio, or case to variable ratio, of ten to one or larger. This suggests that the relationships produced by these techniques are reliable. Reported regression equations and factor solutions are significant at the 5% level. The forward stepwise procedure was used to calculate multiple regression equations. For the factor analyses,



FIG. B.6 Continuous monitoring equipment installed in the deep soil inspection pit in the forest

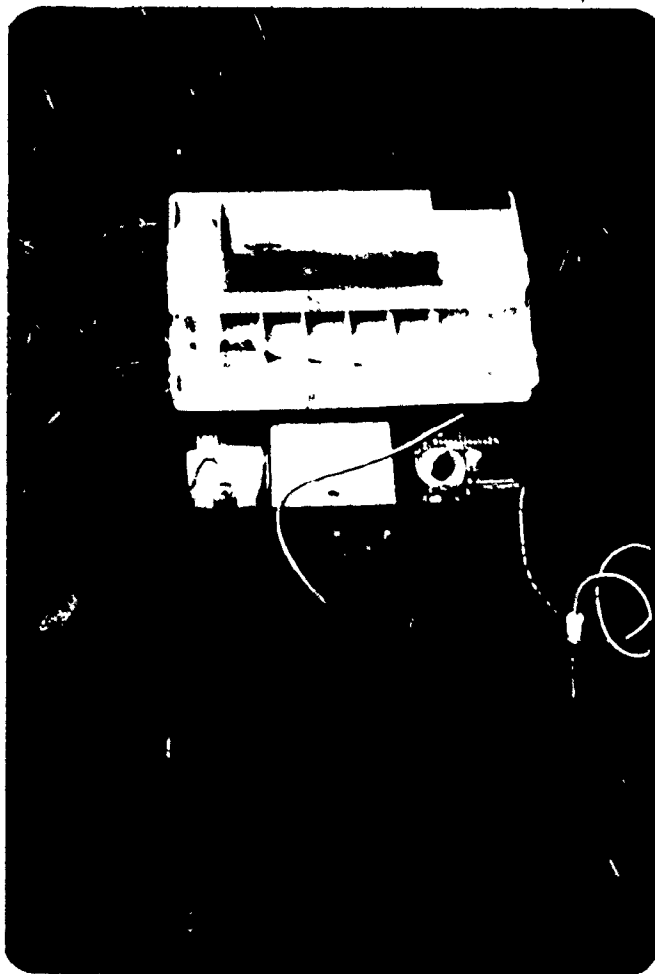


FIG. B.7 Polarographic oxygen electrode in use in the field soil



FIG. B.3 Measurement of soil ionic levels using the Orion field meter

a delta value of zero, which gives a moderately correlated solution, was used with oblique rotation. The multiple squared correlation coefficient was used as the initial communality estimate. Only factors with eigenvalues greater than 1.0 are reported.

APPENDIX C

MASS BALANCE CALCULATIONS

C.1 Translation of Equations into Fortran Statements

- a. Ion Exchange Calculations
- b. Moisture and Nutrient Fluxes
- c. Plant Uptake of Soil Ions
- d. Denitrification
- e. Conversions to Concentrations in meq/cm^3

C.2 Organization of Fortran Programs

- a. The Fortran Program BALAN1
- b. The Fortran Program BALAN2
- c. The Fortran Program BALAN3

C.3 BALAN1, BALAN2, BALAN3

APPENDIX C

MASS BALANCE CALCULATIONS

The translation of mathematical formulas into Fortran is presented. In section C.1, mathematical formulas are translated into Fortran statements. The computer statements are from the program BALAN1. These statements are very similar to those in BALAN2 and BALAN3. Following this, the organization of these statements into programs is discussed in section C.2. The actual programs are listed in section C.3. The terms used in this discussion are included in Appendix A.

C.1 Translation of Equations into Fortran Statements

a. Ion Exchange Calculations

The adsorbed cations are calculated using the ion exchange model in part of a DO loop extending from lines 163 to 171 in BALAN1.

First the a_1 , b_1 , c and z parameters are calculated in lines 163, 164 and 165, respectively. These correspond to a_1 , b_1 and c in equations (10) and (11) and z in equation (7).

Thus from (10) and (11)

$$a_1 = \frac{K_d C_{CaS}}{C_{KS}^2}$$

and in BALAN1

$$X(I) = (YCA(I)*XKD1(I))/(YK(I)**2.)$$

line
163

From (7)

$$C_{NaA} + C_{KA} + C_{CaA} = z$$

and in BALAN1

$$Z(I) = BCA(I) + BK(I) + BNA(I)$$

line
164

From (10) and (11)

$$b_1 = 1 + \frac{C_{NaS}}{K_m C_{KS}}$$

and in BALAN1

$$Y(I) = (1. + YNA(I))/(XKMI(1)*YK(I))$$

line
165

The next four statements in BALAN1 are used to compute the adsorbed

K from a_1 and b_1 according to (12).

From (12)

$$C_{KA} = \frac{-y_1 \pm \sqrt{y_1^2 - 4x_1(z)}}{2x_1} \quad (12b)$$

The value under the square root sign is computed first.

In BALAN1

$$CPX(I) = (Y(I)**2. - (4.0*X(I)*(-Z(I))))$$

line
166

$$CPX(I) = SQRT(CPX(I))$$

line
167

$$CPX(I) = ABS(CPX(I))$$

line
168

The value under the square root sign is then substituted into (12b) to compute adsorbed K.

In BALAN1

$$CBK(I) = (-Y(I) + CPX(I))/(2.0*X(I))$$

line
169

From (2c), C_{NaA} can be calculated

$$C_{NaA} = \frac{K_m \times C_{KA} \times C_{NaS}}{C_{KS}}$$

In BALAN1

$$CBNA(I) = (XKM(I) * CBK(I) * YNA(I)) / YK(I) \quad \text{line 170}$$

From (7), C_{CaA} can now be calculated

$$C_{NaA} + C_{KA} + C_{CaA} = Z$$

and in BALAN1

$$CBCA(I) = Z(I) - (CBNA(I) + CBK(I)) \quad \text{line 171}$$

The equilibrium models were tested in various ways. Multiple correlation coefficients derived from multiple regression analysis provide a test of this empirical approach using equation (13). Multiple R values did not exceed 0.75, suggesting that this method was less than ideal. The analytical model using equation (12) was tested by comparing known levels of adsorbed soil ions with those computed by the model in the program BALAN1. The equations, including equation (12), were checked to see that they were dimensionally balanced.

Equilibrium constants used in the equilibrium model are calculated as follows. In parts (i) and (ii), measurements of adsorbed ions and ions in solution are converted to similar form--concentrations in meq/cm³ of soil solution. In part (iii), equilibrium constants are calculated using these concentrations.

- (i) Formula to calculate soil ion concentration in soil solution
in meq/cm³ of soil solution from concentration in moles/l

$$C_{K4} = \frac{C_{KI}}{1000} \times \frac{EW_K}{1000}$$

In BALAN1 this is:

$$2K(I) = (XK(I)/1000.)*(EK/1000.)$$

line
136

- (ii) Formula to calculate concentration of adsorbed ions in
meq/cm³ of soil solution from meq/100 g soil

$$C_{K3} = \frac{C_{K2}}{100} \times \frac{1}{P_b} \times \frac{100}{\theta_v}$$

In BALAN1 this is:

$$AZK(I) = (AK(I)/100.)*(1./BD(I))*(100./H2O(I))$$

line
139

- (iii) Formulas to calculate equilibrium constants

From the Law of Mass Action, adsorbed cations are calculated from cations in solution using two equations first used by Kerr (1928), reviewed in Marshall (1964) and used by de Wit and van Keulen (1972).

For a monovalent-monovalent system,

$$\frac{\text{cation 1 adsorbed}}{\text{cation 2 adsorbed}} = K_m \times \frac{\text{cation 1 in solution}}{\text{cation 2 in solution}}$$

From experimental data, K_m can be calculated

$$\frac{C_{Na3}}{C_{K3}} = K_m \times \frac{C_{Na4}}{C_{K4}}$$

In BALAN1 this is:

$$XKM1(I) = (ZK(I) * AZNA(I)) / ZNA(I) * AZK(I)) \quad \text{line 142}$$

For a divalent-monovalent system

$$\frac{\text{divalent cation in solution}}{\text{divalent cation adsorbed}} = K_d \left\{ \frac{\text{monovalent cation in solution}}{\text{monovalent cation adsorbed}} \right\}^2$$

From experimental data, K_d can be calculated

$$\frac{C_{Ca3}}{C_{Ca4}} = K_d \left\{ \frac{C_{K3}}{C_{K4}} \right\}^2$$

In BALAN1 this is:

$$XKD1(I) = ((AZK(I) / ZK(I)) ** 2) * (2CA(I) / AZCA(I)) \quad \text{line 143}$$

b. Moisture and Nutrient Fluxes

Vertical and lateral water and nutrient fluxes are calculated separately.

(i) Vertical Fluxes

In BALAN1, vertical water and nutrient fluxes are calculated by four DO loops. The first loop calculates conductivity. The second loop calculates matric potential. The third loop calculates the potential hydraulic gradient and the fourth loop calculates the water flow and then the nutrient fluxes in the water.

1st Loop--Conductivity

The moisture content is tested to see if it is in the linear range of the characteristic curve. If it is not, conductivity is set to saturated conductivity. If the moisture content is in the linear

range of the characteristic curve, conductivity is calculated from the following equation (G. Campbell, 1974):

$$K_{\text{UNSAT}} = K_{\text{SAT}} \left\{ \frac{\theta_v}{\theta_{v\text{SAT}}} \right\}^{(2b + 3)}$$

or

$$KW(I) = KSAT(I) * (H2O(I) / H2SAT(I)) ** ((2 * B(I)) + 3.))$$

line
197

2nd Loop--Matric Potential

A minimum value of soil moisture content is set at 2%. Matric potential is computed in a two-step process. First the log of the matric potential is computed from the soil moisture content and the regression equation between the log of soil moisture and the log of matric potential. Then the matric potential is calculated from the log of the matric potential.

$$\log \psi_m = a (\log \theta_v + b)$$

$$\psi_m = 10^{\log \psi_w}$$

$$XPH2O(I) = AW(I) * (ALOG10(H2O(I)) + BW(I))$$

$$PH2O(I) = 10. ** (XPH2O(I))$$

line
203
line
204

¹To convert matric potential (ψ_w) from bars to cm, the values determined on the pressure membrane apparatus were multiplied by 10³, yielding a 2.2% error since the actual conversion is 1022 (Rose, 1966).

3rd Loop--Potential Gradient

The potential gradient is calculated by subtracting the matric potential of each soil horizon from the horizon above it and dividing by the thickness of the lower horizon.²

$$\frac{d\psi_m}{ds} = \frac{\psi_{ma} - \psi_{mb}}{ds}$$

where $\frac{d\psi_m}{ds}$ is the matric potential gradient (cm/cm)

ds is the thickness of the lower horizon (cm)

ψ_{ma} is the matric potential of the upper horizon (cm)

ψ_{mb} is the matric potential of the lower horizon (cm)

$$DH20(I) = ((PH20(I) - PH20(IX))/DS(I))$$

line
213

4th Loop--Water and Nutrient Flux

Water flows are computed first and then nutrient fluxes are computed as the product of the water flow and ionic concentration. By

²The sample locations for the data in BALAN1 were based on the concept of the soil profile where the soil is assumed to consist of a series of layers of isotropic soil materials. Within the profile concept, therefore, gradients or changes in properties occur at horizon boundaries. The change in moisture content ($d\theta_v$) is the same whether computed over the thickness of the boundary (resulting in a large gradient) or over the thickness of the whole of the lower horizon (resulting in a small gradient). The latter approach was adopted as more conservative and realistic.

The sampling locations for the data in BALAN2 and BALAN3 were based on a regular depth interval sampling design, and the horizon boundaries did not affect the sampling design. In these instances, the actual distances between sample locations were used to compute matric potential gradients.

computing the hydraulic gradient over a distance of 1 cm, it is equal to the sum of the matric potential gradient and the gravity potential gradient which is 1.

$$J_w = -K_w \frac{d\psi_h}{ds}$$

$$\psi_h = \psi_w + \psi_z$$

Assuming $\psi_w = \psi_m$

$$\psi_h = \psi_m + \psi_z$$

Computing ψ_h over 1 cm, $\frac{d\psi_z}{ds} = 1$

$$\frac{d\psi_h}{ds} = \frac{d\psi_m}{ds} + 1$$

$$J_w = -K_w \left\{ \frac{d\psi_m}{ds} + 1 \right\}$$

$$FH20(I) = -KW(I) * (DH20(I) + 1) * 24.$$

line
221

For soil ion transport

$$J_k = J_w \times C_{KS}$$

$$FK = FH20(I) * YK(I)$$

line
222

(ii) Lateral Fluxes

Lateral fluxes are computed in two steps. First a series of statements compute the matrix potential gradient in the horizontal direction between two points, 183 cm apart, in the soil trench at the edge of the septic bed (soil pit no. 3). A DO loop calculates the

lateral water and nutrient fluxes in the second step.

1st Step--Potential Gradient

$$\frac{d\psi_{mnL}}{ds} = \frac{\psi_{mn1} - \psi_{mn2}}{ds}$$

where ψ_{mn1} is the matric potential at the edge of the septic bed

ψ_{mn2} is the matric potential, 183 cm from the edge of the septic bed

$$DH2L(n_1) = (PH20(n_1) - PH20(n_2))/182.88$$

line
236

2nd Step--Lateral Water and Nutrient Flux

$$J_{wL} = K_w \frac{d\psi_m}{ds}$$

$$FLH20(I) = KW(I) * DH2L(I) * 24.$$

line
246

For lateral ionic flux

$$J_{CaL} = J_{wL} \times C_{CaS}$$

$$FCAL(I) = FLH20(I) * YCA(I)$$

line
247

(iii) Comparisons of Moisture Flux Equations with Equations in the Literature

The moisture flux equations can also be derived or related to those of de Wit and van Keulen (1972) and of Taylor and Ashcroft (1972). From de Wit and van Keulen (p. 85) and Ashcroft and Taylor (p. 198)

$$D_w = K_w \left\{ \frac{d\psi_m}{d\theta_v} \right\} \quad (1)$$

de Wit and van Keulen then present the flow equation as

$$\text{FLOW} = D_w \left\{ \frac{d\theta_v}{ds} \right\} + K_w \{ \text{GRAV} \} \quad (2)$$

Substituting (1) in (2) yields (3)

$$\text{FLOW} = K_w \left\{ \frac{d\psi_m}{d\theta_v} \right\} \left\{ \frac{d\theta_v}{ds} \right\} + K_w \{ \text{GRAV} \} \quad (3)$$

For vertical flux, $\text{GRAV} = +1$ and for horizontal flux, $\text{GRAV} = 0$.

For vertical flux,

$$\text{FLOW} = K_w \left\{ \frac{d\psi_m}{ds} + 1 \right\}^3$$

and for horizontal flux,

$$\text{FLOW} = K_w \left\{ \frac{d\psi_m}{ds} \right\}$$

From Taylor and Ashcroft (p. 287)⁴

$$K_w = - \frac{J_w}{\left\{ \frac{d\psi_m}{ds} \right\} + 1}$$

Rearranging:

$$J_w = -K_w \left\{ \frac{d\psi_m}{ds} + 1 \right\}$$

³In de Wit and van Keulen's equation, positive vertical fluxes would be flowing down the profile and negative vertical fluxes would be flowing up the profile. This is due to the positive sign in front of K_w .

⁴In Taylor and Ashcroft's equation and the one used in the mass balance models, negative vertical fluxes would be flowing down the profile and positive vertical fluxes would be flowing up the profile. Thus the sign in front of K_w is negative.

c. Plant Uptake of Soil Ions

Soil ion uptake in $\text{meq cm}^3 \text{ d}^{-1}$ is calculated from ions in soil solution and evapotranspiration as follows:

$$J_{KE} = \left\{ \frac{C_{KI}}{1000} \right\} \left\{ E_t \right\} \left\{ \frac{EW_K}{1000} \right\}$$

In BALAN1 this is:

$$UK(I) = XK(I) * FLUX * (EK/1000.)$$

line
124

d. Denitrification

Denitrification is estimated from the Cl_2/NO_3 ratios in the reference areas.

$$R = \frac{C_{Cl_R}}{C_{Nl_R}} \quad \text{and} \quad D = C_{Nl} - \frac{C_{Cl}}{R}$$

In BALAN1:

$$RATIO(I) = YCL2(I)/YNO3(I)$$

line
151

From the chloride-nitrate ratios (RATIO), it was found that the average ratio in the reference soils, in the absence of obvious plant uptake, was 0.1119. Thus RAT was set to 0.1119.

$$RAT = 0.1119$$

line
153
line
154

$$DENIT(I) = YNO3(I) - (YCL2(I)/RAT)$$

e. Conversions to Concentrations in meq/cm^3

Equivalent weight, for an ion in solution, is determined from the atomic weight and valency as follows:

$$EW = FW/V \quad \text{and} \quad FW = \sum AW$$

$$EW = \frac{\sum AW}{V}$$

Thus:

<u>Ion</u>	<u>EW</u>	<u>AW</u>	<u>V</u>
Ca	20	40	2 ⁺
Mg	12.16	24.3	2 ⁺
Al	9	27	3 ⁺
K	39	39	1 ⁺
NO ₃	62	62	1 ⁻
NH ₄	18	18	1 ⁺
PO ₄	47.5	95	2 ⁻ 5
Na	23	23	1 ⁺
Cl	35.45	35.45	1 ⁻

Concentration of soil ion in solution in meq/cm³ is calculated from the concentration in moles/l as follows:

$$C_{KS} = \left\{ \frac{\theta_v}{100} \right\} \left\{ \frac{C_{K1}}{1000} \right\} \left\{ \frac{EW_K}{1000} \right\}$$

In BALAN1 this is:

$$YK(I) = (H2O(I)/100.) * (XK(I)/100.) * (EK/1000.)$$

line
87

The concentration of adsorbed ions in meq/cm³ of soil is calculated from concentration in meq/100 g of soil as follows:

$$C_{KA} = \left\{ \frac{C_{K2}}{100} \right\} \left\{ \frac{1}{P_b} \right\}$$

⁵The soil ion species measured by the specific ion electrode used.

