Effects of Agricultural Practices on Water Quality as Related to Adjustments of HSPF Parameters, a Literature Review:

PARAMETERS AND CONCEPTS FOR MODELING

TILLAGE PRACTICES
ANIMAL WASTE MANAGEMENT SYSTEMS
AND
VEGETATED FILTER STRIPS

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PREFACE

The Interstate Commission on the Potomac River Basin (ICPRB) and the Maryland Department of the Environment (MDE) are cooperating in a project to mathematically model nitrogen, phosphorus and sediment delivery from the Monocacy River watershed to the Potomac River and to evaluate agricultural nonpoint source pollution reduction strategies for the basin. These strategies prominently feature agricultural "Best Management Practices" or BMPs.

As part of this study, the following literature review was prepared, summarizing field studies of how selected BMPs and other agricultural practices modify physical processes contributing to water pollution. The review is divided into three chapters:

1. Parameters and Concepts for Modeling Tillage Practices

2. Water Quality Impacts of Animal Waste Management Systems

3. Efficacy of Vegetative Filter Strips and Buffer Strips

Emphasis was given to research related to farming practices in the Monocacy watershed, especially studies conducted in the eastern piedmont physiographic region or in the coastal plain. Information on both HSPF parameter values and discussions of BMPs' effects on nutrient and sediment loadings were presented.

CHAPTER 1. PARAMETERS AND CONCEPTS FOR MODELING TILLAGE PRACTICES

INTRODUCTION

EXAMINATION OF ACCEPTED BELIEFS

BMPs have been used successfully for years to reduce soil erosion from farmland. However, their effects on water quality (both ground water and surface water quality) are still being evaluated. The literature leaves some questions unanswered and other answers controversial.

The Maryland Agricultural Water Quality Management Program, the document that outlines Maryland's official approach to managing water pollution from agriculture, (Maryland State Soil Conservation Committee, 1987) states that:

> "The amounts of phosphorus that moves in true solution [from a field] are very small, as are the amounts that move downward through the soil by leaching. Consequently, these sources of phosphorus can be neglected in comparison to the much larger amounts transported in forms absorbed on eroded sediments. Thus the BMPs that are effective in controlling phosphorus movement are those which reduce erosion and the amount of water which flows directly over the soil surface (runoff) with enough velocity to carry sediment.
>
> ... Nitrogen, in contrast to phosphorus, is lost primarily through downward leaching in the soluble nitrate form. Movement of nitrogen in various forms in surface runoff can occur, but these losses are typically negligible relative to those occurring as nitrate in subsurface water flow. Almost all the nitrogen lost in Maryland is in the form of nitrate, which is leached downward through the soil profile and transported to receiving waters in subsurface flow ... Best management practices that control loss of one nutrient do not necessarily control loss of the other."

This is the thinking behind the partitioning of BMPs in Table 1, taken from the above reference.

Examining whether and under what conditions these claims are supported by the experimental literature is one of the major purposes of the MDE-ICPRB literature review.

Table 1
Best Management Practices for Nutrient Loss Reduction
Maryland State Soil Conservation Committee, 1987

A. BMPs for reducing losses of nitrogen and phosphorus from field crops
  - Proper nutrient application rates
  - Appropriate timing of nutrient application
  - Appropriate method of nutrient application
  - Reduced tillage practices
  - Crop rotations
  - Cover crop
  - Critical area seeding
  - Pond

B. BMPs effective in reducing phosphorus loss from field crops
  - Contour farming
  - Strip cropping
  - Grass filter strip
  - Natural vegetation filter strip
  - Terrace
  - Diversion
  - Grassed waterway
  - Sediment control basin

THE PROBLEM OF QUANTIFYING QUALITATIVE STATEMENTS

The second purpose of this document is to provide guidance and support for the choices made in modeling agricultural practices within the framework of HSPF, so that the modeled processes will reflect the agricultural reality as closely as possible.

The problem of choosing parameter estimates for HSPF is both one of understanding the physical processes that are impacted by agriculture as well as determining the extent of the impact.

An excellent analysis of the first part of this problem is found in Donigian et al., 1983. They modeled standard agricultural practices and BMPs in the Four Mile Creek watershed in Iowa using HSPF, and tabulated the expected effects of these practices on the HSPF parameters controlling runoff, erosion, and chemical processes.

An excerpt from their qualitative assessment, covering BMPs being considered in the Monocacy modeling project, is found in Table 2.

Table 2
Relative Impact of Selected BMPs on Processes Affecting Water Quality
Donigian et al., 1983

Process	NO-TILL	MIN-TILL	NON-STRUCTURAL MEASURES CONTOURING	CONTOUR STRIPS	SPRING PLOWING	WINTER COVER	PERMANENT MEADOW
Runoff Related Process							
Interception	+	+	0	0	+	+	+
Depression storage	0,+	0,+	+	+	+,0	0	+
Soil moisture storage	0,+	0,+	0	0	0	0	0,+
Overland flow	-	-	-	-	-	-	-
Infiltration	+	+	0	+	+,0	+?	+
Subsurface Flow	+	+	0,+	0,+	+	+	+
Evapotranspiration	-	0	0	0	0	+,0	+
Sediment Related Process							
Detachment by rainfall	-	-	0	-,0	-	-	-
Detachment by overland flow	-	-	-	-	-	-	-
Transport by overland flow	-	-	-	-	-	-	-
Delivery to stream	0	0	0	0	0	0	0
Gully erosion/scour	-	-	-	-	0,-	0,-	-
Production of sediment fines	-	-	0	-,0	-,0	+,-	-
Aggregation/compaction	-	-	0	0	0?	-,0	-,+
Chemical Related Processes							
Surface & subsurface availability	+	+	0	-	0,+	0,+	-
Transformation	0,-	0,-	0	0	+?	0	0
Uptake/Degradation	0?	0	0	0	0	+	+,-?
Absorption/desorption	+?	+?	0	+?	0	+?	0
Soil Temperature	-,+	-,+	0	0	-,+	0,+	0,+
Soil Bulk Density	+	+	0	0	0	0	+?

Process	STRUCTURAL MEASURES DIVERSIONS	GRASSED WATERWAYS	FILTER STRIPS	INPUT MANAGEMENT OPTIONS NONEXCESSIVE APPLICATIONS	CHEMICAL TILL-IN
Runoff Related Process					
Interception	0	0.+	0,+	0	0
Depression storage	+,0	0	0	0	+
Soil moisture storage	0	0	0	0	0
Overland flow	+	-	-?	0	-
Infiltration	-?	+?	+?	0	+
Subsurface Flow	-?	0	0	0	?
Evapotranspiration	0	0	0	0	+

Table 2, continued

Process	DIVERSIONS	GRASSED WATERWAYS	FILTER STRIPS	NONEXCESSIVE APPLICATIONS	CHEMICAL TILL-IN
Sediment Related Process					
Detachment by rainfall	0	-	-	0	+
Detachment by overland flow	-	-	-	0	+
Transport by overland flow	+?	-	-	0	+
Delivery to stream	-,+	-	-	0	0
Gully erosion/scour	-	-	-	0	0
Production of sediment fines	0	0	0	0	+
Aggregation/compaction	0	0	0	0	-
Chemical Related Processes					
Surface & subsurface availability	0	-	-	-	-
Transformation	0	0	0	0	0
Uptake/Degradation	0	0	0	0	0
Absorption/desorption	0	0?	0?	0	+?
Other					
Soil Temperature	0	0	0	0	+
Soil Bulk Density	0	0	0	0	-

<u>KEY</u>

+	Practice increases or enhances process with respect to conventional practices
-	Practice decreases or retards process with respect to conventional practices
0	Practice does not result in effects different from those of conventional practices
?	Questionable relationship between practice and process
0,+	Effect (0 or +) of practice is determined by it case-specific implementation

The table breaks down BMPs into three categories: non-structural measures, structural measures, and input management options. Non-structural measures include cultural agronomic practices that protect the land surface from erosion. Structural practices require some construction on the watershed and continuing maintenance, as opposed to non-structural measures which attempt to control the source of the pollution rather than arrest its movement. The third category, input management options modify the timing, amounts, and/or placement of chemicals to reduce their availability for transport.

The table shows that non-structural measures consistently reduce overland flow and increase infiltration and subsurface flow. Non-structural practices that include leaving crop residues on the field or close-grown crops or meadow also increase interception and surface depression storage.

Structural measures generally lead to some reduction in overland flow with a corresponding increase in infiltration and subsurface flow. Their major impact is on sediment-associated pollutants by modification of the processes of sediment detachment, sediment transport by overland flow, and sediment delivery from the field.

The goal of nutrient management options is to reduce the amount of chemical that can wash off a field by reducing chemical inputs, increasing the efficiency of applied chemicals, or by reducing the amounts on the land surface by incorporation. It should be noted that practices that require chemical incorporation by tillage will cause changes in runoff and sediment processes from the additional tillage operation.

From Table 2 we have an idea of which BMPs affect nutrient concentrations in runoff and ground water, and what physical processes mediate the relationship between land use and pollution. Now it remains to quantify the relevant rates and effects, to determine the temporal variation in the effects, and to examine the variability reported for the various BMPs' effectiveness.

In addition to overall effects, the modeler must consider effects that are season specific. Tillage systems alter soil characteristics at different times of the year, making a field more or less vulnerable to the forces which move matter. It is important when selecting model parameters not only to identify the significant factors, but also to represent them in adequate temporal detail.

It is hoped that the HSPF practitioner and the water quality manager will use this document

(1) to clarify issues surrounding the water quality effects of tillage systems,

(2) to provide parameters from field studies for starting values in model calibration

(3) to identify the important details necessary for a responsive numerical representation of the agricultural practices, and

(4) to provide flux measurements for comparison to model outputs.[1]

I. TERMINOLOGY

A. CONVENTIONAL TILLAGE

Conventional tillage normally refers to a maximum tillage program consisting of both primary tillage, to invert the soil surface (moldboard plowing), and secondary tillage, to break up clods and leave a smooth surface for planting (disking, harrowing). Cultivation can be performed for weed control. A typical conventional tillage program might consist of the following field operations:

1) Moldboard plowing
2) Disking once or twice
3) Harrowing or field cultivating once or twice
4) Planting and fertilizing
5) Cultivating or using a rotary hoe once or twice
6) Spraying with herbicides once.

The conventional system performs a complete turnover of soil in the plow layer (15 to 20 cm), eliminating residue and fertilizer from the soil surface. The loose structure of the exposed soil provides material that is easily detached by impacting raindrops and surface flow. Surface crusting is usually observed after the first good storm, subsequently reducing the erodibility of the conventionally tilled surface.

B. CONSERVATION TILLAGE

The Soil Conservation Service defines conservation tillage as any planting and tillage system that retains at least 30% crop residue cover on the soil surface after planting. These systems include no-tillage, ridge tillage, strip tillage, mulch tillage, and minimum tillage, among others.

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[1] Flux estimates (with one exception) from <u>modeling</u> studies are intentionally omitted to avoid the perpetuation of modeling errors. This paper supplies <u>empirical</u> information for evaluating and constructing model estimates.

A group of conservation tillage practices are classed as reduced or minimal till. They are intermediate between no-till and conventional till in the amount of surface disturbance they cause.

Brief descriptions of the manifestations of several conservation tillage practices follow. More detailed studies will be presented later comparing the surface and subsurface soil characteristics resulting from different tillages.

1. NO-TILLAGE

In no-tillage seedbed preparation and planting are completed in one trip over the field with no preplant tillage. Less than 10% of the soil surface is disturbed, leaving a protective cover of crop residue on 90% of the surface. There is usually no cultivation except in systems where planting is done on ridges formed by cultivation, or where serious weed problems develop and herbicides do not provide adequate control.

Much of the experimental data that will be presented in this review will describe long term experiments involving continuous production of a single crop. No-till corn is more likely to be grown in some type of rotation with soybeans, forage legumes, or forage grasses, because of insect, weed, and disease control considerations. Though the characteristic no-till soil profile only develops after about three years, studies of shorter duration are included in this report because they are representative of what is actually found in a watershed.

2. REDUCED TILLAGE

There many different types of reduced tillage. They are identified by the main types of equipment they use, and leave the field in a range of conditions. A few of the major reduced tillage systems are described below.

The COULTER system involves tillage of only a 5 to 8 cm row and leaves most of the corn residue and surface applied fertilizer undisturbed on the soil surface. (Use of a coulter is often a component of no-tillage.) The coulter cuts a slit for seed placement. Corn stalk residue on the surface protects the soil surface against raindrop impact and reduces runoff velocities so that the capacity to detach and transport soil particles is reduced.

In RIDGE TILL seeds are planted on ridges formed by cultivation. After harvest the top of the old row ridge is cut off with a wide sweep or set of small disks, pushing soil and residue aside and leaving a protective cover of crop residue and clods between the rows in a 40 cm strip. At least one cultivation is performed to rebuild the ridge for planting the next year's crop. The soil surface is exposed to raindrop impact and surface flow, but to a lesser degree than in the conventional system. Surface runoff amount and velocity are reduced with ridge till because the clods and residue in the between row areas serve as barriers to runoff. When done on the contour, the ridges of ridge till act as small terraces, increasing surface storage, increasing infiltration, and slowing runoff velocity.