In BALAN1 this is:

$$BK(I) = (AK(I)/100.)*(1./BD(I))$$

line
87

C.2 Organization of Fortran Programs

The first program BALAN1 calculates values, including vertical fluxes of water and soil ions in solution, when the effluent flow is continuous. The second program BALAN2 calculates values for soil properties as they respond to the commencement of effluent flow. The third program BALAN3 calculates some of the lateral fluxes out of the septic bed when the effluent flow is continuous.

a. The Fortran Program BALAN1

The program BALAN1 provides information on five sites located in the forest, field and septic bed soils at one point in time. A flow chart indicating the major steps in the program is included in Table C.1 and in Fig. C.1. The program is divided into five steps. In step 1, the concentration of ions in solution, in meq/cm^3 , is calculated from the concentrations of ions in solution in moles/l, and from volumetric soil moisture content and the bulk density. In step 2, plant uptake of soil ions is estimated from evapotranspiration. Adsorbed cations in meq/cm^3 are calculated from adsorbed cations in meq/100 g in the same step.

Adsorbed cations are predicted from cations in the soil solution in steps 3 and 4. In step 3, equilibrium constants are calculated from both adsorbed cations and cations in solution and in step 4, the technique is tested by comparing calculated values with measured values.

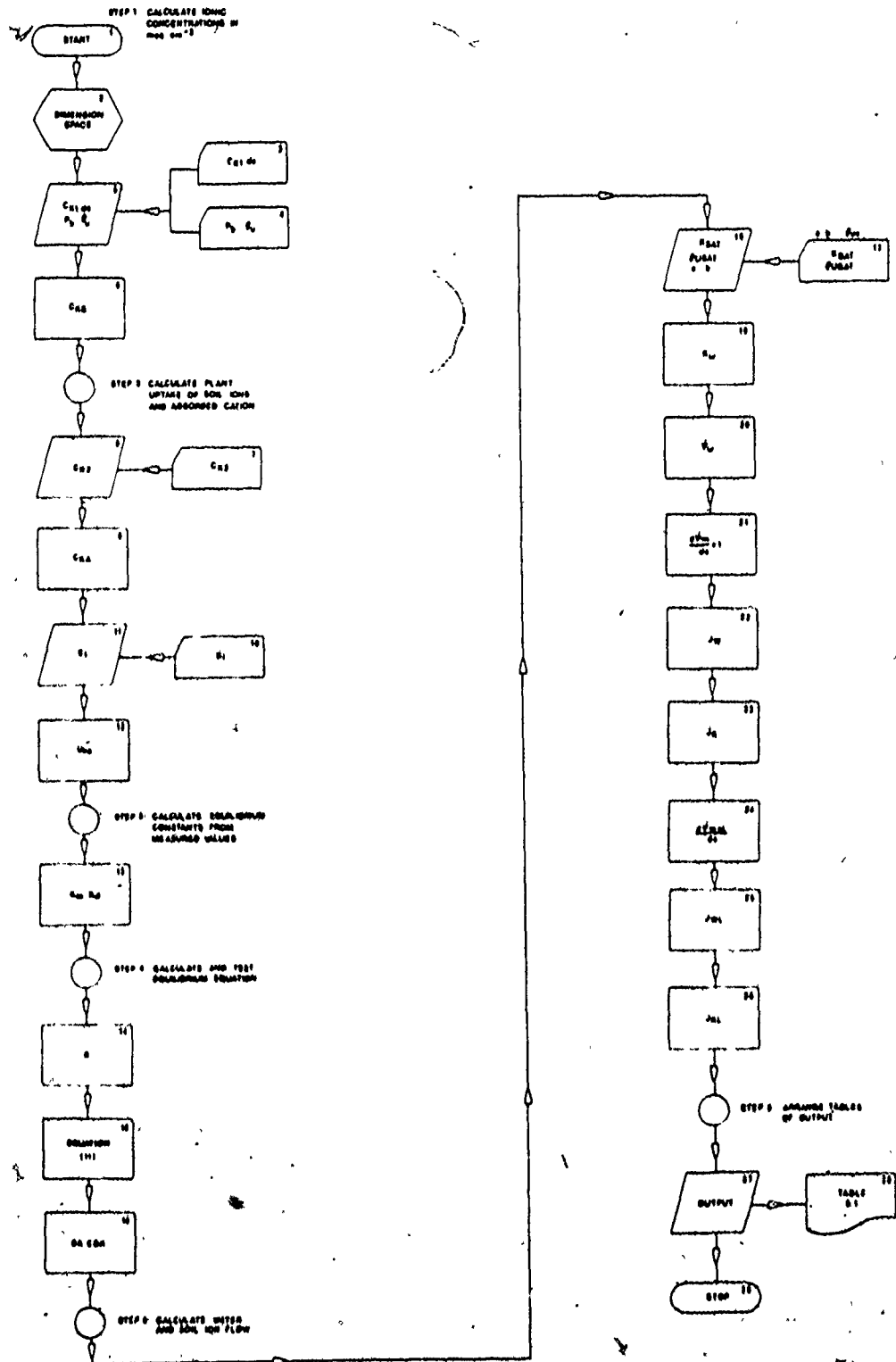


FIG. C.1 Flow chart for the computer program BALANT used to calculate soil water and ionic fluxes

TABLE C.1a -Selected Variables in BALANI Flowchart

Mathematical Symbol	Computer Symbol	Description	Units
C_{K1}, C_{KS}	XX, YK [*]	ionic concentration in solution	moles/l, meq/cm ³
ds	KD	soil sample depth	cm
P_b	BD	bulk density	g/cm ³
θ_v	H2O	volumetric soil moisture	%
	AK, BK [*]	adsorbed cations measured	moles/100 g, meq/cm ³
E_t	FLUX	evapotranspiration	ml cm ⁻² d ⁻¹
J_{KE}	UCA	ionic uptake of Ca	meq cm ⁻³ d ⁻¹
K_m, K_{d1}, K_{d2}	XKM1, XKD1, XKD2 ^{**}	equilibrium constants	dimensionless
	CBK	computed adsorbed cations	meq/cm ³
K_{SAT}	KSAT	saturation conductivity	cm/h
θ_{vSAT}	H2SAT	moisture content at saturation	%
a, b	AW, BW	parameters for log-log equation between θ_v and ψ_p	dimensionless
θ_{vx}	XB	moisture content above which characteristic curve is not linear	%
K_w	KH	actual conductivity	cm/h

Symbol	Mathematical	Computer	Description	Units
ψ_w		PH20	water potential	cm
$\frac{d\psi_m}{ds} + 1$		DH20	vertical hydraulic potential	cm
J_w		FH20	vertical water flow	cm/d
J_k		FK*	vertical ionic flux	meq cm ⁻³ d ⁻¹
$\frac{d\psi_m}{ds}$		DH2L	lateral hydraulic potential	cm
J_{wL}		FLH20	lateral water flow	cm/d
J_{kL}		FKL*	lateral ionic flux	meq cm ⁻³ d ⁻¹
		CEC	cation exchange capacity	meq/100 g

* shown only for potassium (K)

** XKM1 is monovalent K to Na
 XKD1 is divalent K to Ca
 XKD2 is divalent Na to Ca

TABLE C.1b List of Printed Output from Mass Balance Program BALAN1

1. ionic concentrations in solution, meq/cm^3
2. adsorbed cations (measured), meq/cm^3
3. ionic uptake, $\text{meq cm}^{-3} \text{ d}^{-1}$
4. equilibrium constants, dimensionless
5. chloride to nitrate ratios, dimensionless
6. denitrification, $\text{meq cm}^{-3} \text{ d}^{-1}$
7. right and left side for equilibrium equation
8. x and y parameters in equilibrium equation
9. calculated adsorbed cations, meq/cm^3
10. absolute and percentage error in calculated adsorbed cations, %
11. vertical soil water flows, cm/d
12. vertical ionic fluxes, $\text{meq cm}^{-3} \text{ d}^{-1}$
13. lateral soil water flows at edge of septic bed, cm/d
14. lateral ionic fluxes at edge of septic bed, $\text{meq cm}^{-3} \text{ d}^{-1}$

Chloride to nitrate ratios and denitrification rates are also calculated in step 4.

Fluxes of soil water and of individual ions are calculated in the fifth step. First, measurements describing the soil characteristic desorption curve are read in. These include soil moisture content and conductivity at saturation and parameters for the log-log equation between moisture content and pressure potential. From this, actual soil water conductivity and soil water potential in each soil horizon are calculated. These two terms are used to calculate vertical and lateral fluxes of water and soil ions in solution. Vertical water movement is calculated from actual conductivity and the difference in hydraulic potentials. Vertical soil ion flux is then calculated from vertical water flow and soil ion concentration. Lateral water and soil ion fluxes are calculated in the same way as vertical fluxes, except that the lateral differences in hydraulic potential and conductivity are used.

The computed values for soil water and soil ion fluxes and other soil properties are printed by the computer and these are listed in Table C.1. This program consists of about 400 cards of Fortran statements.

b. The Fortran Program BALAN2

Since the program BALAN2 deals with data taken at different times, the rates of change in magnitude and flow of soil ions in solution and water are computed. This results in a larger program and more printed output. Since this makes the program difficult to follow, it

is described in more detail. This program is also divided into five steps, except the steps are not arranged sequentially, as in the first program BALAN1. Originally, the steps were arranged sequentially, but the storage required by the program exceeded that available in the computer. Rearranging the steps permitted some savings in storage used and alleviated this problem. In step 1, the concentrations of soil ions in solution in meq/cm^3 are calculated. In step 2, the soil ions involved in plant uptake and evapotranspiration are calculated. In the third step, the fluxes of soil water and soil ions in solution are calculated. In the fourth step, adsorbed cations are calculated using multiple regression and the equilibrium model. In the fifth step, denitrification is calculated. The output or printing of the calculated values is done after these steps are completed.

In the comments at the beginning of the program, following the title, there is a legend to the various deep soil inspection pits and a second legend indicating how fluxes up and down the profile, and into and out of the septic bed, are identified. Space is then dimensioned for the parameters in the program. Step 1, which calculates soil ion concentrations in solution in meq/cm^3 , is the starting statement. In lines 75 to 77, the dimensions of the data matrix are defined. The equivalent weights of the soil ions are then defined. The data necessary for calculating soil ion concentrations in meq/cm^3 are read in and include soil ion concentrations in moles/l, bulk density and gravimetric moisture content. Cation exchange values and equilibrium constants are also read in. The latter were calculated in the program BALAN1. Volumetric soil moisture is then calculated from gravimetric

soil moisture in lines 96 to 101. Following this, soil ion concentrations in meq/cm^3 are calculated down to line 113 and this ends the first step.

The second step, which is small, begins at about line 115 and calculates soil ions in plant uptake. Measured values for evapotranspiration are read and the ions adsorbed in it are calculated in the following DO loop, which extends to line 146. The next step, number 5, is only 10 lines long and in it, denitrification is calculated from the chloride-nitrate ratio.

The last two steps are the longest steps in the program. Step 4 calculates adsorbed ions and extends from line 157 to 356. First, adsorbed cations are calculated from multiple regression. The parameters in the multiple regression equations are read in lines 165 to 169. The values for adsorbed cations are then calculated in a DO loop from lines 176 to 190. The adsorbed cations are calculated next using the equilibrium model. This is done in a large DO loop from lines 204 to 220.

This is followed by a series of eight nested DO loops which are used to calculate average daily changes in four calculated soil properties. Two nested DO loops are required to calculate the average daily change in one calculated soil property. The first loop arranges a vector of calculated values into a two-dimensional matrix, with one dimension representing the time of sampling. The second DO loop then subtracts each calculated value from the one on the previous sampling for the same pit and depth, and divides it by the elapsed time between the two samplings. Thus the average daily changes in soil ion concentrations, cation adsorption, soil ion uptake and denitrification can be

calculated.

The last major step in the program calculates soil water flow per unit area and fluxes of soil ions in solution. This step extends from line 337 to 512. In line 344, the time interval between samplings is defined. In the next line, the parameters used to calculate water potential and unsaturated hydraulic conductivity from volumetric soil moisture are read. The parameters read include conductivity at saturation, moisture content at saturation, the slope and intercept of the log-log relationship between pressure potential and volumetric soil moisture content, and the moisture content below which this relationship holds. Moisture contents higher than this value are set equal to saturation moisture content.

Following this, in the DO loop from line 348 to 357, unsaturated conductivity is calculated for each soil sample. The next loop from line 359 to 366 calculates the water potential for each soil sample. The third nested DO loop is used to arrange these water potential values in a two-dimensional matrix, in order to assist in the calculation of the water potential gradient in the next DO loop. This DO loop extends from line 389 to 400 and the vertical fluxes of water and soil ions in solution are calculated from it. Soil water flows are calculated from the hydraulic conductivity--which can be saturated or unsaturated--and the hydraulic potential gradient. This is multiplied by 24 to convert the flow from hourly to daily measurements. The soil ion fluxes are then calculated from the water flow and the concentration of soil ions in solution.

The next two nested DO loop are used to calculate the average

daily change in water flows and the fluxes of soil ions in solution. The approach is the same as that used to calculate average daily changes in soil ion concentrations, for example, in lines 222 to 254 and discussed in the previous step.

From measurements in the soil trench at the interface between the field and forest, lateral water flows, and fluxes of soil ions in solution are calculated in lines 439 to 512. A nested DO loop is used to calculate the lateral water potential gradient from water potentials already calculated. In the next nested DO loop extending from line 459 to 471, the lateral water flow and fluxes of soil ions in solution are calculated using the same technique as for the vertical fluxes. The direction of flux, either up or down the profile, or in or out of the bed, is indicated by the sign of the value, as explained in the legend at the beginning of the program and this discussion. The last two nested DO loops in this step are used to calculate the average daily change in lateral fluxes in the soil trench. The approach is the same as that used to calculate the other, previously discussed, average daily changes. A small DO loop was inserted to convert soil depth measurements in inches to centimeters.

The rest of the program consists of output or the printing of calculated values. Forty-five pages of results are printed. This represents 20 different sets of values. For each set of values, a title across the top of the page is printed. This indicates the group of values to be printed and usually includes the units of measurement. Beneath this, headings are printed identifying each column of values. Down the right-hand side of the page of printed values, the soil pit,

sample depth in cm and the time of sampling are indicated for each sample. The time of sampling is time elapsed in days from the commencement of effluent flow. The location of each soil pit is indicated in the legend at the beginning of the program, as previously mentioned. All measurements of soil ion concentrations, either in solution or adsorbed on the exchange complex, are in meq/cm^3 . All measurements of fluxes of soil ions in solution and denitrification are in $\text{meq cm}^{-3} \text{ d}^{-1}$. All measurements of daily change in soil ion concentrations in solution and adsorbed are also in $\text{meq cm}^{-3} \text{ d}^{-1}$. Soil water flows per unit area and daily changes in these flows per unit area are in cm/d . The program consists of 709 cards of Fortran statements.

c. The Fortran Program BALAN3

This program calculates lateral fluxes of water and nitrate in solution and the rate of denitrification during continuous effluent flow. It is the smallest of the three programs, consisting of 115 lines of Fortran statements. Following the title of the program, a legend indicates fluxes into and out of the bed which are distinguished by their sign. Space is dimensioned for the variables in lines 16 to 20. The equivalent weights of the soil ions are defined in line 23, along with the size of the data matrix and the lateral distance between sample points. In lines 24 to 26, the soil ion concentrations in solution, in moles/l; the gravimetric soil moisture content, soil depth and sample site identification are read and stored in the computer. In lines 27 and 28, the parameters needed to calculate water flow from values of moisture content are read. These parameters include hydraulic

conductivity at saturation, moisture content at saturation, the slope and intercept of the log-log relationship between pressure potential and moisture content, and the moisture content below which this relationship is valid.

In the small DO loop extending from lines 30 to 34, volumetric soil moisture is calculated from gravimetric soil moisture and bulk density. In the next DO loop, the concentration of soil ions in meq/cm^3 is calculated from their concentration in moles/l and from their equivalent weights. The next DO loop extending from line 42 to 49 calculates unsaturated soil water conductivity for each sample. The next DO loop calculates denitrification from the chloride to nitrate ratio.

Following this, water potential is calculated for each sample in a DO loop which ends at line 62. The next DO loop calculates the difference in lateral hydraulic potential between the sample inside the septic bed and the nearest sample outside the septic bed at the same depth. The distance between these samples is approximately 182 cm. The next loop extends from line 67 to 74 and is used to calculate lateral soil water flows and lateral fluxes of soil ions in solution. In line 68, soil water flows are calculated from the soil water conductivity and hydraulic potential gradient. Lateral fluxes of soil ions in solution were then calculated from the flow of soil water and the concentration of soil ions in solution. The next loop calculates the difference between lateral water flow and the lateral flux of nitrate in solution. Attempts were unsuccessful at correlating this with other parameters, including calculated denitrification, as a means of explaining variations in this difference. The last DO loop in the main part of the

program converted measurements of soil depth in inches to centimeters.

The remaining 34 lines of Fortran print the calculated values.

The headings, labelling and units of measurement of the calculated values are very similar to those in the programs BALAN1 and BALAN2.

The concentrations of soil ions in solution are in meq/cm^3 and the fluxes of soil ions in solution and denitrification are in $\text{meq cm}^{-3} \text{ d}^{-1}$.

The water flows per unit area are in cm/d .