The CHISEL system is a deep tillage operation which modifies soil compactness to a depth of 20 cm over a narrow row (4 cm width). This tends to leave much of the stalk residue and fertilizer at or near the surface of the soil profile. A twisted shank chisel will bury up to 75% of the residue on the field surface, but a straight shank chisel will cover only about 20% per trip. The unincorporated residue protects the soil against raindrop impact and surface flow, while the loose soil structure promotes high water infiltration rates. When done across slope, chisel plowing produces ridges which pond runoff, allowing sediment to settle out. However, when these ponds are breached during larger storm events, the water retention properties of the ridges are lost.

The DISK system loosens soil, breaks up residue and mixes soil, residue, and fertilizer to a depth of 10 cm over the entire area.

II. SURFACE PROPERTIES AFFECTED BY VARIOUS TILLAGES

A. RESIDUE COVER

1. QUANTIFYING COVER

The amount of residue cover remaining on a field is usually quantified in one of three ways:

1) the percent of the surface <u>area</u> covered with residue
2) the percent of the harvested residue <u>weight</u> which is re-applied to the field (this can exceed 100%)
3) the weight of residue applied.

Several authors estimated the residue cover (area) left after planting corn under various tillage regimes (Table 3). Lindstrom and Onstad (1984) measured cover for two harvesting practices, "harvesting for grain" (reside left) and "harvesting for silage" (70% residue weight removed).

Table 3
Residue Left after Corn Planting by Different Tillage Systems

	Römkens et al., 1973	Johnson et al., 1979	Laflen et al., 1978	Griffith et al., 1977	Lindstrom & Onstad 1984 (for grain)	Lindstrom & Onstad 1984 (silage)	Barisas et al., 1978	Siemens & Oschwald, 1978
			% Cover					
Conventional	<1	2	3	1	0	0	5	3.5
Chisel	38			19	33 ± 8	14 ± 6	19	54
Ridge till	20				64 ± 7	21 ± 3	35	
Disk	43-62			13				85
Coulter	75						55	54
Till-plant		11	20	8				
No-till		59	45	76				

Burr et al. (1987) compared the residue (area) left by nine tillage and planting systems for soybeans (Table 4). Only no-till and blade plow plant systems left greater than 30% residue cover after planting. The field cultivate-plant and blade plow-till plant systems retained between 20 and 30% residue cover. Tillage systems which used disking or chisel plowing retained less than 20% residue cover.

Table 4
Residue Cover Remaining After Planting Soybeans
for Nine Tillage and Planting Systems
Burr et al., 1987

Tillage and Planting System	% Remaining Residue Cover[1]
No-till/Plant	62.3 a
Blade Plow/Plant	31.7 b
Field Cultivate/Plant	23.6 c
Blade Plow/Till/Plant	22.9 c
Till/Plant	18.5 d
Disk/Plant	17.8 d
Disk/Field Cultivate/Plant	16.1 d e
Chisel Plow/Disk/Plant	13.2 e
Disk/Disk/Plant	13.0 e

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[1] Residue values with different letters are significantly different with Tukey's test at the 10% significance level

Dickey et al. (1985) in a similar study found that only no-till consistently left more than a 20% residue cover following soybean harvest.

Gilley et al. (1986) give empirical relationships for converting residue mass to percent cover for sorghum and soybean residue in metric tons per hectare (t/ha). Coefficients of determination (r^2) for equations 1 and 2 were 0.96 and 0.94, respectively.

$$\text{Sorghum \% surface cover} = 100(1 - \exp(-0.091 \text{ residue mass})) \quad [1]$$

$$\text{Soybean \% surface cover} = 100(1 - \exp(-0.135 \text{ residue mass})) \quad [2]$$

2. RESIDUE EFFECTS ON WATER QUANTITY AND QUALITY

That residue cover protects the soil surface from soil erosion is well established (Gilley et al., 1986 for a literature summary). Infiltration is often enhanced, reducing runoff.

One-shot simulation studies (that is, those featuring a single application of rainfall) have limited applicability because, over a crop season, residue cover is modified by decomposition and changes in the growing crop. Rainfall simulator studies are usually targeted to the period immediately after planting, capturing conditions which change dramatically with canopy closure.

Lindstrom and Onstad (1984) examined the effect of residue removal on runoff volume and soil loss, immediately after planting for continuous corn (third year) on a 2% slope Barnes loam (fine-loamy, mixed, Udic Haploboroll).

The authors compared three tillage systems: (1) conventional: fall moldboard plow, spring disk, (2) reduced till: fall chisel plow, spring disk, and (3) ridge till: a modified no-till, consisting of cultivation after harvest of the previous crop to establish ridges for planting.

Tillage was not done after harvest. Seventy percent of the corn stover was harvested from one half of each plot area for a test of the effect of residue harvesting during the <u>brief</u> period 6 to 17 days after the planting of the next corn crop.

Their experiment involved two artificial storms immediately after planting, separated by a 24 hour rest period. Measurements were taken three times in two days:

(1) Before the dry run. (This measurement gave soil properties for the antecedent moisture condition.)

(2) After the dry run. (The dry run consisted of applying 5.6 cm of simulated rainfall to the plots in 1 hr.) and

(3) After the wet run. (The wet run took place 24 hr after the first simulated rainfall. It consisted of a 4.2 cm application in 45 minutes).

The effects of tillage and residue harvesting on runoff and soil loss during seedbed crop stage[2] are summarized in Tables 5 and 6.

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[2] Crop stages commonly referred to are: (a) Fallow: inversion plowing to secondary tillage, (b) Seedbed: secondary tillage for seedbed preparation to 10% canopy cover, (c) Crop Stage 1, "plant establishment": seedbed to 50% canopy cover, (d) Crop Stage 2, "development": 50% to 75% canopy cover, (e) Crop Stage 3, "maturing crop": 75% cover to harvest, and (f) Crop Stage 4, "residue and stubble": harvest to plowing or new seeding.

Table 5
Water Runoff, Tillage, and Residue Harvest Relationships During Seedbed On a Barnes Loam
Lindstrom and Onstad, 1984

Tillage	Dry Run	Wet Run
	Runoff, cm	
Residue not harvested		
Conventional	0.21 ± 0.18	2.02 ± 0.19
Chisel tillage	0.16 ± 0.17	2.04 ± 0.19
Ridge till	2.04 ± 0.46	2.23 ± 0.48
70% Residue harvested		
Conventional	0.17 ± 0.11	1.87 ± 0.19
Chisel tillage	0.27 ± 0.18	2.31 ± 0.38
Ridge till	2.28 ± 0.50	2.89 ± 0.33
LSD (0.05)	0.46	0.33

There is a clear difference between wet and dry runs. When the residue was not harvested, the ridge-till treatment produced more runoff than the other treatments during the dry run, but after the wet run (when the Ap horizon was saturated) the differences disappeared. When the previous season's residue had been harvested, however, the difference between tillages was detectable (though not as dramatic) even after the wet run.

Table 6
Soil Loss, Tillage, and Residue Harvest Relationships During Seedbed
Lindstrom and Onstad, 1984

Tillage	Dry Run	Wet Run
	Soil Loss, kg/ha	
Residue not harvested		
Conventional	1590 ± 520	3190 ± 1150
Chisel tillage	780 ± 200	1750 ± 700
Ridge till	1600 ± 1100	2090 ± 880
70% Residue harvested		
Conventional	1320 ± 140	3660 ± 1100
Chisel tillage	1400 ± 710	2520 ± 1520
Ridge till	4590 ± 2350	6660 ± 1040
LSD (0.05)	1670	1630

discernable difference in soil loss among the harvesting treatments on either the dry run or the wet run, however, when 70% of the residue was removed, the ridge-till plot lost more soil (during seedbed) than the others, on both the wet and dry run. After the wet run, rills in the inter-row area of the ridge-till/residue removed plots were observed. Differences between soil loss during seedbed by conventional and reduced tillage plots are not seen.

Barisas et al. (1978), Siemens and Oschwald (1974) and Römkens et al. (1973) found that conservation tillage practices were ineffective in reducing the loss of water soluble nutrients; however, conservation tillage reduced total nutrient loss by controlling erosion and sediment bound nutrients.

Barisas et al. (1978) studied the effects of residue cover on nutrient losses. They grew corn under six tillage regimes on three soils, Ida silt loam, Tama silty clay loam, and Kenyon sandy loam in Iowa. Runoff samples from one simulated rainstorm 16 to 35 days after fertilization were analyzed. Their results are reported in Table 7. The six tillage practices (and average percent cover) were conventional (5%), till-plant (15%), chisel plow (19%), disk (30%), ridge plant (35%), and fluted coulter (55%). (Though there seem to be some correlations between percent cover and nutrient losses, a study of this design does <u>not</u> demonstrate a causal relationship.)

Table 7
Dissolved Nutrients in Runoff from a Simulated Storm as a Function of Residue Cover on Corn Plots on Three Soils
Barisas et al., 1978

Percent Cover	Nitrogen Losses mg/ha dissolved[1]	particulate[2]	Phosphorus Losses mg/ha dissolved[3]	particulate[4]
5	1	72	1	38
15	3	48	3	63
19	3	42	4	50
30	3	27	5	32
35	4	20	7	25
55	4	10	7	12

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[1] NO_3-N, NH_4-N
[2] Total Particulate Nitrogen (Organic N and Fixed Inorganic NH_4-N)
[3] PO_4-P
[4] Total Available Particulate Phosphorus

In Johnson et al. (1979) as in the Barisas study, greater total phosphorus losses were observed from minimum till than from conventional till. No-till produced less P loss than the other two treatments. Their results are not reported here because 1) runoff sampling was conducted only during hydrograph recession,

2) sedimentation occurred in their weirs, reducing the sediment bound nutrients by some unknown amount, and 3) they only sampled during the growing season. For these reasons their annual load estimates are in question.

Baker and Laflen, 1982 reported that increasing residue amounts hardly affected the concentration of dissolved nutrients (NO_3-N, PO_4-P, and NH_4-N) in runoff (Table 8). In their rainfall simulator study they applied 0, 375, 750 and 1500 kg/ha corn residue to disked plots. Using equation 1, this corresponds to 0, 3, 7, and 13% cover. No crop was planted. The plots were disked, and fertilizer and residue were surface applied 1 day before the test. The residue was collected and stacked uncovered 11 months before the test. The results of the study should not be generalized to apply to higher cover levels.

Table 8
Concentrations of Dissolved Nutrients with Increasing Residue
Baker and Laflen, 1982

Cover,%	Runoff,mm	NH_4-N,mg/l	NO_3-N,mg/l	PO_4-P,mg/l
0	65	0.3	2.4	0.18
3	56	0.3	3.6	0.18
7	50	0.3	2.3	0.26
13	20	0.3	4.2	0.26
LSD(0.05)		0.1	1.8	0.08

Timmons et al. (1970) leached samples of alfalfa, bluegrass, barley straw and oat straw in a laboratory. The leaching of soluble nutrients was greatly enhanced by freezing or drying, especially for alfalfa and bluegrass. Table 9 lists the maximum amount of nutrients leached from the plant samples over three water extractions. The authors suggest this is the kind of nutrient release that could be expected in the first snowmelt runoff leaving fields and grassed areas.

Table 9
Maximum Amount of Soluble Nutrients Leached From Plant Residues
Timmons et al., 1970

Residue type	Inorganic P µg/g dry wt.	Organic P µg/g	Total P µg/g	Total N µg/g
Alfalfa	2100	305	2405	6541
Bluegrass	1246	673	1919	3456
Barley straw	9	33	41	388
Oat straw	10	35	44	237

Schreiber and McDowell (1985) found relationships between (1) the intensity of simulated rainfall and nutrient concentrations in runoff from winter wheat straw, and (2) the concentration of nutrients and the amount of time passed since the onset of runoff. For 25 mm of simulated rainfall at 7, 12, 25, 53 and 105 mm/hr they reported concentrations ranging from 0.37 to 0.51 mg/l nitrate-N, 0.11 to 0.15 mg/l ammonium-N, and 0.56 to 1.04 mg/l phosphate-P. The lower concentrations corresponded to higher intensities. The nutrient concentrations coming from fresh straw would not be the same as those leaving decomposing straw over several months.

Thompson et al. (1983) placed clover and wheat straw residue in nylon mesh bags beneath sorghum plants and monitored the total Kjeldahl nitrogen (TKN) loss over 22 weeks. About half the TKN from the clover sample was lost in that time, but the wheat straw showed no significant decrease in TKN content after the 22 weeks.

Holt (1979) offers the following estimates of the N in corn and soybean residue (without reference to its environmental destination): corn, from 62 to 71 kg (presumably total) N per ha; and soybeans, from 68 to 73 kg N per ha.

Sharpley and Smith (1989) studied the mineralization and leaching of phosphorus from soil incubated with surface-applied and incorporated crop residue. In their laboratory tests finely ground samples of alfalfa, peanut, soybean and wheat residues were placed on and in samples of eight different soils and incubated for 84 days at near optimal moisture and temperature conditions for minearilization. Mineralizaition and movement of inorganic P through the soil was consistently greater for surface-applied residues than for incorporated residues. Averaged for all soils and residue types 2.4 kg P ha^{-1} was leached from incorporated residue samples while 3.1 kg P ha^{-1} was leached from surface applied samples. These results do not quantitatively reflect field conditions.

B. SURFACE ROUGHNESS AS A FUNCTION OF TILLAGE

Two types of roughness are produced by tillage implements. Oriented roughness is composed of regular repeating features, such as the ridges and furrows occurring between the rows of lister and ridge planting or the undulations of plow furrow slices and cultivator furrows. Random roughness refers to the peaks and depressions unrelated to the direction in which the tillage operation was performed.