PROGRAM BALANI

PALE

```

115 IF (K0(I)).GT.6) GO TO 40
116 IF (KP(I)).EQ.1) FLUX=FLUX(1)
117 IF (KP(I)).EQ.2) FLUX=FLUX(1)
118 IF (KP(I)).EQ.3) FLUX=FLUX(2)
119 IF (KP(I)).EQ.4) FLUX=FLUX(2)
120 IF (KP(I)).EQ.5) FLUX=FLUX(3)
121 L=L+1
122 UCA(I)=XCA(I)*FLUX*(ECA/1000.)
123 UHA(I)=XHA(I)*FLUX*(EHA/1000.)
124 UK(I)=XK(I)*FLUX*(EK/1000.)
125 UCL2(I)=XCL2(I)*FLUX*(ECL2/1000.)
126 UKO3(I)=XKO3(I)*FLUX*(EKO3/1000.)
127 UHG(I)=XHG(I)*FLUX*(EHG/1000.)
128 UHH4(I)=XHH4(I)*FLUX*(EHH4/1000.)
129 40 CONTINUE
130 STEP 3 CALCULATE EQUILIBRIUM CONSTANTS DIMENSIONLESS
131 DO 50 I=1,N
132 Z(A(I))=(XNA(I)/1000.)*(ENA/1000.)
133 ZCA(I)=(XCA(I)/1000.)*(ECA/1000.)
134 ZK(I)=(XK(I)/1000.)*(EK/1000.)
135 AZCA(I)=(ACA(I)/100.)*(1./80(I))*(100./H2O(I))
136 AZKA(I)=(AK(I)/100.)*(1./80(I))*(100./H2O(I))
137 50 CONTINUE
138 DO 60 I=1,N
139 XKH1(I)=(ZK(I)*AZHA(I))/(ZHA(I)*AK(I))
140 XKH2(I)=(ZK(I)*ZK(I))/(ZCA(I)*AZCA(I))
141 XKH3(I)=(AZHA(I)/ZHA(I))*2*(ZCA(I)/AZCA(I))
142 60 CONTINUE
143 STEP 4 CALCULATE CL2 H2O3 RATIOS USING MEQ PER CC
144 AND DENSIFICATION
145 DO 90 I=1,N
146 RAT(I)=YCL2(I)/YNO3(I)
147 IF (RAT(I).LT.0.15) GO TO 90
148 RAT=0.119
149 DENSIF(I)=YNO3(I)-(YCL2(I)/RAT)
150 90 CONTINUE
151 STEP 4 CALCULATE EQUILIBRIUM EQUATION 14 FOR PREDICTING
152 CATIONS
153 TEST OF EQUILIBRIUM EQUATION 14 FOR CALCULATING ANIONS
154 DO 15 I=1,N
155 Y(I)=XCA(I)*XK01(I)/YK(I)*2.0)
156 Y(I)=(XCA(I)*XK(I)+XHA(I))
157 Y(I)=(XCA(I)*XK(I)+XHA(I))/(XK(I)*YK(I))
158 CPX(I)=(Y(I)*2.0-(4.0-X(I))*(-2(I)))
159 CPX(I)=SUM(CPX(I))
160 CPX(I)=ABS(CPX(I))
161 CBK(I)=(-Y(I))*CPX(I)/(2.0*X(I))
162 CBK(I)=(XKH1(I)*CBK(I)+YHA(I))*YK(I)/YK(I)
163 CBK(I)=2(I)-(CBNA(I)+CBK(I))
164 15 CONTINUE
165 4
166
167

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PROGRAM BALANI

```

175      CELK(I)=CK(I)-BK(I)
        DELNA(I)=CONA(I)-ONA(I)
        DELCA(I)=CACA(I)-BCA(I)
        ZEC(I)=CHNA(I)+CQA(I)+CBK(I)
        IF (DELCA(I).EQ.0.) GO TO 15
        DELPCA(I)=(DELCA(I)/BCA(I))-100.0
        IF (DELNA(I).EQ.0.) GO TO 15
        DEPHA(I)=(DELNA(I)/ONA(I))-100.0
        IF (DELK(I).EQ.0.) GO TO 12
        DEPK(I)=(DELK(I)/BK(I))-100.0
15      CONTINUE
        GO TO 17
185      XAL(I)=BK(I)+((CK(I)-2)*X(I))
17      CONTINUE

STEP 5 CALCULATE WATER FLUXES AND IONIC FLUXES IN WATER

117      READ (5,117) ((KJAT(I),H2SAT(I),XB(I),OS(I),AM(I),UM(I)),I=1,M)
        GO TO 118
118      IF (M20(I).GT.XB(I)) GO TO 116
        KH(I)=KJAT(I)+((H2O(I)/H2SAT(I))*((2.*B(I))+3.))
        GO TO 115
116      XW(I)=KJAT(I)
115      CONTINUE
        GO TO 121
121      IF (H2O(I).LT.2.0) H2O(I)=2.0
        XPH2O(I)=A*(I)+((ALOG10(H2O(I))+BW(I))
        PH2O(I)=10.**XPH2O(I))
121      CONTINUE
        I=2
        IF (KPI(I).EQ.3.AND.KO(I).EQ.1) GO TO 123
        IF (KPI(I).EQ.4.AND.KO(I).EQ.1) GO TO 123
        IF (KPI(I).EQ.5.AND.KO(I).EQ.1) GO TO 122
        IF (KPI(I).EQ.2.AND.KO(I).EQ.1) GO TO 122
        IF (KPI(I).EQ.1.AND.KO(I).EQ.1) GO TO 122
        IX=I-1
        OH2O(I)=((PH2O(I)-PH2O(IX))/OS(I))
122      CONTINUE
        I=2
        IF (KPI(I).EQ.3.AND.KO(I).EQ.1) GO TO 123
        IF (KPI(I).EQ.4.AND.KO(I).EQ.1) GO TO 123
        IF (KPI(I).EQ.5.AND.KO(I).EQ.1) GO TO 123
        IF (KPI(I).EQ.1.AND.KO(I).EQ.1) GO TO 123
        IF (KPI(I).EQ.2.AND.KO(I).EQ.1) GO TO 123
        FK(I)=FH2O(I)+YK(I)
        FCL(I)=FH2O(I)+YCL(I)
        FCA(I)=FH2O(I)+YCA(I)
        FHC(I)=FH2O(I)+YHC(I)
        FNA(I)=FH2O(I)+YNA(I)
        FHM4(I)=FH2O(I)+YHM4(I)

```



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PROGRAM BALANI
290      WRITE (6,103) (KP(I),KO(I),UK(I),UCL2(I),UMOS(I),UCA(I),
1      UMG(I),UMA(I),UMH4(I))
103      FORMAT (1X,12,3X,12,7E10.3)
210 CONTINUE
295      FROM STEP 3 EQUILIBRIUM CONSTANTS DIMENSIONLESS
300      WRITE (6,104)
104      FORMAT (1X,12,3X,12,7E10.3)
305      FROM STEP 4 CHLORIDE TO NITRATE RATIOS DIMENSIONLESS
310      AND DENITRIFICATION
315      WRITE (6,105)
105      FORMAT (1X,12,3X,12,7E10.3)
320      FROM STEP 5 EQUILIBRIUM EQUATION 14
325      TEST OF EQUILIBRIUM EQUATION 14 FOR CALC
330      ULATING ABSORBED CATIONS// 31M PIT DEPTH RIGHT SYOL LEFT SIDE//
335      FROM STEP 6 EQUILIBRIUM EQUATION 14//
340      OUTPUT OF WATER FLUXES

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PROG: A 84478

[illegible]


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PROGPA+ BALANZ
60 DIMENSION XKH1(170),XKQ1(170),XKQ2(170),UNH4(170)
   DIMENSION CKA(170),CQA(170),CJA(170),CZ(170)
   DIMENSION YKA(170),YQA(170),YNA(170),YCLA(170),YPO(170),UK(170)
   DIMENSION TFH20(28,6),TFCA(28,6),TFNO3(28,6),TFCL2(29,6)
   DIMENSION TFK(28,6),TFNA(28,6),TFNH4(28,6),TFPO(28,6)
   DIMENSION RATI(28,6),A(168),FPCAL(168)
   DIMENSION DM2L(100),FLM20(100),FCL(100),FNOL(100),FCL(100)
   DIMENSION FCL(100),FHL(100),FHL(100),DFL(100),DFL(100)
   DIMENSION DELCA(100),DELH(100),DFLH(100),DFL(100),DFL(100)
   DIMENSION DELM2(100),DFLH(100),DFL(100),DFL(100),DFL(100)
   DIMENSION TELM2(28,6),TFCL(28,6),TFCL(28,6),TFCL(28,6),TFCL(28,6)
   REAL KM,KSAT
70 STEP1 CALCULATE MEQ PER CC IONIC CONCENTRATIONS
   H=168 & H=28
   HA=28
   HB=28
   HMA=28
   HMB=28
75
80 READ IN IONIC CONCENTRATIONS
   READ IN SOIL MOISTURE AND BULK DENSITY VALUES
85
90 READ (5,10) ((KP(I),KT(I),KO(I),XCA(I),XNO3(I),XCL2(I),YPO(I),
   1XK(I),XNA(I),XNH4(I),H20M(I)),I=1,N)
   100 FOPMA(1,1)=1X,12,7F10.7/16X,F5.2)
   110 FOPMA(1,10)=16(J),CEC(J),J=1,N3)
   120 READ (5,10) ((XKH1(J),XKQ1(J),XKQ2(J)),J=1,NB)
   130 FOPMA(1,10)=16(J),CEC(J),J=1,N3)
95 CALCULATE VOLUMETRIC SOIL MOISTURE FROM GRAVIMETRIC
   J=1
   DO 10 I=1,N
     IF (J.EQ.29) J=1
     H20(I)=H20M(I)*BD(J)
     J=J+1
100 CONTINUE
105 CALCULATE MEQ PER CC IONIC CONCENTRATIONS
   DO 20 I=1,N
     YCA(I)=(H20(I)/100.)*(XCA(I)/100.)*(ECA/1000.)
     YNO3(I)=(H20(I)/100.)*(XNO3(I)/100.)*(ENO3/1000.)
     YCL2(I)=(H20(I)/100.)*(XCL2(I)/100.)*(ECL2/1000.)
     YPO4(I)=(H20(I)/100.)*(XPO4(I)/100.)*(EPO4/1000.)
     YK(I)=(H20(I)/100.)*(XK(I)/100.)*(EK/1000.)
     YNA(I)=(H20(I)/100.)*(XNA(I)/100.)*(ENA/1000.)
     YNH4(I)=(H20(I)/100.)*(XNH4(I)/100.)*(ENH4/1000.)
20 CONTINUE

```


PROGRAM BALAN2

PAGE 4

```

175 IF (J.EQ.29) J=1
    CEC(I)=CEC(J)
    J=J+1
180 DO 84 I=1,N
    DEPTH(I)=XK0(I)
    CZ(I)=(DEPTH(I)+XK(1))+(YNH4(I)+XK(2))+(YCA(I)+XK(3))
    + (YNO3(I)+XK(4))+(YCL2(I)+XK(5))+(YNA(I)+XK(6))
    + (YV(I)+XK(7))+(YH(I)+XK(8))+(YCK(1)+XK(9))
    CK(I)=(DEPTH(I)+XK(1))+(YNH4(I)+XK(2))+(YCA(I)+XK(3))
    + (YNO3(I)+XK(4))+(YCL2(I)+XK(5))+(YNA(I)+XK(6))
    + (YV(I)+XK(7))+(YH(I)+XK(8))+(YCK(1)+XK(9))
    CCA(I)=(DEPTH(I)+XK(1))+(YNH4(I)+XK(2))+(YCA(I)+XK(3))
    + (YNO3(I)+XK(4))+(YCL2(I)+XK(5))+(YNA(I)+XK(6))
    + (YV(I)+XK(7))+(YH(I)+XK(8))+(YCK(1)+XK(9))
    CHA(I)=(DEPTH(I)+XK(1))+(YNH4(I)+XK(2))+(YCA(I)+XK(3))
    + (YNO3(I)+XK(4))+(YCL2(I)+XK(5))+(YNA(I)+XK(6))
    + (YV(I)+XK(7))+(YH(I)+XK(8))+(YCK(1)+XK(9))
84 CONTINUE

```

CALCULATE EQUILIBRIUM EQUATION 14 FOR PREDICTING ADSORBED CATIONS

```

195 J=1
    DO 12 I=1,N
    IF (J.EQ.29) J=1
    XK01(I)=XK01(J)
    XKH1(I)=XKH1(J)
    J=J+1
12 CONTINUE
    Y(I)=YCA(I)+XK01(I)/(YK(I)+2)
    Y(I)=Y(I)+YNA(I)/(XKH1(I)+YK(I))
    IF (Y(I).EQ.0) GO TO 15
    IF (X(I).EQ.0) GO TO 15
    CPX(I)=(Y(I)+2)/((4.2*X(I)+(-CZ(I))))
    CPX(I)=ABS(CPX(I))
    CPX(I)=SQRT(CPX(I))
    CBK(I)=1-Y(I)+CPX(I)/(2.0*X(I))
    GO TO 16
14 CBK(I)=0
16 CBNA(I)=(XKH1(I)+CBK(I)+YNA(I))/YK(I)
    CBKA(I)=CZ(I)-CBNA(I)+CBK(I)
    Z(I)=CBKA(I)+CBNA(I)+CBK(I)
    ZCEC(I)=CEC(I)-CZ(I)
15 CONTINUE
    I=1
    CALCULATE CHANGE IN IONIC CONCENTRATIONS OVER TIME
    T(2)=3, & T(3)=4, & T(4)=5, & T(5)=4, & T(6)=3,
    JH=20 & KH=6
    I=1 DO 55 K=1,KH