Ridges, surface roughness, and contouring can affect runoff quality. The differences in the particulate N and P concentrations in runoff between conservation tillage treatments are primarily due to selective soil erosion in which small colloidal clay particles are removed preferentially to the larger silt or sand size particles (Römkens et al., 1973). Ridges across slope and/or corn residue on the soil surface act as sediment traps and barriers to runoff. Tillage practices that induce these surface conditions, such as chisel and coulter, yield sediments enriched in the clay fraction relative to practices producing less surface roughness.

Table 10 gives ridge heights formed by three tillage implements.

Table 10
Average Ridge Heights for Three Conservation Tillages
Meyer et al., 1970

chisel	11.3 cm
ridge till	8.5 cm
coulter	5 cm

Cogo et al. (1984) measured random roughness before and after 2 hours of simulated rainfall at 6.4 cm/hr in the absence of surface residue (Table 11). Roughness index reduction by rainfall was the smallest for no-till, intermediate for sweep tillage only, and highest for systems involving secondary tillage (disk). The test soils were a Russell silt loam and a Martinsville loam (both Typic Hapludalfs, fine-loamy, mixed, mesic).

By regression analysis, they found that residue cover was more influential than roughness in predicting soil loss. Runoff and soil loss decreased significantly with an increase in either residue cover or surface roughness, but the effects of the two were not independent. Also the roughness effect diminished as antecedent moisture increased.

Table 11
Roughness Index Before and After a Simulated Rain Event
Cogo et al., 1984

Tillage type	Roughness index, cm		Percent
	before rain	after rain	Decrease
No-till	0.54	0.43	20
Fall moldboard plow + spring disk	1.50	0.60	60
Fall chisel + spring disk	1.70	0.71	60
Spring-sweep tillage only	2.66	1.46	45

Siemens and Oschwald (1978) measured surface roughness and other soil characteristics for several conservation tillage treatments for double cropped corn-soybeans at a time after the corn harvest but before spring tillage (Table 12).

Table 12
Soil Conditions Following Corn and Before Spring Tillage
Siemens and Oschwald, 1978

Treatment	Surface Roughness	Surface Cover[1] %	Surface Residue t/ha	Cone Index[2] k Pa
Conventional	1.05	3.5	0.0	386
Disk-chisel	1.14	49.9	5.8	627
Coulter-chisel	1.20	54.2	5.4	621
Chisel	1.10	53.8	3.4	579
Disk	1.18	85.2	6.3	676

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[1] the ratio of soil area exposed to the corresponding area of a horizontal plane
[2] standard penetrometer readings of 0-15 cm layer

Lindstrom and Onstad (1984) measured surface and subsurface properties affected by tillage, highlighting the difference between spring and fall primary tillage. Table 13 shows the random roughness data for the third year of continuous corn under three tillages: (1) fall moldboard plow, spring disk; (2) fall chisel plow, spring disk; and (3) ridge-till on ridges established by cultivation after harvest of the previous crop.

In general roughness values were low on all plots. This is common in planted fields. After fall primary tillage, surface elevations may be quite variable, giving high random roughness with the capacity for considerable surface storage of water, but after a winter of weathering, secondary tillage, and planting, random roughness declines.

Table 13
The Decline of Random Roughness Coefficients During
Seedbed Crop Stage After Simulated Rainfall
Lindstrom and Onstad, 1984

	Conventional	Chisel	No-till
	Random Roughness, cm		
Initial moisture conditions	1.02 ± 0.14	0.93 ± 0.13	0.87 ± 0.30
After 5.6 cm simulated rain[1]	0.91 ± 0.21	0.86 ± 0.11	0.79 ± 0.19
After additional 4.2 cm rain[2]	0.80 ± 0.16	0.77 ± 0.11	0.77 ± 0.25
LSD (0.05)	0.18	0.13	0.26

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[1] one hour at 5.6 cm/hr
[2] applied 24 hrs after first simulated rainfall

As described previously, their experiment involved two artificial storms immediately after planting, separated by a 24 hour rest period. Measurements were taken three times in two days:

(1) Before the dry run. (This measurement gave soil properties for the antecedent moisture condition.)
(2) After the dry run. (The dry run consisted of applying 5.6 cm of simulated rainfall to the plots in 1 hr.) and
(3) After the wet run. (The wet run took place 24 hr after the first simulated rainfall. It consisted of a 4.2 cm application in 45 minutes).

Differences among tillage treatments and especially between residue variables were small, so roughness coefficients from residue removed plots (70% removed) were averaged with residue left plots for each tillage.

The authors state that the decline in random roughness in the 2-day period implies surface breakdown and soil particle re-orientation. The conventional and reduced tillage treatments showed reduced random roughness from the initial conditions to the final condition ($p = 0.05$). The reduction with no-till was not significant, indicating a more stable surface condition.

A less statistical, but more to the point analysis is that after a winter of weathering, roughness differences are slight between tillages, as well as before and after rainfall.

Burwell et al. (1966) reported roughness index and total pore space of the top 15 cm of soil for a variety of tillage practices (Table 14) on a Barnes loam alfalfa-brome sod, before and after a 13 cm/hr rain.

Table 14
Pore Space and Random Roughness of a Barnes Loam
as Affected by Preplant Tillage and Rainfall
Burwell et al., 1966

Tillage Treatment	Before Tillage	After Tillage	At First Runoff	After 5 cm Runoff
		total pore space, cm		
Untilled	8.1		8.1	8.1
Plow	7.6	13.7	11.7	11.2
Plow-disk-harrow	8.1	12.4	11.4	11.2
Cultivated	8.1	9.6	9.4	9.6
		random roughness, cm		
Untilled	0.8		0.8	0.6
Plow	0.8	5.0	3.3	2.7
Plow-disk-harrow	0.7	2.5	1.8	1.4
Cultivated	0.8	2.9	2.2	1.7

Table 15
Random Roughness Changes Over Two Years as Affected by Tillage
Allmaras et al., 1966

Random Roughness Index, cm[1]

1961

Preplanting	Postplanting	Before Tillage (May 29)	After Preplanting Tillage (June 8)	After 1st Cultivation (June 22)	After 2nd Cultivation (July 5)	After 3rd Cultivation (July 17)
Plow-disk-harrow	Cultivated	0.86	1.12a	1.37a	2.24b	1.98b
	Noncultivated	0.86	1.12a	0.84a	0.89a	0.81a
Plow	Cultivated	0.86	2.18b	2.82c	2.92c	2.57c
	Noncultivated	0.86	2.18b	2.03b	2.21b	1.83b
Untilled	Cultivated	0.86	0.86a	1.98b	2.74c	2.90c
	Noncultivated	0.86	0.86a	0.86a	0.86a	0.86a

1962

Preplanting	Postplanting	Before Tillage (May 13)	After Preplanting Tillage (June 13)	After 1st Cultivation (July 10)	After 2nd Cultivation (July 25)	End of Season (September 25)
Plow-disk-harrow	Cultivated	0.64	1.02b	2.13b	2.34b	1.07b
	Noncultivated	0.64	1.02b	0.58a	0.51	0.58a
Plow	Cultivated	0.64	1.91c	2.41bc	2.34b	0.86a
	Noncultivated	0.64	1.91c	0.89a	0.76a	0.74a
Untilled	Cultivated	0.64	0.66a	2.54c	2.41b	1.30b
	Noncultivated	0.64	0.66a	0.66a	0.58a	0.66a

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[1] Within a column, values not followed by the same letter are significantly different at $p = 0.05$.