```

PROGRAM BAL 4142

```

230      DO 50 J=1,JH
231        TCA(J,K)=YCA(I)
232        THA(J,K)=YHA(I)
233        TNO(J,K)=YNO(I)
234        TOL2(J,K)=YCL2(I)
235        TPO4(J,K)=YPO4(I)
236        THM4(J,K)=YHM4(I)
237        TK(J,K)=YK(I)
238        I=I+1
239      CONTINUE
240      I=1
241      DO 70 K=2,KH
242        DO 65 J=1,JH
243          KI=K-1
244          OCA(I)=TCA(J,KI)-TCA(J,K)
245          ONO(I)=TNO(J,KI)-TNO(J,K)
246          OCL2(I)=TCL2(J,KI)-TCL2(J,K)
247          OPO4(I)=TPO4(J,KI)-TPO4(J,K)
248          OHM4(I)=THM4(J,KI)-THM4(J,K)
249          OK(I)=TK(J,KI)-TK(J,K)
250          I=I+1
251        CONTINUE
252      CONTINUE
253      I=1
254      DO 26 K=1,KH
255        DO 25 J=1,JH
256          TKA(J,K)=CK(K,I)
257          TCA(J,K)=CBKA(I)
258          THA(J,K)=CBNA(I)
259          TCZ(J,K)=CZ(I)
260          TCC(J,K)=ZCC(I)
261          I=I+1
262        CONTINUE
263      CONTINUE
264      I=1
265      DO 36 K=2,KH
266        DO 35 J=1,JH
267          KI=K-1
268          OCA(I)=TCA(J,KI)-TCA(J,K)
269          ONO(I)=TNO(J,KI)-TNO(J,K)
270          OCL2(I)=TCL2(J,KI)-TCL2(J,K)
271          OPO4(I)=TPO4(J,KI)-TPO4(J,K)
272          OHM4(I)=THM4(J,KI)-THM4(J,K)
273          OK(I)=TK(J,KI)-TK(J,K)
274          I=I+1
275        CONTINUE
276      CONTINUE
277      I=1
278      DO 66 K=1,KH
279        DO 65 J=1,JH
280          KI=K-1
281          OCA(I)=TCA(J,KI)-TCA(J,K)
282          ONO(I)=TNO(J,KI)-TNO(J,K)
283          OCL2(I)=TCL2(J,KI)-TCL2(J,K)
284          OPO4(I)=TPO4(J,KI)-TPO4(J,K)
285          OHM4(I)=THM4(J,KI)-THM4(J,K)
286          OK(I)=TK(J,KI)-TK(J,K)
287          I=I+1
288        CONTINUE
289      CONTINUE
290      I=1
291      DO 66 K=1,KH
292        DO 65 J=1,JH
293          KI=K-1
294          OCA(I)=TCA(J,KI)-TCA(J,K)
295          ONO(I)=TNO(J,KI)-TNO(J,K)
296          OCL2(I)=TCL2(J,KI)-TCL2(J,K)
297          OPO4(I)=TPO4(J,KI)-TPO4(J,K)
298          OHM4(I)=THM4(J,KI)-THM4(J,K)
299          OK(I)=TK(J,KI)-TK(J,K)
300          I=I+1
301        CONTINUE
302      CONTINUE
303      I=1
304      DO 66 K=1,KH
305        DO 65 J=1,JH
306          KI=K-1
307          OCA(I)=TCA(J,KI)-TCA(J,K)
308          ONO(I)=TNO(J,KI)-TNO(J,K)
309          OCL2(I)=TCL2(J,KI)-TCL2(J,K)
310          OPO4(I)=TPO4(J,KI)-TPO4(J,K)
311          OHM4(I)=THM4(J,KI)-THM4(J,K)
312          OK(I)=TK(J,KI)-TK(J,K)
313          I=I+1
314        CONTINUE
315      CONTINUE
316      I=1
317      DO 66 K=1,KH
318        DO 65 J=1,JH
319          KI=K-1
320          OCA(I)=TCA(J,KI)-TCA(J,K)
321          ONO(I)=TNO(J,KI)-TNO(J,K)
322          OCL2(I)=TCL2(J,KI)-TCL2(J,K)
323          OPO4(I)=TPO4(J,KI)-TPO4(J,K)
324          OHM4(I)=THM4(J,KI)-THM4(J,K)
325          OK(I)=TK(J,KI)-TK(J,K)
326          I=I+1
327        CONTINUE
328      CONTINUE
329      I=1
330      DO 66 K=1,KH
331        DO 65 J=1,JH
332          KI=K-1
333          OCA(I)=TCA(J,KI)-TCA(J,K)
334          ONO(I)=TNO(J,KI)-TNO(J,K)
335          OCL2(I)=TCL2(J,KI)-TCL2(J,K)
336          OPO4(I)=TPO4(J,KI)-TPO4(J,K)
337          OHM4(I)=THM4(J,KI)-THM4(J,K)
338          OK(I)=TK(J,KI)-TK(J,K)
339          I=I+1
340        CONTINUE
341      CONTINUE
342      I=1
343      DO 66 K=1,KH
344        DO 65 J=1,JH
345          KI=K-1
346          OCA(I)=TCA(J,KI)-TCA(J,K)
347          ONO(I)=TNO(J,KI)-TNO(J,K)
348          OCL2(I)=TCL2(J,KI)-TCL2(J,K)
349          OPO4(I)=TPO4(J,KI)-TPO4(J,K)
350          OHM4(I)=THM4(J,KI)-THM4(J,K)
351          OK(I)=TK(J,KI)-TK(J,K)
352          I=I+1
353        CONTINUE
354      CONTINUE
355      I=1
356      DO 66 K=1,KH
357        DO 65 J=1,JH
358          KI=K-1
359          OCA(I)=TCA(J,KI)-TCA(J,K)
360          ONO(I)=TNO(J,KI)-TNO(J,K)
361          OCL2(I)=TCL2(J,KI)-TCL2(J,K)
362          OPO4(I)=TPO4(J,KI)-TPO4(J,K)
363          OHM4(I)=THM4(J,KI)-THM4(J,K)
364          OK(I)=TK(J,KI)-TK(J,K)
365          I=I+1
366        CONTINUE
367      CONTINUE
368      I=1
369      DO 66 K=1,KH
370        DO 65 J=1,JH
371          KI=K-1
372          OCA(I)=TCA(J,KI)-TCA(J,K)
373          ONO(I)=TNO(J,KI)-TNO(J,K)
374          OCL2(I)=TCL2(J,KI)-TCL2(J,K)
375          OPO4(I)=TPO4(J,KI)-TPO4(J,K)
376          OHM4(I)=THM4(J,KI)-THM4(J,K)
377          OK(I)=TK(J,KI)-TK(J,K)
378          I=I+1
379        CONTINUE
380      CONTINUE
381      I=1
382      DO 66 K=1,KH
383        DO 65 J=1,JH
384          KI=K-1
385          OCA(I)=TCA(J,KI)-TCA(J,K)
386          ONO(I)=TNO(J,KI)-TNO(J,K)
387          OCL2(I)=TCL2(J,KI)-TCL2(J,K)
388          OPO4(I)=TPO4(J,KI)-TPO4(J,K)
389          OHM4(I)=THM4(J,KI)-THM4(J,K)
390          OK(I)=TK(J,KI)-TK(J,K)
391          I=I+1
392        CONTINUE
393      CONTINUE
394      I=1
395      DO 66 K=1,KH
396        DO 65 J=1,JH
397          KI=K-1
398          OCA(I)=TCA(J,KI)-TCA(J,K)
399          ONO(I)=TNO(J,KI)-TNO(J,K)
400          OCL2(I)=TCL2(J,KI)-TCL2(J,K)
401          OPO4(I)=TPO4(J,KI)-TPO4(J,K)
402          OHM4(I)=THM4(J,KI)-THM4(J,K)
403          OK(I)=TK(J,KI)-TK(J,K)
404          I=I+1
405        CONTINUE
406      CONTINUE
407      I=1
408      DO 66 K=1,KH
409        DO 65 J=1,JH
410          KI=K-1
411          OCA(I)=TCA(J,KI)-TCA(J,K)
412          ONO(I)=TNO(J,KI)-TNO(J,K)
413          OCL2(I)=TCL2(J,KI)-TCL2(J,K)
414          OPO4(I)=TPO4(J,KI)-TPO4(J,K)
415          OHM4(I)=THM4(J,KI)-THM4(J,K)
416          OK(I)=TK(J,KI)-TK(J,K)
417          I=I+1
418        CONTINUE
419      CONTINUE
420      I=1
421      DO 66 K=1,KH
422        DO 65 J=1,JH
423          KI=K-1
424          OCA(I)=TCA(J,KI)-TCA(J,K)
425          ONO(I)=TNO(J,KI)-TNO(J,K)
426          OCL2(I)=TCL2(J,KI)-TCL2(J,K)
427          OPO4(I)=TPO4(J,KI)-TPO4(J,K)
428          OHM4(I)=THM4(J,KI)-THM4(J,K)
429          OK(I)=TK(J,KI)-TK(J,K)
430          I=I+1
431        CONTINUE
432      CONTINUE
433      I=1
434      DO 66 K=1,KH
435        DO 65 J=1,JH
436          KI=K-1
437          OCA(I)=TCA(J,KI)-TCA(J,K)
438          ONO(I)=TNO(J,KI)-TNO(J,K)
439          OCL2(I)=TCL2(J,KI)-TCL2(J,K)
440          OPO4(I)=TPO4(J,KI)-TPO4(J,K)
441          OHM4(I)=THM4(J,KI)-THM4(J,K)
442          OK(I)=TK(J,KI)-TK(J,K)
443          I=I+1
444        CONTINUE
445      CONTINUE
446      I=1
447      DO 66 K=1,KH
448        DO 65 J=1,JH
449          KI=K-1
450          OCA(I)=TCA(J,KI)-TCA(J,K)
451          ONO(I)=TNO(J,KI)-TNO(J,K)
452          OCL2(I)=TCL2(J,KI)-TCL2(J,K)
453          OPO4(I)=TPO4(J,KI)-TPO4(J,K)
454          OHM4(I)=THM4(J,KI)-THM4(J,K)
455          OK(I)=TK(J,KI)-TK(J,K)
456          I=I+1
457        CONTINUE
458      CONTINUE
459      I=1
460      DO 66 K=1,KH
461        DO 65 J=1,JH
462          KI=K-1
463          OCA(I)=TCA(J,KI)-TCA(J,K)
464          ONO(I)=TNO(J,KI)-TNO(J,K)
465          OCL2(I)=TCL2(J,KI)-TCL2(J,K)
466          OPO4(I)=TPO4(J,KI)-TPO4(J,K)
467          OHM4(I)=THM4(J,KI)-THM4(J,K)
468          OK(I)=TK(J,KI)-TK(J,K)
469          I=I+1
470        CONTINUE
471      CONTINUE
472      I=1
473      DO 66 K=1,KH
474        DO 65 J=1,JH
475          KI=K-1
476          OCA(I)=TCA(J,KI)-TCA(J,K)
477          ONO(I)=TNO(J,KI)-TNO(J,K)
478          OCL2(I)=TCL2(J,KI)-TCL2(J,K)
479          OPO4(I)=TPO4(J,KI)-TPO4(J,K)
480          OHM4(I)=THM4(J,KI)-THM4(J,K)
481          OK(I)=TK(J,KI)-TK(J,K)
482          I=I+1
483        CONTINUE
484      CONTINUE
485      I=1
486      DO 66 K=1,KH
487        DO 65 J=1,JH
488          KI=K-1
489          OCA(I)=TCA(J,KI)-TCA(J,K)
490          ONO(I)=TNO(J,KI)-TNO(J,K)
491          OCL2(I)=TCL2(J,KI)-TCL2(J,K)
492          OPO4(I)=TPO4(J,KI)-TPO4(J,K)
493          OHM4(I)=THM4(J,KI)-THM4(J,K)
494          OK(I)=TK(J,KI)-TK(J,K)
495          I=I+1
496        CONTINUE
497      CONTINUE
498      I=1
499      DO 66 K=1,KH
500        DO 65 J=1,JH
501          KI=K-1
502          OCA(I)=TCA(J,KI)-TCA(J,K)
503          ONO(I)=TNO(J,KI)-TNO(J,K)
504          OCL2(I)=TCL2(J,KI)-TCL2
```


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PROGRAM BALPH2

```

345      JH=20 $ KN=6
          T(2)=3. $ T(3)=4. $ T(4)=5. $ T(5)=6. $ T(6)=7.
          READ (5,117) ((KCAT(J),H2SAT(J),X8(J),AW(J),BM(J)),J=1,NA)
117      FORMAT (10X,2F10.5/10X,2F10.5/10X,2F10.5)
          J=1
          DO 115 I=1,N
              IF (J.EQ.29) J=1
              IF (H20(I).GT.X3(J)) GO TO 116
              KW(I)=84(J)
              KW(I)=KSAT(J)*(H20(I)/H2SAT(J))**(2.*8(J)+3.)
              J=J+1
              GO TO 115
          116      KW(I)=KSAT(J)
          117      J=J+1
          118      CONTINUE
          J=1
          DO 121 I=1,N
              IF (J.EQ.29) J=1
              IF (H20(I).LT.2.) H20(I)=2.6
              IF (H20(I).GT.H2SAT(J)) H20(I)=H2SAT(J)
              XPH20(I)=4.*J*(2.001J(H20(I))+84(J))
              PH20(I)=10.*(XPH20(I))
              J=J+1
          121      CONTINUE
          I=1
          DO 136 K=1,KH
              GO 137 J=1, JH
          137      CONTINUE
          138      I=I+1
          139      CONTINUE
          I=1
          DO 126 K=1,KH
              J=1, JH
              IF (KJ(I).EQ.3) GO TO 125
              A(I)=LEGVAF(H20(J,K))
              IF (A(I).NE.0.) GO TO 125
              A(I)=LEGVAF(H20(J1,K))
              IF (A(I).NE.0.) GO TO 125
              IF (H20(J,K).EQ.H20(J1,K)) GO TO 125
              CH20(I)=H20(J,K)-H20(J1,K)/7.
              I=I+1
          125      CONTINUE
          126      CONTINUE
          I=1
          DO 145 I=1,NH
              A(I)=LEGVAF(H20(I))
              IF (A(I).NE.0.) GO TO 145
              PH20(I)=PH20(I)+CM20(I)*1.1*24.
              FCA(I)=FCA(I)+YCA(I)
              FLO3(I)=FLO3(I)+YPO3(I)
              FLO2(I)=FLO2(I)+YCL2(I)
              FPO7(I)=FPO7(I)+YPO7(I)
              FKA(I)=FKA(I)+YKA(I)
              FNA(I)=FNA(I)+YNA(I)
              FHH(I)=FHH(I)+YHH(I)
          145      CONTINUE

```

THE EFFECT OF SEPTIC TANKS ON THE ENVIRONMENT


```

PROGRAM BALAN2
515 DO 9 I=1,N
      KO(I)=KO(I)*2.54
      9 CONTINUE
      OUTPUT OF RESULTS
      FROM STEP 1 MEQ PER CC IONIC CONCENTRATIONS
520 WRITE (6,201)
      201 FORMAT (1H1, 42H CA MEQ PER CC IONIC CONCENTRATIONS// 82H P
      1 11T DEPTH TIME CA NO3 PO4 K
      1 NA NH4/)
525 WRITE (6,101) ((VP(I),KO(I),KT(I),YCA(I),YNO3(I),YCL2(I),YPO4(I),
      1 YK(I),YNA(I),YNH4(I)),I=1,N)
      101 FORMAT (1X,11,3X,12,3X,12,2X,7E10.3)
530 FROM STEP 2 MEQ PER CC PER DAY FREE IONS IN UPTAKE
      WRITE (6,203)
      203 FORMAT (1H1, 51H IONIC UPTAKE FLUXES IN MEQ PER CC PER DAY
      1 1K // 82H PIT DEPTH TIME CA NO3 CL2 FO4
      1 NA NH4/)
535 DO 202 I=1,N
      IF (VC(I).GT.7) GO TO 202
      WRITE (6,103) (KP(I),KO(I),KT(I),UCA(I),UNHC3(I),UCL2(I),
      1 UPO4(I),UK(I),UNA(I),UNH4(I))
      103 FORMAT (1X,11,4X,12,3X,12,4X,7E10.3)
540 202 CONTINUE
      FROM STEP 5 CHLORIDE TO NITRATE RATIO DIMENSIONLESS
545 WRITE (6,205)
      205 FORMAT (1H1, 49H CHLORIDE TO NITRATE RATIO DIMENSIONLESS/
      1 12H PIT DEPTH TIME CL2 TO NO3 RATIO/)
550 WRITE (6,105) ((KP(I),KO(I),KT(I),RATIO(I)),I=1,N)
      105 FORMAT (1X,12,3X,12,3X,12,4X,7E10.5)
555 WRITE (6,701)
      701 FORMAT (1H1, 53H DENITRIFICATION IN MEQ CC DAY// 33H PIT
      1 DEPTH TIME DENITRIFICATION/)
      DO 106 I=1,N
      IF (RATIO(I).LT.1.3) GO TO 106
      WRITE (6,700) (KP(I),KO(I),KT(I),DENIT(I))
      700 FORMAT (1X,12,3X,12,3X,12,10X,7E10.3)
560 106 CONTINUE
      238 FROM STEP 6, 23A)
      1 REGRESSION// 53H SUM OF CATIONS PREDICTED FROM MULTIPLE R
      1 DEPTH TIME SUM OF CA NA K/)
565 WRITE (6,136) ((KP(I),KO(I),KT(I),I=1,N)
      136 FORMAT (1X,12,3X,12,3X,12,2X,7E10.3)
      242 FROM STEP 6, 242)
      1 REGRESSION// 61H ADOPTED CATIONS PREDICTED FROM MULTIPLE
      1 DEPTH TIME CA NA K/)
570 WRITE (6,142) ((KP(I),KO(I),KT(I),CCA(I),CA,NA))
      142 FORMAT (1X,12,3X,12,3X,12,2X,7E10.3)
      604 FROM STEP 6, 604)
      604 FORMAT (1H1, 47H X PARAMETERS IN EQUILIBRIUM EQUATION 14//
      1 31H PIT DEPTH TIME X

```



```

PROGRAM BALANCE
575 WRITE (6,605), ((KPI(I),KO(I),KI(I),X(I),Y(I)),I=1,N)
580 FORMAT (1X,12,3X,12,3X,12,3X,2E10.3)
581 WRITE (6,611)
582 FORMAT (1H1, 23H CALCULATED CATIONS// 42H PIT DEPTH TIME
583 K
584 WRITE (6,610) ((KPI(I),KO(I),KI(I),C3K(I),C8CA(I),C9HA(I)),I=1,N)
585 FORMAT (1X,12,3X,12,3X,12,3X,3E10.3)
586 WRITE (6,631)
587 FORMAT (1H1, 21H SATURATION OF EXCHANGE CAPACITY// 53H PI
588 IT DEPTH TIME ACCUMULATED CATIONS UNSATURATED CAPACITY//
589 WRITE (6,630) ((KPI(I),KO(I),KI(I),CZ(I),C2C(I),C2C(I)),I=1,N)
590 FORMAT (1X,12,3X,12,3X,12,3X,3E10.3)
591 WRITE (6,636)
592 FORMAT (1H1, 55H DAILY CHANGE IN DENITRIFICATION//
593 19 CC// 36H PIT DEPTH TIME
594 DO 220 I=1,N
595 IF (A(I)-1E-3) GO TO 220
596 WRITE (6,634) ((KPI(I),KO(I),KI(I),X(I),Y(I)),I=1,N)
597 FORMAT (1X,12,3X,12,3X,12,3X,2E10.3)
598 CONTINUE
599
600 FF04 STEP 4 CHANGES IN IONIC CONCENTRATIONS OVER TIME
601
602 WRITE (6,204)
603 FORMAT (1H1, 50H CHANGE IN IONIC CONCENTRATIONS OVER TIME
604 1// 82H PIT DEPTH TIME
605 1X, 11H
606 DO 206 I=1,N
607 IF (KPI(I)-E0.17) GO TO 206
608 WRITE (6,205) ((KPI(I),KO(I),KI(I),OCA(I),ONOS(I),OCL2(I),
609 19PDS(I),C3K(I),C8CA(I),C9HA(I),C2C(I),C2C(I)),I=1,N)
610 FORMAT (1X,11,4X,12,3X,12,3X,7E10.3)
611 CONTINUE
612 WRITE (6,232)
613 FORMAT (1H1, 42H DAILY CHANGE IN ADSORBED CATIONS//
614 1-CC// 36H PIT DEPTH TIME
615 DO 207 I=1,N
616 IF (KPI(I)-E0.17) GO TO 207
617 WRITE (6,233) ((KPI(I),KO(I),KI(I),OCA(I),OCAA(I),OCAA(I),OCZ(I),
618 19PDS(I),C3K(I),C8CA(I),C9HA(I),C2C(I),C2C(I)),I=1,N)
619 FORMAT (1X,12,3X,12,3X,12,3X,5E10.3)
620 CONTINUE
621 WRITE (6,236)
622 FORMAT (1H1, 34H DAILY CHANGE IN IONIC UPTAKE// 82H PIT D
623 1EPH TIME
624 DO 210 I=1,N
625 IF (KPI(I)-E0.17) GO TO 210
626 IF (KO(I)-E0.17) GO TO 210
627 WRITE (6,237) ((KPI(I),KO(I),KI(I),OCA(I),OCAA(I),OCAA(I),OCZ(I),
628 19PDS(I),C3K(I),C8CA(I),C9HA(I),C2C(I),C2C(I)),I=1,N)
629 FORMAT (1X,12,3X,12,3X,12,3X,7E10.3)
630 CONTINUE
631
632 WRITE (6,157)

```

```

PROGRAM          BALANZ
157 FORMAT (1H1, 4EH      COMPUTED SOIL WATER FLUXES// 33H PIT DEP
DO 155 1H1, 4EH      MAT-R FLUXES/)
155 1H1, 4EH      EQ.7) GO TO 155
IF (K01) = 0.0) GO TO 155
A11) = LEQVAR(FH20(I))
IF (A(I) - NE.0) GO TO 155
WRITE (6,156) (KP(I), K01(I), KI(I), FH20(I))
FORMAT (1X,12,3X,12,3X,12,2X,E10.3)
156 CONTINUE
158 WRITE (6,157) 45H      COMPUTED IONIC FLUXES IN SOIL WATER// K
167 FORMAT (1H1, 45H      103      CL2
1 84H PIT DEPTH TIME      NH4/)
DO 165 1H1, 45H      EQ.7) GO TO 165
IF (K01) = 0.0) GO TO 165
A11) = LEQVAR(FH20(I))
IF (A(I) - NE.0) GO TO 165
WRITE (6,156) (KP(I), K01(I), KI(I), FH20(I))
FORMAT (1X,12,3X,12,3X,12,2X,E10.3)
166 CONTINUE
168 WRITE (6,157) 43H      DAILY CHANGE IN SOIL WATER FLUXES//
244 FORMAT (1H1, 43H      FLUX/)
DO 195 1H1, 43H      EQ.7) GO TO 195
IF (K01) = 0.0) GO TO 195
A11) = LEQVAR(FH20(I))
IF (A(I) - NE.0) GO TO 195
WRITE (6,156) (KP(I), K01(I), KI(I), FH20(I))
FORMAT (1X,12,3X,12,3X,12,2X,E10.3)
196 CONTINUE
246 WRITE (6,246) 44H      ONLY CHANGE IN IONIC WATER FLUXES// K
1 84H PIT DEPTH TIME      NH4/)
DO 197 1H1, 44H      EQ.7) GO TO 197
IF (K01) = 0.0) GO TO 197
A11) = LEQVAR(FH20(I))
IF (A(I) - NE.0) GO TO 197
WRITE (6,198) (KP(I), K01(I), KI(I), OFCA(I), OFNO3(I), OFCL2(I),
OFPO4(I), OFK(I), OFNA(I), OFNH4(I))
FORMAT (1X,12,3X,12,3X,12,2X,E10.3)
199 CONTINUE
C OUTPUT OF LATERAL WATER AND IONIC FLUXES IN SOIL TRENCH
WRITE (6,74) 52H      LATERAL WATER FLUXES AT EDGE OF SEPTIC T
1 84H PIT DEPTH TIME WATER FLUX/)
DO 6 1H1, 45H      EQ.7) GO TO 6
IF (K01) = 0.0) GO TO 6
A11) = LEQVAR(FH20(I))
IF (A(I) - NE.0) GO TO 6
WRITE (6,75) (KP(I), K01(I), KI(I), FH20(I))
FORMAT (1X,12,3X,12,3X,12,2X,E10.3)
76 CONTINUE
8 WRITE (6,76)

```

PROGRAM BALAN2

```

685 70 FORMAT (1H1, 52H LATERAL IONIC FLUXES AT EDGE OF SEPTIC B,
1ED// 84H PIT DEPTH TIME (HH4//) CA R03 CL2 PO4
100 K
00 7 I=1,N (1),NE,1) GO TO 7
IF (A(1),NE,1) GO TO 7
WRITE (6,85) (K(1),K(1),KT(1),FCAL(1),FNOL(1),FCL(1),FPL(1),
1FNL(1),FNL(1),FNL(1))
05 FORNAT (1H1, 4X, 12, 2X, 7E10.3)
07 CONTINUE
08 WRITE (6,86)
08 FORNAT (1H1, 52H DAILY CHANGE IN WATER AND IONIC FLUXES A
1ED// 84H PIT DEPTH TIME (HH4//) CA R03 CL2 PO4
100 K
00 8 I=1,N (1),NE,1) GO TO 8
IF (K(1),NE,1) GO TO 8
A(1)=LEGVAR(DFLH2(1))
IF (A(1),NE,1) GO TO 8
WRITE (6,87) (K(1),K(1),KT(1),DFLH2(1),DFLCA(1),DFLNO(1),
1DFLCL(1),DFLPO(1),DFLNA(1),DFLNA(1),DFLNA(1))
07 FORNAT (1H1, 4X, 12, 2X, 7E10.3)
08 CONTINUE
08 STOP
END

```

PAGE 1

```

PROGRAM BAL13 (INPUT,OUTPUT,PUNCH,TAPE5=INPUT,TAPE6=OUTPUT,
1  ITAPE7=PUNCH)

```

```

PROGRAM TO COMPUTE LATERAL FLUXES OF IONS AND WATER AROUND THE EDE
5  CF THE SEPTIC BED FROM SPATIAL DATA

```

```

LEGEND TO THE FLUXES

```

```

10 IF POSITIVE FLUX IS INTO SEPTIC BED

```

```

IF NEGATIVE FLUX IS OUT OF BED

```

```

15 DIMENSION BDI(2),PATIO(50),CENTI(50),FLNA(50),XB(2),ENH(2),
DIMENSION DEPM(50),H2O(50),XNO3(50),XCL2(50),XCA(50)
DIMENSION XE(50),Y(50),B(2),BH(2),AM(2),M2SAT(2),XM(50)
20 DIMENSION XPM2C(50),YNO3(50),YCL2(50),YCA(50),OH2L(50)
DIMENSION PM2O(50),FLH2O(50),FLNO3(50),FLCL2(50),FLCA(50),H2OH(50)
REAL KSAT, KW

```

```

J=16 $ DS=182.88 $ ENO3=62. $ ECL2=35.5 $ ENA=23. $ ECA=20.
READ (5,10) ((X(I),Y(I)),(EFT(I),H2OH(I),XNO3(I),XCL2(I),XCA(I)),
1 XCAT(I)),I=1,N)

```

```

10 FORMAT (10X,1P10.5/15X,4F10.8)

```

```

12 FCORAT (10X,6F10.5)
J=1

```

```

30 J=1,N
IF (I.GT.18) J=2

```

```

IF (I.GT.20) J=3

```

```

4 CONTINUE
IF (P2O(I).GT.H2SAT(J)) H2O(I)=H2SAT(J)

```

```

5 CONTINUE

```

```

70 I=1,N
IF (I.GT.18) J=2

```

```

IF (H2C(I).GT.XB(J)) GC TO 15

```

```

80 J=1,N

```

```

90 J=1,N
IF (H2C(I).GT.XB(J)) GC TO 15

```

```

100 J=1,N
IF (H2C(I).GT.XB(J)) GC TO 15

```

```

110 J=1,N
IF (H2C(I).GT.XB(J)) GC TO 15

```

```

120 J=1,N
IF (H2C(I).GT.XB(J)) GC TO 15

```

```

130 J=1,N
IF (H2C(I).GT.XB(J)) GC TO 15

```

```

140 J=1,N
IF (H2C(I).GT.XB(J)) GC TO 15

```

```

150 J=1,N
IF (H2C(I).GT.XB(J)) GC TO 15

```

```

160 J=1,N
IF (H2C(I).GT.XB(J)) GC TO 15

```

```

170 J=1,N
IF (H2C(I).GT.XB(J)) GC TO 15

```

```

180 J=1,N
IF (H2C(I).GT.XB(J)) GC TO 15

```

```

190 J=1,N
IF (H2C(I).GT.XB(J)) GC TO 15

```

```

200 J=1,N
IF (H2C(I).GT.XB(J)) GC TO 15

```

```

210 J=1,N
IF (H2C(I).GT.XB(J)) GC TO 15

```

```

220 J=1,N
IF (H2C(I).GT.XB(J)) GC TO 15

```


APPENDIX D

DERIVATION OF FIGURES IN TABLES 2.5, 5.1 AND 5.2

D.1 Derivation of Figures in Table 2.5

D.2 Derivation of Figures in Tables 5.1 and 5.2

- a. Total Nitrate Flux in Septic Bed Soil
- b. Total Water Flow in Septic Bed Soil
- c. Other Nutrient Fluxes

APPENDIX D

DERIVATION OF FIGURES IN TABLES 2.5, 5.1 and 5.2

D.1 Derivation of Figures in Table 2.5

Table 2.5 represents an attempt at assessing overall output of N and P from septic systems in the United States and Ontario, using published data.

Ryden et al. (1973) estimated total domestic wastes in the U.S. at 91×10^6 to 227×10^6 kg P/yr. This represents a wastewater concentration of 35 to 90 $\mu\text{g P/l}$. He suggested that regional variation was mainly due to variation in use and composition of detergents. Porcella and Bishop (1975) estimated average P and N total domestic waste production at 2.18 kg P per capita/d and 10.8 kg N per capita/d. Of this, 0.96 kg P per capita/d was estimated as detergent. Their estimate of total domestic P use was 176 to 203×10^9 kg/h. Reneau and Pettry (1975) estimated that there were 32 million people on septic systems year round. Their daily water consumption was estimated at 378 l per capita/d ($100 \text{ gal capita}^{-1} \text{ d}^{-1}$). This is twice the average per capita water use commonly reported. Using the figure for average water use of $50 \text{ gal capita}^{-1} \text{ d}^{-1}$, the number of people on septic systems in the United States (32 million) and estimates of the maximum total P and N output from septic tanks, it is possible to estimate N and P input from septic systems. Assuming from Table 2.2 that maximum total P and N

output from septic tanks is 20 and 50 mg/l respectively, N and P input is estimated at 8454 and 2114 kg/d respectively. Brandes et al. (1975), using the 1971 Canada census, computed that there were $\frac{1}{2}$ million conventional septic systems in operation in Ontario, serving more than $1\frac{1}{2}$ million people. Using this figure, the maximum P and N concentrations and average household water use, estimated P and N loads for Ontario are 396 and 990 kg/d respectively.

Using figures from Walker et al. (1973b), a third estimate of N input from septic systems can be calculated. Based on their own measurements of effluent levels in septic beds, they estimated the N input was 33 kg N/yr per septic system and 8.2 kg N/yr per person. These figures produce two very different estimates of N input for Ontario and one for the United States. The average of these estimates for Ontario is 39,450 kg N/yr and 718,902 kg N/yr for the United States.

D.2 Derivation of Figures in Tables 5.1 and 5.2

Tables 5.1 and 5.2 summarize the fluxes of water and nutrients being lost to groundwater from the septic bed and reference soils.

a. Total Nitrate Flux in Septic Bed Soil

(i) Lateral Flux

From BALAN3, the average lateral NO_3 flux = -0.4418×10^{-6} ⁶
 $\text{NO}_3 \text{ meq cm}^{-3} \text{ d}^{-1}$.

⁶The sign indicates flux direction; for lateral flux, - = flux out of septic bed, + = flux into septic bed, for vertical flux, - = flux down the profile, + = uptake.

Since the perimeter of the septic bed is 36 m or 3600 cm,

$$\begin{aligned}\therefore \text{flux for a } 1 \text{ cm}^3 \text{ slice at 30 cm} &= -.4418 \times 10^{-6} \times 3600 \\ &= -.15905 \times 10^{-2} \text{ NO}_3 \text{ meq cm}^{-3} \text{ d}^{-1}.\end{aligned}$$

If the flux were the same for the whole profile (0-100 cm), total flux would be $-.15905 \text{ NO}_3 \text{ meq cm}^{-3} \text{ d}^{-1}$.

If the flux came through in 10 cm, total flux would be $-.15905 \times 10^{-1}$.

\therefore total lateral flux must be between $-.1$ and $-.01 \text{ NO}_3 \text{ meq cm}^{-3} \text{ d}^{-1}$.

Using data on lateral fluxes at the edge of the septic bed from BALAN1, the range can be narrowed. BALAN1 indicates lateral flux at 22 to 32 cm is 66% of the average profile flux. The NO_3 flux reaches a maximum at 30 to 60 cm, where it is 1.92 times as large as the average flux.

Assuming 30 cm depth in BALAN3 approximates flux from 32 to 60 cm in BALAN1, then for 28 cm between 32 and 60 cm:

$$\therefore \text{total lateral flux} = 28 \times -.15905 \times 10^{-2} = -.44533 \times 10^{-1}.$$

For 100 cm between 0 and 100 cm, assuming the average flux is the maximum divided by 1.92, then:

$$\begin{aligned}\text{total lateral flux} &= \frac{-.15905 \times 10^{-2} \times 10^2}{1.92} \\ &= -.82838 \times 10^{-1} \text{ NO}_3 \text{ meq/d.}\end{aligned}$$

Or, of the whole profile, about 60% of the lateral NO_3 flux comes through between 30 and 60 cm.

(ii) Vertical Flux

From BALAN1, vertical flux of NO_3 at the edge of the septic bed and bottom of the profile is: $-.234 \times 10^{-5} \text{ NO}_3 \text{ meq cm}^{-3} \text{ d}^{-1}$. Since the area of the septic bed is 72 m^2 (6 m x 12 m), total vertical flux

$$=(-.234 \times 10^{-5}) \times 72 \times 10^4 = -1.685 \text{ meq/d.}$$

(iii) Comparison of Vertical and Lateral Fluxes

Total flux = 1.768 meq/d for the septic bed or on a cm^3 basis,

$$\text{total flux} = \frac{1.768}{72 \times 10^4} = .245 \times 10^{-5} \text{ meq cm}^{-3} \text{ d}^{-1}.$$

Vertical flux = 95% of total flux although lateral flux does reach up to 90% of the average vertical flux when denitrification is limited.

b. Total Water Flow in Septic Bed Soil

(i) Lateral Flow

From BALAN3, the average lateral water flow = -34.25 cm/d.

Since the perimeter of the septic bed is 36 m or 3600 cm,

$$\therefore \text{flux for a } 1 \text{ cm}^3 \text{ slice at 30 cm} = -34.25 \times 3600 = -123300 \text{ cm/d.}$$

If the flow were the same for the whole profile (0-100 cm) or if the flow came through in 10 cm, total flux would be between $.123 \times 10^9$ and 10^8 cm/d.

Similarly to the computations of nitrate fluxes, this range can be narrowed. For the 28 cm between 32 and 60 cm, total lateral flow

$$28 \times .123 \times 10^6 = .344 \times 10^7 \text{ cm/d.}$$

For 100 cm between 0 and 100 cm, total lateral flow

$$\frac{.123 \times 10^6 \times 10^2}{1.92} = .641 \times 10^7 \text{ cm/d}$$

or, of the whole profile, about 54% of the lateral water flow comes through between 32 and 60 cm.

(ii) Vertical Flow

From BALAN1, vertical flow of water at the edge of the septic bed and bottom of the profile is: 60 cm/d.

Since the area of the septic bed is 72 m^2 ,

$$\text{total vertical flow} \triangleq 60 \times 72 \times 10^4 = .432 \times 10^8 \text{ cm/d.}$$

(iii) Comparison of Vertical and Lateral Flows

Vertical water flow \ll lateral water flow.

Total water flow = $.4961 \times 10^8 \text{ cm/d.}$ for the septic bed or 69 cm/d for 1 cm.

Of this, 87% is vertical flow.

c. Other Nutrient Fluxes

(i) Vertical Phosphate Fluxes in the Septic Bed

BALAN2 computes the PO_4 fluxes entering the C horizon from the lower B horizon at an average level of: $.499 \times 10^{-3} \text{ meq cm}^{-3} \text{ d}^{-1}$.

Precipitation in the C horizon reduces the concentration to the reference levels (Fig. 4.6). Since vertical water flux in the C horizon of the septic bed soil is 25 to 50 times the flux in the reference soils, PO_4 flux in the C horizon is estimated at: $.131 \times 10^{-4} \text{ meq cm}^{-3} \text{ d}^{-1}$ (average of 133.2 and 128.04).

(ii) Total PO_4 and NH_4 Fluxes in the Septic Bed

Ignoring lateral flux, total PO_4 and NH_4 fluxes leaving the septic bed are computed:

$$\text{Total NH}_4 = .915 \times 10^{-7} \times 72 \times 10^4 = .06588 \text{ meq/d}$$

$$\text{Total PO}_4 = .131 \times 10^{-4} \times 72 \times 10^4 = 9.432 \text{ meq/d.}$$

APPENDIX E

COMPUTER MAPPING OF SOIL IONS AND SOIL MOISTURE
IN THE SITE AREA AND IN A TRANSECT THROUGH THE SEPTIC BED

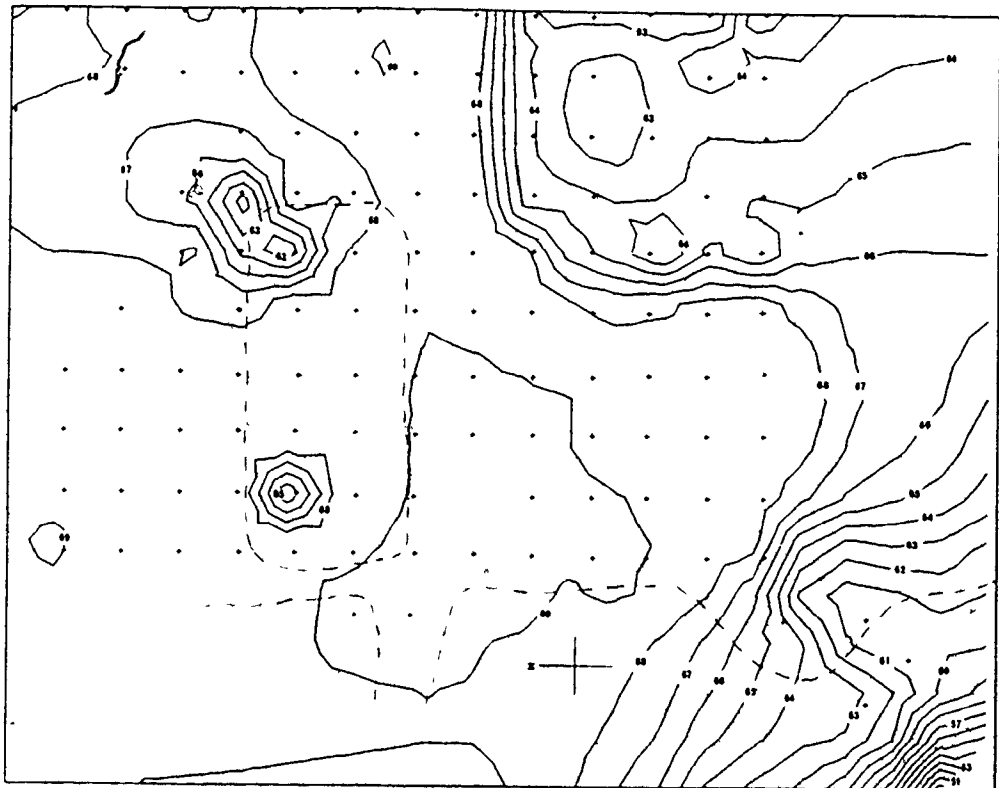


FIG. E.1 Soil pH at 30 cm depth in the site area during continuous effluent flow, contour interval 0.1, 10⁻¹ pH units