Allmaras et al. (1966) traced the changes in inter-row random roughness through two crop seasons on a Barnes Loam (Table 15). This study is notable, not only for establishing the relationship between field cultivation activities and random roughness, but for demonstrating the variability inherent in these measurements.

C. MANNING'S N

Donigian and Davis (1978) give the following guidance for assigning n values to agricultural land surfaces:

Table 16
Manning's n Values for Agricultural Surfaces
Donigian and Davis, 1978

	Manning's n
Conventional practices	0.15 - 0.25
Smooth fallow	0.15 - 0.20
Rough fallow, cultivated	0.20 - 0.30
Crop residues, light turf	0.25 - 0.35
Meadow, heavy turf	0.30 - 0.40

Engman (1986) suggest the following values for Manning's n, based on runoff plot data.

Table 17
Manning's n Values for Agricultural Surfaces
Engman, 1986

	Residue(t/ac)	Manning's n	Recommended Value
Fallow, no residue		0.006- 0.16	0.05
Chisel plow	< 0.25	0.006- 0.17	0.07
Chisel plow	0.25 - 1	0.07 - 0.34	0.18
Chisel plow	1 - 3	0.19 - 0.47	0.30
Chisel plow	> 3	0.34 - 0.46	0.40
Disk/harrow	< 0.25	0.008- 0.41	0.08
Disk/harrow	0.25 - 1	0.10 - 0.25	0.16
Disk/harrow	1 - 3	0.14 - 0.53	0.25
Disk/harrow	> 3	-	0.30
No-till	< 0.25	0.03 - 0.07	0.04
No-till	0.25 - 1	0.01 - 0.13	0.07
No-till	1 - 3	0.16 - 0.47	0.30
Moldboard plow/fall		0.02 - 0.10	0.06
Coulter		0.05 - 0.13	0.10
Grass (bluegrass sod)		0.39 - 0.63	0.45
Mixed grasses and alfalfa		0.17 - 0.30	0.24
Concrete or asphalt		0.01 - 0.013	0.011

D. SURFACE DEPRESSION STORAGE

Infiltration and surface runoff are affected by micro-relief surface storage. Agricultural soils have varying degrees of micro-relief surface storage. This storage is dependent on the recent history of the soil surface, being modified by the action of rain, wind, tillage and cultivation practices.

Moore and Larson (1979) examined the relationship between random roughness[3] and depression storage on plowed and unplowed plots on Barnes loam soil in Minnesota. The site was gently sloping. They measured the surface roughness before and after the application of 0.5 cm at 0.9 cm/hr artificial rain.

Random roughness on planted fields in the corn belt averages about 1 cm. In this experiment, random roughness was temporarily increased by plowing from an average of 0.84 cm to 2.22 cm.

Both random roughness and maximum storage were decreased significantly by the action of rainfall. On the average, the maximum storage was increased 3.5 times by plowing. Typically, rainfall decreased storage on plowed surfaces by 45%, resulting in storages roughly twice the volume of the original (before rain) unplowed plots. The unplowed plots also lost a substantial part (40%) of their surface storage during rainfall.

The authors proposed the following relationship between the amount of storage before (S(before), mm) and after (S(after), mm) rainfall when S(before) exceeds 2.0 mm ($R^2 = 0.77$, $p \leq 0.01$, standard error of estimate = 1.9 mm):

$$S(\text{after}) = -0.523 + 0.627 \cdot S(\text{before}) \qquad [3]$$

The inclusion of rainfall volume and duration, random roughness and total porosity of the plow layer as independent variables did not significantly improve equation 3.

The results of their field measurements of random roughness and estimates of associated maximum storage, averaged over 16 plots, are found in Table 18.

If it does not bother you to write a linear regression of maximum depression storage on roughness (both in mm) based on the 4 data points in Table 17, then that equation (R^2=0.92 and standard error of Y estimate = 1.4 mm) would be:

$$\text{Depression Storage} = -1.9 + 0.54 * (\text{Roughness Index}) \qquad [4]$$

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[3] defined by Allmaras et al. (1966) as the standard error of point elevations or heights within a plot.

Table 18
Changes with Rainfall in Mean Values of Random Roughness and Storage Volume for Plowed and Unplowed Fields
Moore and Larson, 1979

	Random Roughness mm	Maximum Storage Volume mm
Plowed Plots		
before rain	22.2	11.2
after rain	17.9	6.2
Unplowed Plots		
before rain	8.4	3.2
after rain	7.1	1.9

IV. SUBSURFACE SOIL PROPERTIES AFFECTED BY TILLAGE

A. PHYSICAL PROPERTIES

1. SOIL TEMPERATURE

Griffith et al. (1977) compared the average soil temperature for the first 8 weeks after planting corn on several Indiana soils (Table 19). The first two soils are in northern Indiana and the corn height at 8 weeks was roughly 1 meter. The last two soils were from eastern and southern Indiana, respectively, and their average crop height after 8 weeks was 2 meters. Soil temperatures generally are cooler early in the growing season with no-tillage. Differences of 2 to 3 °C at a 10 cm depth are common during May and June.

Table 19
Soil Temperature and Tillage at Eight Weeks after Planting Corn
Griffith et al., 1977

Tillage System	Tracy sandy loam °C	Runnymede loam °C	Blount silt loam °C	Bedford silt loam °C
Conventional spring plow	22.4	21.7	24.3	26.1
Conventional fall plow	22.6	22.0	24.7	--
Chisel	20.1	19.6	22.4	24.2
Ridge till	21.1	20.8	23.4	25.1
Coulter (no-tillage)	18.8	18.2	22.1	23.4

Tollner et al. (1984) compared soil temperature, bulk density, water retention, and soil texture for conventional till and no-till soybeans (six years of double cropped soybeans after small grain) grown on an erosive Piedmont soil in Georgia. They reported that near surface temperatures (in the top 5 cm of soil) in the no-till plots were consistently 5 to 7°C (9 to 13°F) lower than in the conventional tillage plots. Near surface temperatures in the conventional plots frequently exceeded 40°C (104°F).