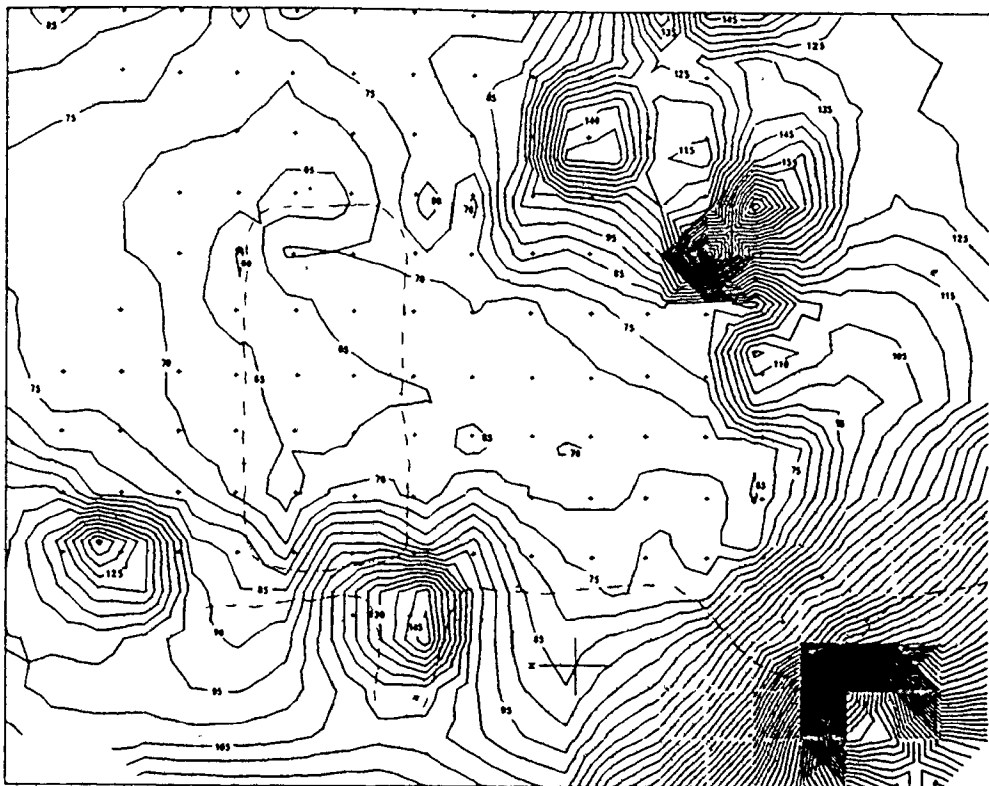


FIG. E.2 Sodium in soil solution at 30 cm depth in the site area during continuous effluent flow, contour interval 0.005, 10^{-4} moles/l

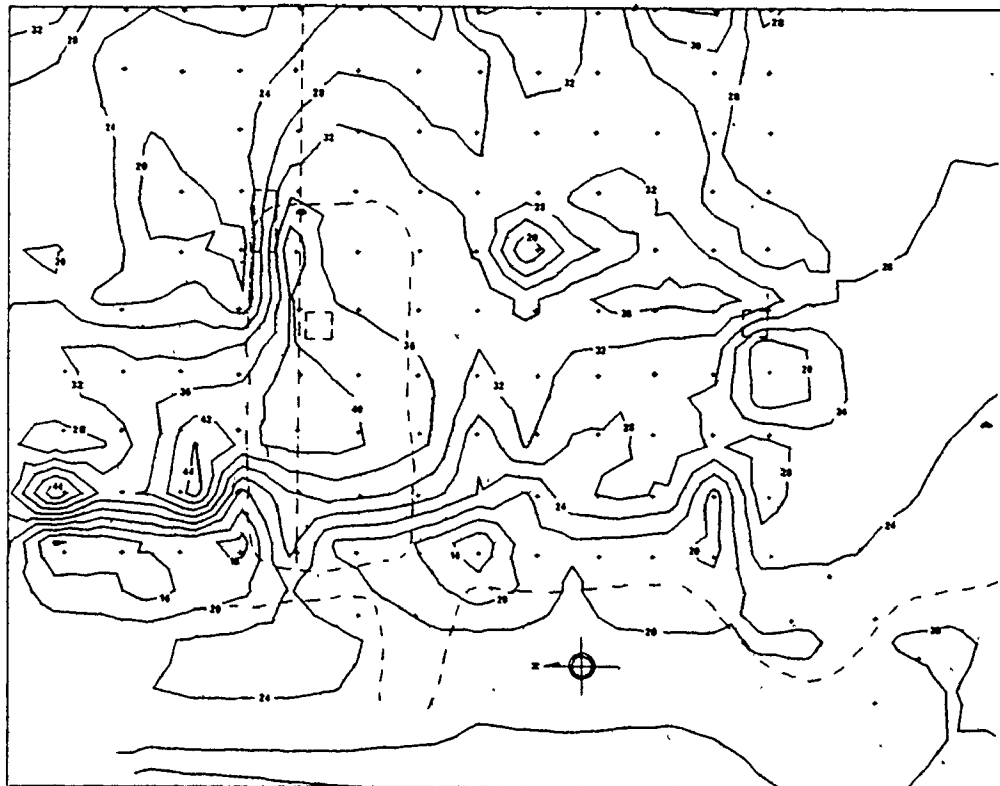


FIG. E.3 Soil moisture by weight at 30 cm depth in the site area during continuous effluent flow, contour interval 4.0, 10⁰ %

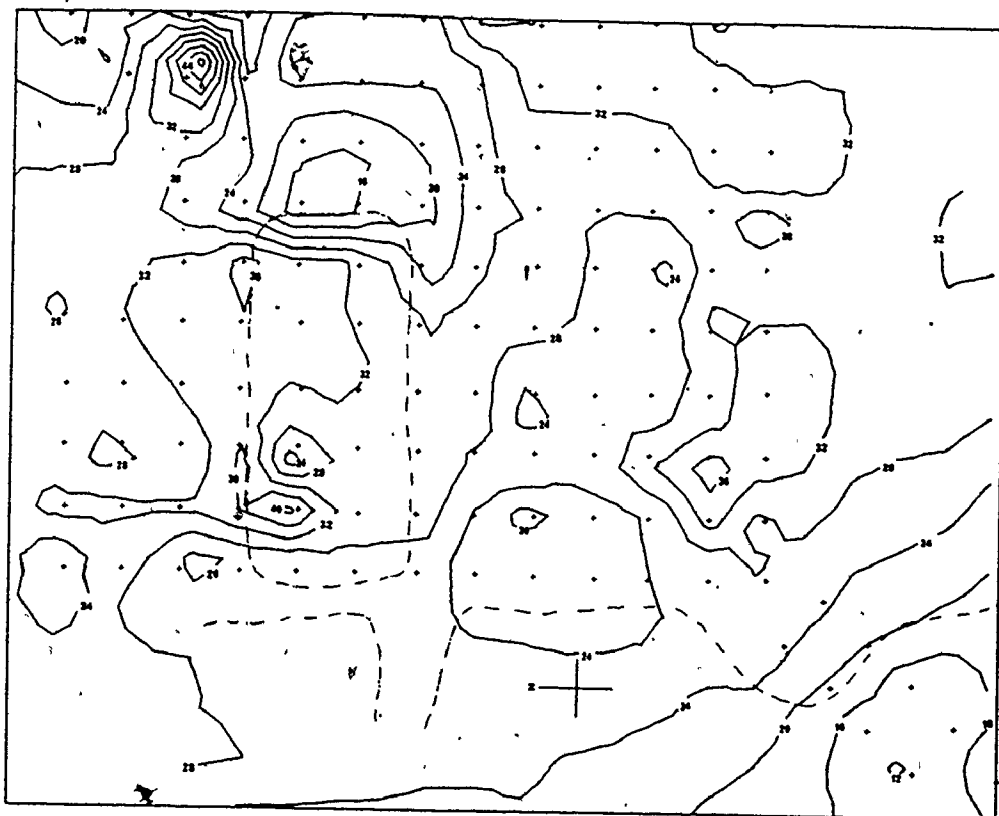


FIG. E.4 Soil oxygen diffusion rate at 7 cm depth in the site area during continuous effluent flow, contour interval 4.0, $10^{-10} \text{ ug cm}^{-2} \text{ min}^{-1}$

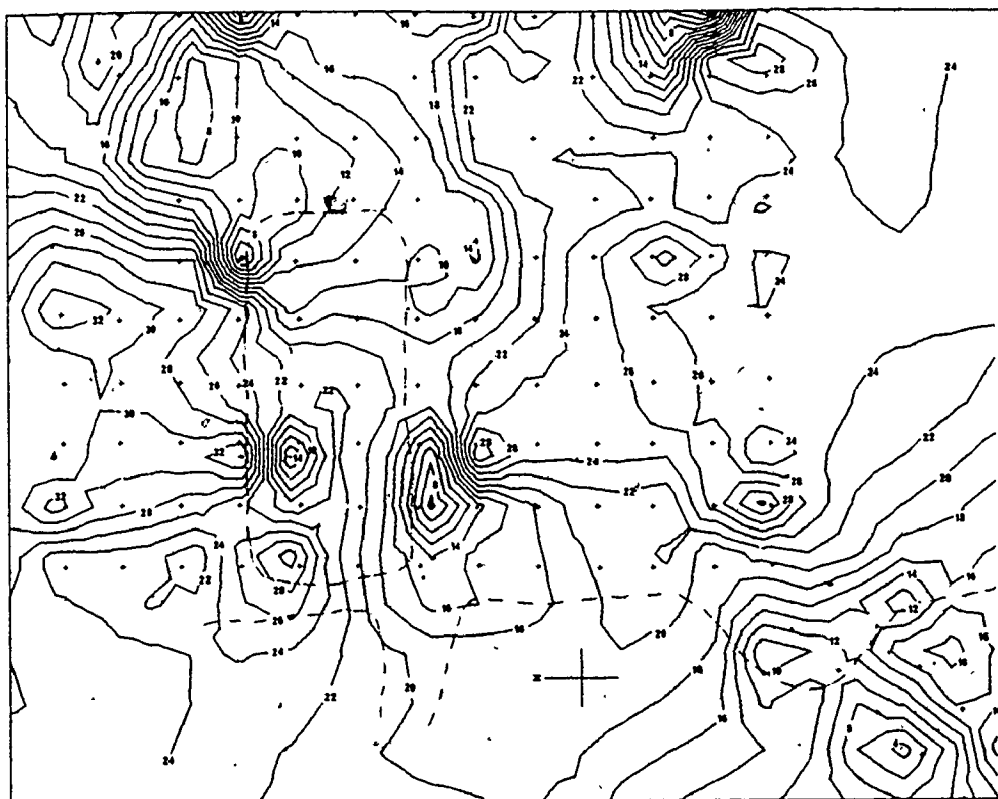


FIG. E.5 Soil oxygen diffusion rate at 30 cm depth in the site area during continuous effluent flow, contour interval 2.0, $10^{-7} \text{ ug cm}^{-2} \text{ min}^{-1}$

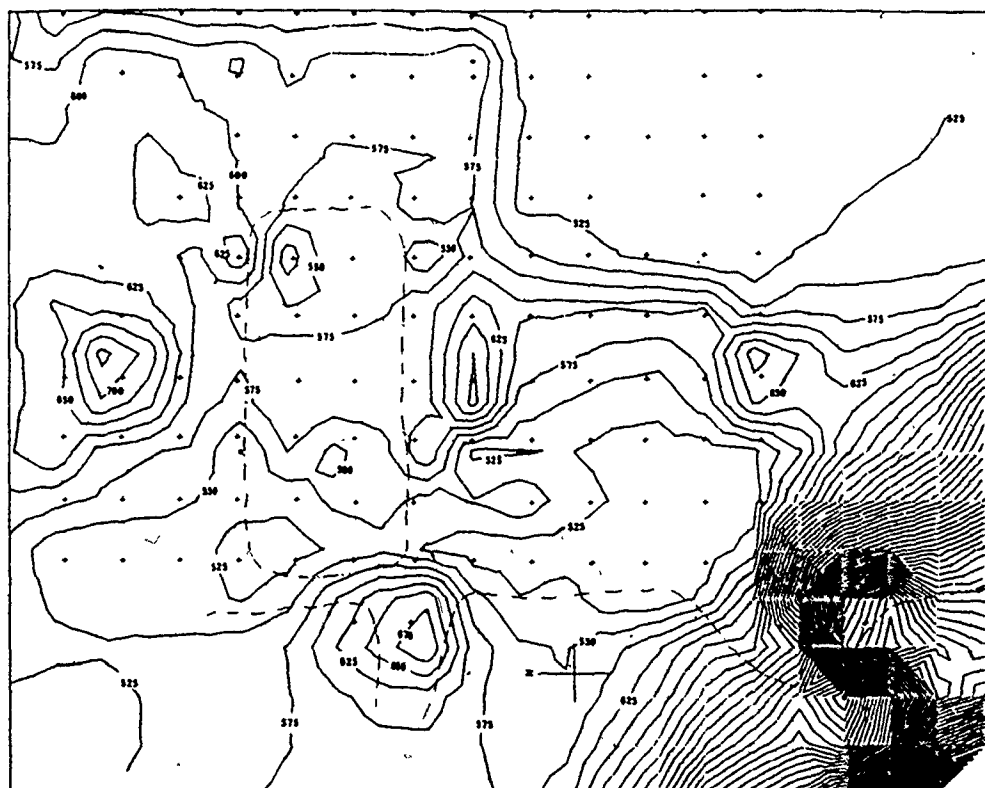


FIG. E.6 Nitrate in soil solution at 30 cm depth in the site area during continuous effluent flow, contour interval 0.00025, 10^{-5} moles/l

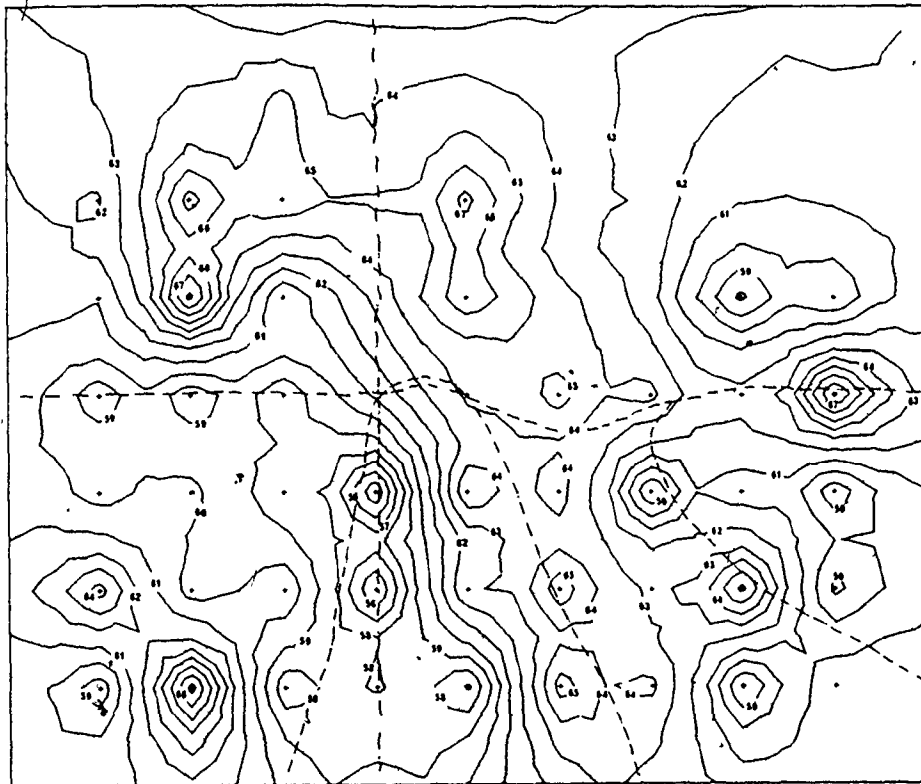


FIG. E.7 Soil pH in a transect through the septic bed soil 13 days after cessation of effluent flow, contour interval 0.1, 10^{-1} pH units

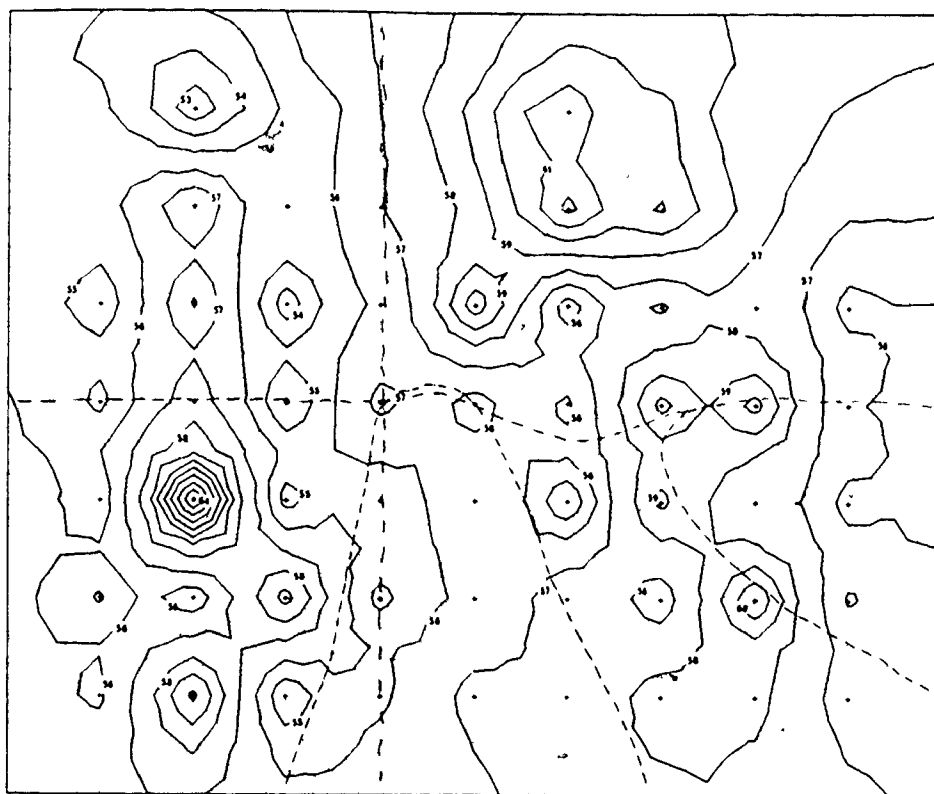


FIG. E.8 Soil pH in a transect through the septic bed soil 26 days after cessation of effluent flow, contour interval 0.1, 10^{-1} pH units

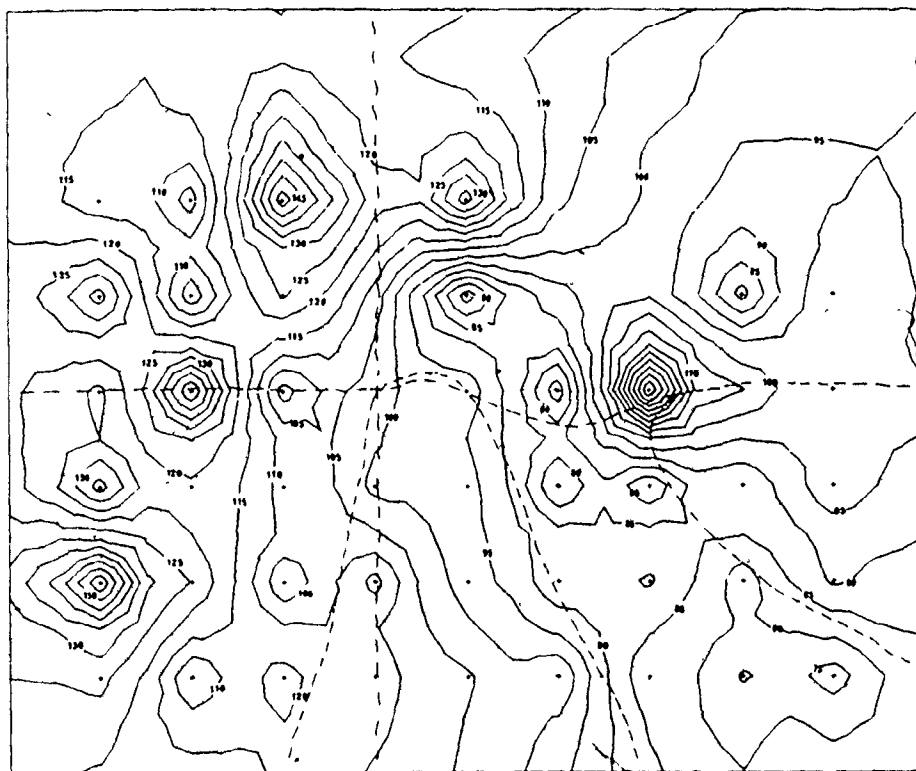


FIG. E.9 Sodium in solution in a transect through the septic bed soil 13 days after cessation of effluent flow, contour interval $0.0005, 10^{-4}$ moles/l

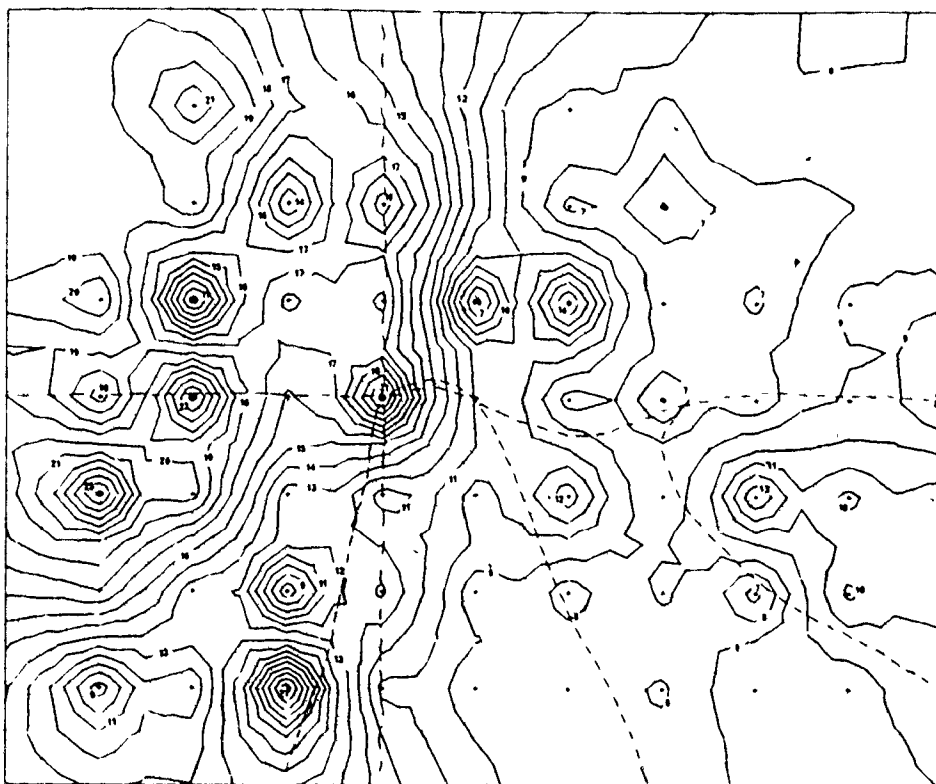


FIG. E.10 Sodium in solution in a transect through the septic bed soil
26 days after cessation of effluent flow, contour interval
 $0.001, 10^{-3}$ moles/l

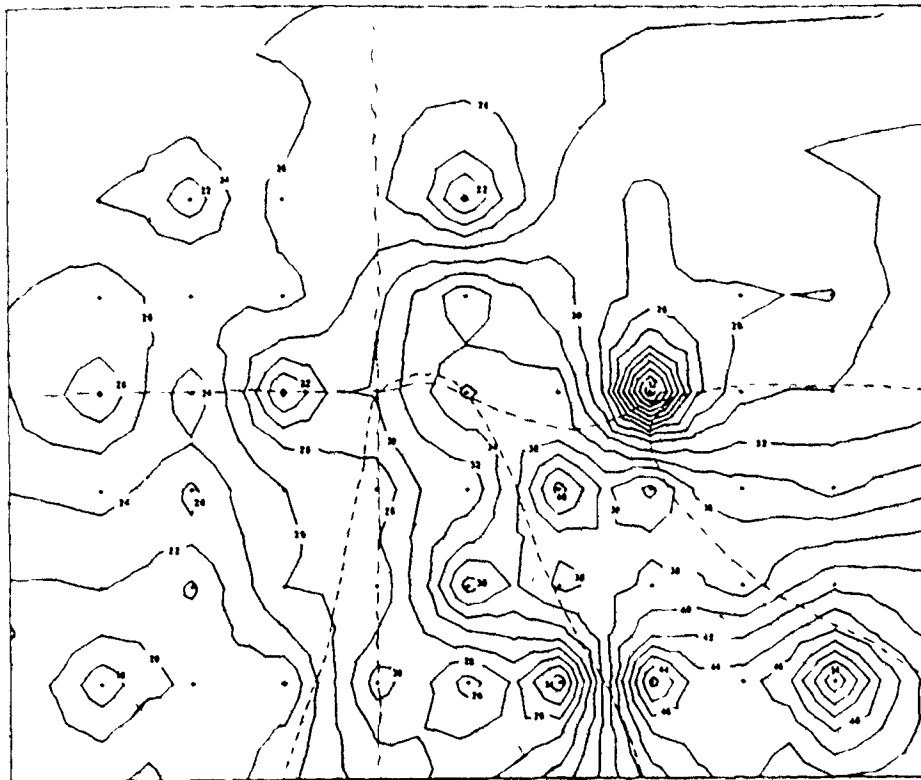


FIG. E.11 Soil moisture by weight in a transect through the septic bed 13 days after cessation of effluent flow, contour interval 2.0, 10 %

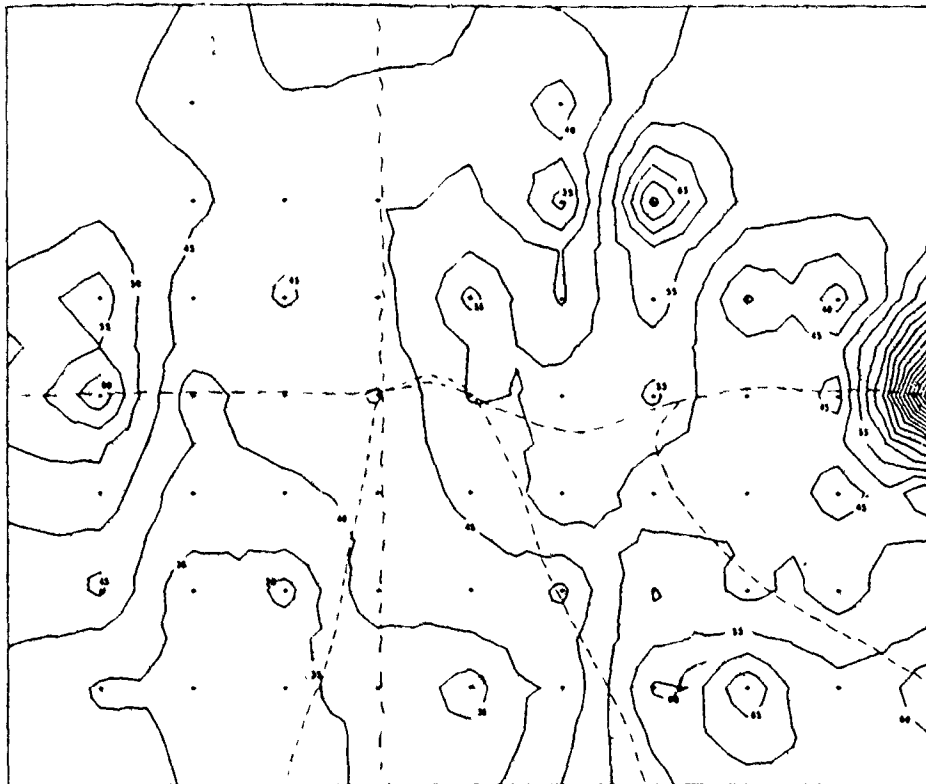
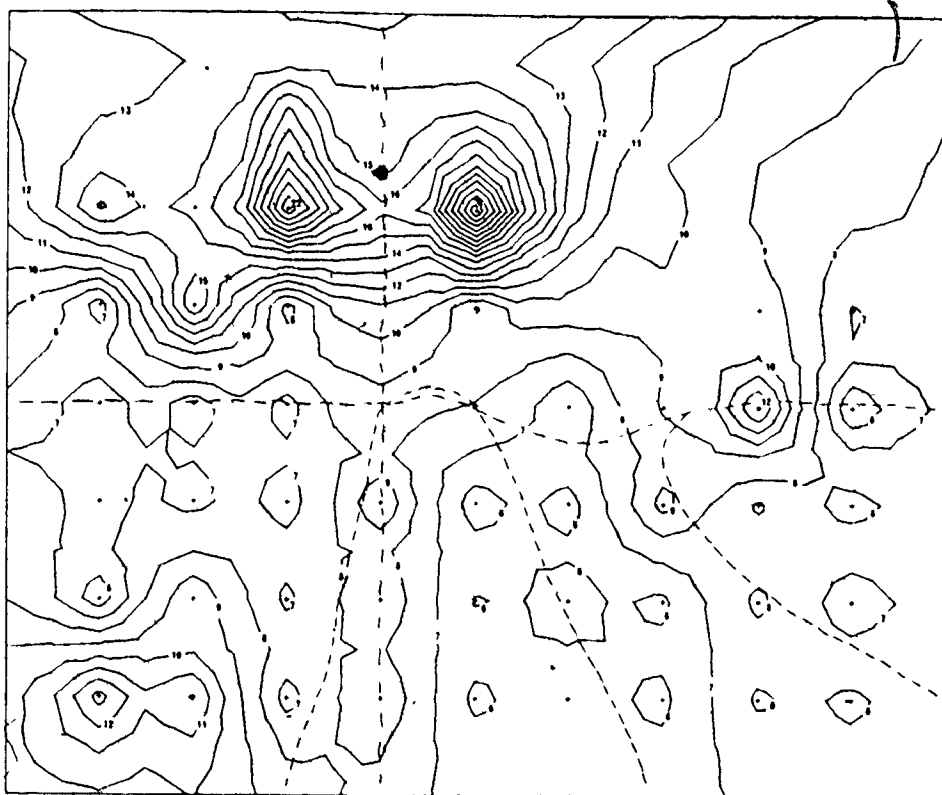


FIG. E.12 Soil moisture by weight in a transect through the septic bed 26 days after cessation of effluent flow, contour interval 5.0, 10 %



◆ FIG. E.13 Nitrate in solution in a transect through the septic bed soil 13 days after cessation of effluent flow, contour interval 0.001, 10⁻³ moles/l

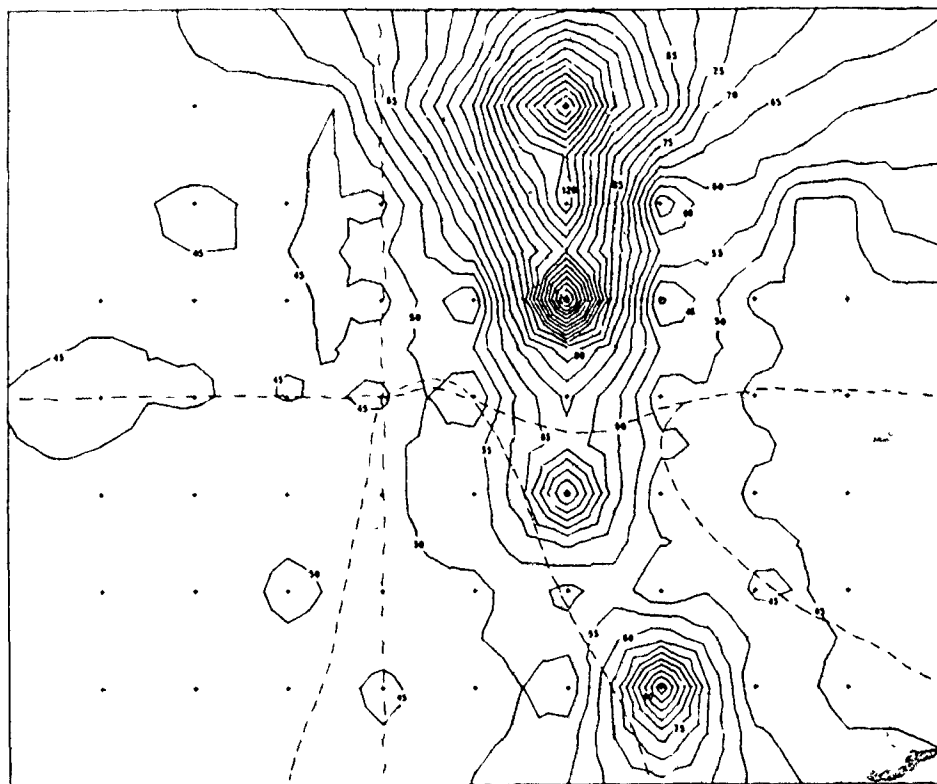


FIG. E.14 Nitrate in solution in a transect through the septic bed soil 26 days after cessation of effluent flow, contour interval $0.005, 10^{-4}$ moles/l

REFERENCES

- Anomine, S.; 1962. A review of research on redox potentials of paddy soils in Japan. *Soil Sci.*, 94: 6-13.
- Baker, J. L., Campbell, K. L., Johnson, H. P., Hanway, J. J.; 1975. Nitrate, phosphorus and sulfate in subsurface drainage water. *J. Environ. Quality*, 4: 406-412.
- Banin, A., and Amiel, A., 1970. A correlative study of the chemical and physical properties of a group of natural soils of Israel. *Geoderma*, 3: 185-198.
- Beatty, M. T., and Bouma, J., 1973. Application of soil surveys to selection of sites for on-site disposal of liquid household wastes. *Geoderma*, 10: 113-122.
- Beek, J., and Frissel, M. J., 1973. Simulation of Nitrogen Behaviour in Soils. PUDOC, Wageningen.
- Beek, J., and de Haan, F. A. M., 1974. Phosphate removal by soil in relation to waste disposal. *Int. Conf. on Land for Waste Management*, Ottawa, Canada: 77-86 (Le Droit, Ottawa).
- Bell, P., and Coombe, D., 1967. Strasburger's Textbook of Botany (Longmans, Suffolk).
- Bennett, E. R., Linstedt, K. D., Felton, J. T., 1975. Rural household water characteristics. *Proc. of Nat. Home Sewage Disposal Symposium*: 74-78 (St. Joseph, Michigan).
- Bolt, G. H., and Bruggenwert, M. G. M., 1976. Soil Chemistry A. Basic Elements (Elsevier, Amsterdam).
- Bouma, J., 1974. New concepts in soil survey interpretation for on-site disposal of septic tank effluent. *Soil Sci. Soc. Amer. Proc.*, 38: 941-946.
- Bouma, J., 1975. Innovative on-site soil disposal and treatment systems for septic tank effluent. *Proc. of Nat. Home Sewage Disposal Symposium*: 152-162 (St. Joseph, Michigan).
- Bouma, J., Converse, J. C., Magdoff, F. R., 1974a. Dosing and resting to improve adsorption beds. *Amer. Soc. Agr. Eng. Trans.*, 17: 295-298.