Table 20 presents temperature traces (at a 5 cm depth) for two situations (1) after two weeks of dry weather and (2) after a saturating rain. Notice that the temperature of bare ground exceeds ambient temperature in the afternoon for the dry weather situation but is cooler than air temperatures during the night.

Table 20
Temperature of 5 cm Deep Soil as a Function of Time and Cover
Tollner et al., 1984

Time	No-till	Conventional	Ambient	Bare Ground
A. Temperature[1] (°C) of dry Soil, August 17-18, 1981 after 2 weeks without rain				
10:00	18.2	16.0	20.4	21.0
12:00 noon	21.2	21.3	24.9	25.3
14:00	22.6	22.5	26.8	32.0
16:00	23.3	23.6	27.6	34.1
18:00	23.2	23.4	27.8	32.1
20:00	22.1	22.6	24.9	27.9
22:00	21.2	20.9	20.5	24.2
24:00 midnight	20.7	19.9	19.3	23.0
2:00	20.1	19.2	18.2	23.0
4:00	19.8	19.1	18.0	20.8
6:00	19.8	19.1	17.2	20.8
8:00	19.3	18.5	17.0	19.7
10:00	19.1	18.8	22.5	23.7
12:00 noon	20.5	20.8	25.8	27.7
14:00	21.9	22.3	28.0	-
16:00	23.1	24.2	29.2	-
B. Temperature of wet Soil, August 24-25, 1981 after a saturating rain				
10:00	21.2	20.5	21.5	23.6
12:00 noon	21.9	22.0	24.2	26.9
14:00	23.1	23.3	25.7	29.9
16:00	22.6	22.8	25.0	28.7
18:00	22.4	21.9	23.9	26.4
20:00	22.0	21.7	22.9	24.7
22:00	21.3	20.9	22.5	24.2
24:00 midnight	21.2	20.8	22.0	23.6
2:00	21.4	20.9	21.2	23.0
4:00	21.0	20.3	20.8	22.1

---

[1] read from a plot

The conventionally tilled plots were often 5-6°C warmer <u>early in the season</u>, however, towards the end of the season, little difference was detected. Daily minimum soil temperatures were not substantially different at any time throughout the study. The authors suggest that the bare ground temperatures be considered similar to those of conventional till plots between rows before canopy closure (Tollner and Hargrove, 1982).

At a depth of 50 cm, the time temperature relationship (over 5 days) was practically flat and within 2°C (3.6°F) of the temperatures at 100 cm.

Using 54 days of data beginning 38 days after planting, the authors found that maximum and minimum soil temperatures for either tillage could be predicted by linear regression on ambient maximum and minimum temperatures with a significance level of 0.01. Other variables that did not improve the fit of the regression were daily evaporation, daily solar radiation, maximum and minimum standard soil temperature measured at a Class A weather station near the experimental site.

Daily rainfall was also a good predictor of minimum soil temperature for both tillages. Daily rainfall was also strongly related to maximum near surface soil temperature for no-till (significance of 0.01). Rainfall predicted minimum soil temperature for no-till with significance of 0.05.

The authors warn that spurious correlations existed in the soil moisture and rainfall data "because of poor rainfall distribution" (whatever that means).

Canopy development could predict maximum temperature for no-till and conventional till at $p = 0.01$ but the same relationship was not significant for minimum temperatures.

2. SOIL BULK DENSITY

Tollner et al. (1984) examined mechanical impedance (penetration resistance) and bulk density of soils after six years of continuous double cropped soybeans after small grain. Penetration resistance was greater under no-till (1000 kiloPascals, or 145 psi) than under conventional tillage (550 kiloPascals, or 80 psi).

In the no-till plots, the maximum force was required in the top 15 to 25 cm, while in the conventional tillage plots the maximum occurred between 30 and 40 cm. This pattern with depth was reflected in bulk density data in Table 21. These data are averages of 6 years of measurements at unspecified times. The authors attributed the elevated mechanical impedance levels at the 30-40 cm level under conventional tillage to a "traffic pan" caused by twice annual plowing.

Table 21
Dry Bulk Density Differences with Depth and Tillage
On Continuous Soybean Plots in the Georgia Piedmont
Tollner et al., 1984

Tillage	Depth 0 - 15 cm	 30 - 35 cm
	bulk density, g/cm^3	
No-till	1.52	1.43
Conventional	1.38	1.53

The greater sand content of the surface soil in the no-till plots was given as an important factor in the greater bulk density of this stratum. Studies on other soils do, however, show a similar tillage effect (see below).

Lindstrom and Onstad (1984) studied soil properties immediately after planting for continuous corn (third year) on a 2% slope Barnes loam (fine-loamy, mixed, Udic Haploboroll).

The authors compared three tillage systems: (1) conventional: fall moldboard plow, spring disk, (2) fall chisel plow, spring disk, and (3) ridge-till, consisting of growing season cultivation of the previous crop to establish ridges for planting. Tillage was not done after harvest.

Seventy percent of the corn stover was harvested from one half of each plot area for a test of the effect of residue harvesting on soil properties during the brief period 6 to 17 days after the planting of the next corn crop.

Bulk density of soil and saturated hydraulic conductivity observations are presented in Table 22. These data indicate that the surface conditions developed from the tillage treatments immediately after planting were substantially different.

Residue harvest type for the two years prior to the study did not affect the soil parameters.

The bulk density of the Ap horizon for the no-till system was statistically higher than that for the tilled treatments. However, bulk densities at the lower horizons were similar for all treatments. If a traffic pan did exist below 30 cm in this soil, as was the case in the previous study, reporting the average bulk density over the 30-75 cm depth would have obscured this.

Table 22
Bulk Density and Saturated Hydraulic Conductivity of Soil Horizons As Affected by Tillage During Seedbed Crop Stage
Lindstrom and Onstad, 1984

Treatment	Bulk Density g/cm^3	Saturated Hydraulic Conductivity cm/hr
Ap Horizon, 0 - 15 cm		
Conventional	1.04 ± 0.09	18.0 ± 9.7
Chisel till	1.17 ± 0.08	9.9 ± 9.6
Ridge till	1.34 ± 0.03	0.3 ± 0.3
LSD (0.05)	0.07	6.9
B2 Horizon, 15 - 30 cm		
Conventional	1.16 ± 0.12	6.1 ± 8.4
Chisel till	1.28 ± 0.13	2.4 ± 1.9
Ridge till	1.25 ± 0.08	2.4 ± 1.7
LSD (0.05)	not significant	not significant
C1ca Horizon, 30 - 75 cm		
Conventional	1.29 ± 0.07	4.5 ± 3.3
Chisel till	1.32 ± 0.13	3.3 ± 1.2
Ridge till	1.31 ± 0.10	4.0 ± 2.7
LSD (0.05)	not significant	not significant

Griffith et al. (1977) compared the bulk density of the first 20 cm of five soils 2 to 3 weeks after planting corn (Table 23). The bulk density of the plow layer changes over the year, reflecting the action of tillage and weathering. The greatest difference between tillages is evident immediately after planting.