- Bouma, J., Converse, J. C., Otis, R. J., Walker, W. G., Ziebell, W. A., 1975. A mound system for on-site disposal of septic tank effluent in slowly permeable soils with seasonally perched water tables. *J. Environ. Quality*, 4: 382-388.
- Bouma, J., Converse, J. C., Ziebell, W. A., Magdoff, F. R., 1974b. An experimental mound system for disposal of septic tank effluent in shallow soils over creviced bedrock. *Int. Conf. on Land for Waste Management*, Ottawa, Canada: 367-377 (Le Droit, Ottawa).
- Bouma, J., and Denning, J. L., 1972. Field measurement of unsaturated hydraulic conductivity by infiltration through gypsum crusts. *Soil Sci. Soc. Amer. Proc.*, 36: 846-847.
- Bouma, J., Hillel, D. T., Hole, F. D., Amerman, C. R., 1971. Field measurement of unsaturated hydraulic conductivity by infiltration through artificial crusts. *Soil Sci. Soc. Amer. Proc.*, 35: 362-364.
- Bower, H., 1961. A double tube method for measuring hydraulic conductivity in soil in situ above a water table. *Soil Sci. Soc. Amer. Proc.*, 25: 334-339.
- Bower, H., 1962. Field determination of hydraulic conductivity above a water table with a double tube method. *Soil Sci. Soc. Amer. Proc.*, 26: 330-335.
- Bower, H., 1971. Land disposal of sewage effluents. *Proceedings of a Symposium on Nitrogen in Soil and Water, 1971*. Dept. of Soil Sci., Univ. Guelph, Guelph, Ontario.
- Brandes, M., Chowdhry, N. A., Chang, W. W., 1975. Experimental study on removal of pollutants from domestic sewage by underdrained soil filters. *Proc. of Nat. Home Sewage Disposal Symposium*: 29-36 (St. Joseph, Michigan).
- Brandt, G. H., Wolcott, A. R., Erickson, A. E., 1964. Nitrogen transformations in soil as related to structure, moisture and oxygen diffusion rate. *Soil Sci. Soc. Amer. Proc.*, 28: 71-75.
- Bremner, J. M., and Nelson, D. W., 1968. Chemical decomposition of nitrite in soils. *Trans. 9th Int. Congr. Soil Sci.*, 2: 495-503.
- Broadbent, F. E., 1971. Nitrogen in soil and water. *Proceedings of a Symposium on Nitrogen in Soil and Water, 1971*. Dept. of Soil Sci., Univ. Guelph, Guelph, Ontario.
- Buckman, H. O., and Brady, N. C., 1960. The Nature and Properties of Soils (MacMillan, New York).

- Bunting, B. T., and Campbell, J. A., 1976. Pedology Laboratory Manual: An Abbreviated Selection of Routine Techniques of Soil Analysis. Pedology Laboratory, Geography Department, McMaster University, Hamilton, Ontario.
- Buresh, R. J., and Moraghan, J. T., 1976. Chemical reduction of nitrate by ferrous iron. *J. Environ. Quality*, 5: 320-325.
- Burwell, R. E., Schuman, G. E., Saxton, K. E., Heinemann, H. G., 1976. Nitrogen in subsurface discharge from agricultural watersheds. *J. Environ. Quality*, 5: 325-329.
- Cady, F. B., and Bartholomew, W. V., 1961. Influence of low pO_2 on denitrification process and products. *Soil Sci. Soc. Amer. Proc.*, 25: 362-365.
- Campbell, G. S., 1974. A simple method for determining unsaturated conductivity from moisture retention data. *Soil Sci.*, 117: 311.
- Campbell, J. A., 1974. Some impacts of septic effluent on hydromorphic soils. M. Sc. thesis, McMaster University, Hamilton, Ontario.
- Chichester, F. W., 1977. Effects of increased fertilizer rates on nitrogen content of runoff and percolate from monolith lysimeters. *J. Environ. Quality*, 6: 211-217.
- Converse, J. C., Anderson, J. L., Ziebell, W. A., Bouma, J., 1975. Pressure distribution to improve soil adsorption systems. *Proc. of Nat. Home Sewage Disposal Symposium*: 152-162 (St. Joseph, Michigan).
- Deamish, R., 1972. Eutrophication, or where did all the algae come from? in Keep It Clean: A Manual for the Preservation of the Cottage Environment (Pollution Probe, Univ. of Toronto).
- Duffy, J., Chung, C., Boast, C., Franklin, M., 1975. A simulation model of biophysiochemical transformations of nitrogen in tile drained corn belt soil. *J. Environ. Quality*, 4: 477-486.
- Engelstad, O. P., ed., 1970. *Symposium on Nutrient Mobility in Soils: Accumulation and Losses*, SSSA Special Publication Series, No. 4 (Detroit, 1969).
- Gambrell, R. P., Gilliam, J. W., Weed, S. B., 1975a. Denitrification in subsoils of the North Carolina coastal plain as affected by soil drainage. *J. Environ. Quality*, 4: 311-316.
- Gambrell, R. P., Gilliam, J. W., Weed, S. B., 1975b. Nitrogen losses from soils of the North Carolina coastal plain. *J. Environ. Quality*, 4: 317-323.

- Gilbert, R. B., Robinson, J. B., Miller, J. B., 1974. Microbiology and nitrogen transformations of a soil recharge basin used for water renovation. Int. Conf. on Land for Waste Management, Ottawa, Canada: 87-96.
- Hesse, P. R., 1971. A Textbook of Soil Chemical Analysis (William Clowes and Sons Ltd., London).
- Hill, D. E., 1972. Waste water renovation in Connecticut soils. J. Environ. Quality, 1: 163-167.
- Hines, M., Favreau, R. E., 1975. Recirculating sand filter: an alternative to traditional sewage adsorption systems. Proc. of Nat. Home Sewage Disposal Symposium: 130-136 (St. Joseph, Michigan).
- Hodgman, C. D., ed., 1962. Handbook of Chemistry and Physics, 43rd ed. (The Chemical Rubber Publishing Co., Cleveland, Ohio).
- Jackson, W. A., Asumussen, L. E., Hauser, E. W., White, A. W., 1973. Nitrate in surface and subsurface flow from a small agricultural watershed. J. Environ. Quality, 2: 480-482.
- Jenny, H., 1961. Derivation of state factor equations of soils ecosystems. Soil Sci. Soc. Amer. Proc., 25: 385-388.
- Jones, E. E., Jr., 1975. Domestic water use in individual homes and hydraulic loading of and discharge from septic tanks. Proc. of Nat. Home Sewage Disposal Symposium: 89-103 (St. Joseph, Michigan).
- Karikari, T. J., Beer, C. E., Smith, R. J., 1975. Treatment of a residential septic tank effluent in an aerobic lagoon. Proc. of Nat. Home Sewage Disposal Symposium: 144-151 (St. Joseph, Michigan).
- Kerr, H. W., 1928. The nature of base exchange and soil acidity. J. Amer. Soc. Agron., 20: 309.
- Kilmer, V. J., Gilliam, J. W., Lutz, J. F., Joyce, R. T., Eklund, C. D., 1974. Nutrient losses from fertilized grassed watersheds in Western North Carolina. J. Environ. Quality, 3: 214-219.
- Kimble, J. M., Bartlett, R. J., McIntosh, J. L., Varney, K. E., 1972. Fate of nitrate from manure and inorganic nitrogen in a clay soil cropped to continuous corn. J. Environ. Quality, 1: 413-418.
- Kirkham, D., Powers, W. L., 1972. Advanced Soil Physics (Wiley Interscience, New York).

- Klausner, S. D., Zwerman, P. J., Ellis, D. F., 1974. Surface losses of soluble nitrogen and phosphorus under two systems of soil management. *J. Environ. Quality*, 3: 42-46.
- Klausner, S. D., Zwerman, P. J., Ellis, D. F., 1976. Nitrogen and phosphorus losses from winter disposal of dairy manure. *J. Environ. Quality*, 5: 47-49.
- Laak, R., Kolega, J. J., Corenza, B. J., Weinberg, M. S., 1975. Feasibility studies on utilizing sodium bicarbonate with septic tank systems. *Proc. of Nat. Home Sewage Disposal Symposium*: 202-209 (St. Joseph, Michigan).
- Lähde, E., 1969. Biological activity in some natural and drained peat soils with special reference to oxidation-reduction conditions. *Acta Forestalia Fennica*, Vol. 94.
- Lance, J. C., 1972. Nitrogen removal by soil mechanisms. *J. Water Pollution Control Fed.*, 44: 1352-1361.
- Lance, J. C., and Whisler, F. D., 1972. Nitrogen balance in soil columns intermittently flooded with secondary sewage effluent. *J. Environ. Quality*, 1: 180-185.
- Lance, J. C., and Whisler, F. D., 1974. Nitrogen removal during land filtration of sewage water. *Int. Conf. on Land for Waste Management*, Ottawa, Canada: 174-182 (Le Droit, Ottawa).
- Lance, J. C., Whisler, F. D., Bower, H., 1973. Oxygen utilization in soils flooded with sewage water. *J. Environ. Quality*, 2: 345-350.
- Lance, J. C., Whisler, F. D., Rice, R. C., 1976. Maximizing denitrification during soil filtration of sewage water. *J. Environ. Quality*, 5: 102-107.
- Larsen, S., 1974. Food. *Neth. J. Agric. Sci.*, 22: 270-274.
- Loehr, R. C., 1970. Changing practices in agriculture and their effect on the environment. *Critical Reviews in Environmental Control* 1, 1: 59-99.
- Magdoff, F. R., and Bouma, J., 1975. The development of soil clogging in sands leached with septic tank effluent. *Proc. of Nat. Home Sewage Disposal Symposium*: 37-47 (St. Joseph, Michigan).
- Magdoff, F. R., Keeney, D. R., Bouma, J., Ziebell, W. A., 1974. Columns representing mound-type disposal systems for septic tank effluent: II, nutrient transformations and bacterial populations. *J. Environ. Quality*, 3: 228-233.

- Mann, L. D., Focht, D. D., Joseph, H. A., Stolzy, L. H., 1972. Increased denitrification in soils by additions of sulfur as an energy source. *J. Environ. Quality*, 1: 329-332.
- Marshall, C. E., 1964. The Physical Chemistry and Mineralogy of Soils: Volume 1: Soil Materials (Wiley, New York).
- Martel, Y. A., and Paul, E. A., 1974. Effects of cultivation on the organic matter of grassland soils as determined by fractionation and radiocarbon dating. *Can. J. Soil Sci.*, 54: 419-426.
- McKeague, J. A., ed., 1976. Manual on Soil Sampling and Methods of Analysis. Subcommittee of Canada Soil Survey Committee on Methods of Analysis.
- Metson, A. J., 1961. Methods of Chemical Analysis for Soil Survey Samples, Soil Bureau Bulletin 12, New Zealand.
- Nagelberg, I., Braddock, L. E., Barbero, G. J., 1969. Divalent phosphate electrode. *Science*, 166: 1403-1404.
- Nicholls, K. H., and MacCrimmon, H. R., 1974. Nutrients in subsurface and runoff waters of the Holland Marsh, Ontario. *J. Environ. Quality*, 3: 31-35.
- Nie, N. H., Hull, C. H., Jenkins, J. G., Steinbrenner, K., Bent, D. H., 1975. Statistical Package for the Social Sciences (McGraw-Hill, New York).
- Novak, L. T., and Adriano, D. C., 1975. Phosphorus movement in soils: soil-orthophosphate reaction kinetics. *J. Environ. Quality*, 4: 261-266.
- Novak, L. T., Adriano, D. C., Coulman, G. A., Shah, D. B., 1975. Phosphorus movement in soils: theoretical aspects. *J. Environ. Quality*, 4: 93-99.
- Otis, R. J., Boyle, W. C., Sauer, D. K., 1975. The performance of household wastewater treatment units under field conditions. *Proc. of Nat. Home Sewage Disposal Symposium*: 191-201 (St. Joseph, Michigan).
- Porcella, D. B., and Bishop, A. B., 1976. Comprehensive Management of Phosphorus Water Pollution (Ann Arbor Science, Ann Arbor, Michigan).
- Pratt, P. F., Jones, W. W., Hunsaker, V. E., 1972. Nitrate in deep soil profiles in relation to fertilizer rates and leaching volume. *J. Environ. Quality*, 1: 97-102.

- Reneau, R. B., Jr., and Pettry, D. E., 1975. Movement of coliform bacteria from septic tank effluent through selected coastal plain soils of Virginia. *J. Environ. Quality*, 4: 41-44.
- Reneau, R. B., Jr., and Pettry, D. E., 1976. Phosphorus distribution from septic tank effluent in coastal plain soils. *J. Environ. Quality*, 5: 34-39.
- Reneau, R. B., Jr., Elder, J. H., Jr., Pettry, D. E., Weston, C. W., 1975. Influence of soils on bacterial contamination of a watershed from septic sources. *J. Environ. Quality*, 4: 249-252.
- Research Branch, Canada Department of Agriculture and the Ontario Agricultural College, 1962. Soil Survey of Parry Sound District: Report No. 31 of the Ontario Soil Survey.
- Rich, L. G., 1973. Environmental Systems Engineering (McGraw-Hill, Toronto).
- Rose, C. W., 1966. Agricultural Physics (Pergamon, Toronto).
- Rummell, R. J., 1970. Applied Factor Analysis (Northwestern University Press, Evanston).
- Ryden, J. C., Syers, J. K., Harris, R. F., 1973. Phosphorus in runoff and streams. *Advances in Agronomy*, 25: 1-45.
- Schuman, G. E., Burwell, R. E., Piest, R. F., Spomer, R. G., 1973. Nitrogen losses in surface runoff from agricultural watersheds on Missouri Valley loess. *J. Environ. Quality*, 2: 299-302.
- Schwiesow, W. F., 1975. Home sewage disposal systems--past and present. *Proc. of Nat. Home Sewage Disposal Symposium*: 2-8 (St. Joseph, Michigan).
- Shah, D. B., Coulman, G. A., Novak, L. T., Ellis, B. G., 1975. A mathematical model for phosphorus movement in soils. *J. Environ. Quality*, 4: 87-92.
- Sikora, L. J., and Keeney, D. R., 1975. Laboratory studies on stimulation of biological denitrification. *Proc. of Nat. Home Sewage Disposal Symposium*: 64-73 (St. Joseph, Michigan).
- Skinner, F. A., 1968. The anaerobic bacteria in soil. *An International Symposium*, eds., T. R. G. Gray, D. Parkinson (Univ. of Toronto Press, Toronto).
- Soil Conservation Service, U. S. Dept. of Ag., 1967. Soil Survey Laboratory Methods and Procedures for Collecting Soil Samples, Report No. 1 (U. S. Govt. Printing Office, Washington, D. C.).

- Swahney, B. L., and Hill, D. E., 1975. Phosphate sorption characteristics of soils treated with domestic wastewater. *J. Environ. Quality*, 4: 342-346.
- Taylor, S. A., and Ashcroft, G. L., 1972. Physical Edaphology (Freeman, San Francisco).
- Turner, F. T., and Patrick, W. H., Jr., 1968. Chemical changes in waterlogged soils as a result of oxygen depletion. *Trans. 7th Intern. Congr. Soil Sci.*, : 53-63.
- Varazashvili, L. E., Lytaye, I. A., Petrova, M. V., 1976. Statistical soil moisture parameters as functions of the soil moisture potential and method for determining them. *Sov. Soil Sci.*, 8: 98-104.
- Viraraghavan, T., and Warnock, R. G., 1974. Treatment through soil of septic tank effluent. *Int. Conf. on Land for Waste Management*, Ottawa, Canada: 214-223 (Le Droit, Ottawa).
- Viraraghavan, T., and Warnock, R. G., 1975. Treatment efficiency of a septic tile system. *Proc. of Nat. Home Sewage Disposal Symposium*: 48-57 (St. Joseph, Michigan).
- Walker, W. G., Bouma, J., Keeney, D. R., Magdoff, F. R., 1973a. Nitrogen transformations during subsurface disposal of septic tank effluent in sands: I, soil transformations. *J. Environ. Quality*, 2: 475-480.
- Walker, W. G., Bouma, J., Keeney, D. R., Olcott, P. G., 1973b. Nitrogen transformations during subsurface disposal of septic tank effluent in sands: II, groundwater quality. *J. Environ. Quality*, 2: 521-525.
- Wall, G. J., and Webber, L. R., 1970. Soil characteristics and subsurface sewage disposal. *J. of Public Health*, 61: 47-54.
- Walter, M. F., 1974. Nitrate movement in soil under early spring conditions. NTIS publication PB-240 094, Ph. D. thesis, Univ. of Wisconsin.
- Webber, L. R., 1974. Soil properties and processes in relation to waste recycling and disposal. *Int. Conf. on Land for Waste Management*, Ottawa, Canada: 61-67 (Le Droit, Ottawa).
- Whisler, F. D., Lance, J. C., Lineburger, R. S., 1974. Redox potentials in soil columns intermittently flooded with sewage water. *J. Environ. Quality*, 3: 68-74.
- White, E. M., 1973. Water leachable nutrients from frozen or dried prairie vegetation. *J. Environ. Quality*, 2: 104-107.

de Wit, C. T., and van Keulen, H., 1972. Simulation of Transport Processes in Soils. Centre for Agricultural Publishing and Documentation, Wageningen.

Witt, M., Siegrist, R., Boyle, W. C., 1975. Rural household wastewater characteristics. Proc. of Nat. Home Sewage Disposal Symposium: 79-88 (St. Joseph, Michigan).

Ziebell, W. A., Nero, D. H., Denninger, J. F., McCoy, E., 1975. Use of bacteria in assessing waste treatment and soil disposal systems. Proc. of Nat. Home Sewage Disposal Symposium: 58-63 (St. Joseph, Michigan).

Zwimmerman, P. J., and de Haan, F. A. M., 1973. Significance of the soil in environmental improvement--a review. The Science of the Total Environment, 2: 121-155.