Blevins et al. (1977 and 1983a,b) reported no difference due to tillage in bulk density measurements made in April (prior to tillage) of a Maury silt loam (Typic Paleudalfs) after 10 years of continuous corn (Table 24).

Also the bulk density of the top 7.5 cm of soil from bluegrass plots had comparable bulk density (1.25 g/cm^3) to soils from corn plots, implying the conventional plot had recovered from the associated soil disturbances. The difference in hydraulic conductivity with tillage was not statistically significant.

They also reported bulk densities in the 0 to 7.5 cm layer of a somewhat poorly drained Johnsburg silt loam (Typic Fragiaqualfs) under chisel, conventional, and ridge till soybeans as 1.30, 1.37 and 1.37.

Table 23
Effect of Tillage on Bulk Density of the Top 20 cm of Five Soils Two to Three Weeks After Planting Corn
Griffith et al., 1977

	Bulk Density, g/cm^3				
Tillage System Pweano	Tracy sandy loam	Runnymede loam	Bedford silt loam	silt loam	Blount silty clay loam
Conventional, spring plow	1.20	1.13	1.13	1.21	1.06
Chisel	1.25	1.11	1.06	1.13	1.01
Till-plant	1.35	1.33	1.16	1.28	1.15
No-tillage	1.56	1.43	1.33	1.39	1.17

Table 24
Bulk Density of Maury Silt Loam Before Planting
Blevins et al., 1983b

Tillage	Soil Depth cm	Bulk Density g/cm^3	Saturated Hydraulic Conductivity cm/hr
No-till	0.0 - 7.5	1.25	1.9
	7.5 - 15	1.28	
Conventional	0.0 - 7.5	1.29	1.5
	7.5 - 15	1.31	
Bluegrass sod	0.0 - 7.5	1.27	
	7.5 - 15	1.29	

Tollner et al. (1984) cited Blevins et al. (1977) as an example of a study where there was no difference in bulk density with tillage. Tollner attributes the lack of difference in Blevins' results to the fine texture and good structure of the Maury silt loam soil, rather than to an annual recovery phenomenon after a winter of weathering.

Edwards (1982) measured the bulk density of a Rayne silt loam soil (Typic Hapludult) under no-till and conventional till corn production. At the onset of the study both plots were conventionally tilled, and the bulk density of the soil was less than 1.4 g/cm^3 but after two years, the bulk density of the no-till plots had increased to 1.6 g/cm^3. Table 25 shows monthly bulk density determinations. Each monthly entry represents 40 samples collected at 5 to 15 cm depth.

Table 25
Monthly Changes in Bulk Density in the Plow Layer with Tillage
Edwards, 1982

Date	No-till wheel-row	No-till open row	Conventional wheel-row	Conventional open row
	Bulk Density[1], g/cm^3			
OCT 78	1.59	1.58	1.31	1.31
MAR 79	1.56	1.55	1.31	1.30
APR 79			(plowing done)	
JUN 79	1.61	1.59	1.36	1.15
SEP 79	1.59	1.61	1.36	1.20
DEC 79	1.59	1.58	1.36	1.37

---

[1] read from a graph

3. POROSITY

Edwards (1982) observed that with continuous no-tillage management, plow layer porosity decreased from 50 to 40%, but the preserved macropores in the soil and crop residue on the surface reduced runoff to 1/20 of that from a nearby conventionally tilled watershed. Tillage increased the immediately available pore space near the surface, but decreased the amount of infiltrating water moving deeper into the profile.

Allmaras et al. (1966) studied the effect of tillage on soil porosity. They estimated the initial total porosity of the top 15.24 cm (6 inches) of a Barnes loam soil in Minnesota as a function of bulk density (equation 5). Subsequent estimates of porosity (Table 26) were simply the initial estimate modified by change in average height of the plot surface. Though non-standard, these results are internally consistent and present an interesting time trace.

$$\text{(Initial Porosity, in.)} = (6\text{in.})(2.65 - \text{Bulk Density, g/cc})/2.65 \qquad [5]$$

Note the between-year variation. Other factors influencing porosity include soil type, soil management history, and moisture content at the time of tillage. Disking and harrowing decreased the porosity when performed on friable soils, but increased porosity on wetter soils.

Table 26
Porosity of the Top 15 mm of Soil as Affected by Tillage
Allmaras et al., 1966

1961

Porosity, cm pore/cm soil[1]

Preplanting	Postplanting	Before Tillage (May 29)	After Preplanting Tillage (June 8)	After 1st Cultivation (June 22)	After 2nd Cultivation (July 5)	After 3rd Cultivation (July 17)
Plow-disk-harrow	Cultivated	0.53	0.73a	0.73a	0.60ab	0.61ab
	Noncultivated	0.53	0.73a	0.71a	0.66bc	0.67b
Plow	Cultivated	0.53	0.84b	0.85b	0.76cd	0.71b
	Noncultivated	0.53	0.84b	0.84b	0.83d	0.84c
Untilled	Cultivated	0.53	0.53c	0.61c	0.60ab	0.54a
	Noncultivated	0.53	0.53c	0.54c	0.54a	0.55a

1962

Preplanting	Postplanting	Before Tillage (May 13)	After Preplanting Tillage (June 13)	After 1st Cultivation (July 10)	After 2nd Cultivation (July 25)	End of Season (September 25)
Plow-disk-harrow	Cultivated	0.54	0.74b	0.68b	0.67bc	1.07b
	Noncultivated	0.54	0.74b	0.73b	0.71bc	0.58a
Plow	Cultivated	0.54	0.81b	0.77b	0.73c	0.86a
	Noncultivated	0.54	0.81b	0.75b	0.73d	0.74a
Untilled	Cultivated	0.54	0.53a	0.56ab	0.63b	1.30b
	Noncultivated	0.54	0.53a	0.52a	0.53a	0.66a

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[1] Within a column, values not followed by the same letter are significantly different at $p = 0.05$.

Lindstrom et al., (1984) studied the effects of porosity on infiltration rate after corn planting. See Tables 27 and 28. Like others they found higher infiltration rates for conventionally plowed fields relative to minimum till fields immediately after planting activities.

They attributed the higher infiltration rate to a large volume of macropores in the conventionally tilled plot, rather than to the effects of inversion plowing on surface storage. Their findings are not really in conflict with other authors who claim that plowing obliterates macropores (see section IV.B.1.b.), because Lindstrom et al. were defining macropore differently. The largest diameter pore measured in the Lindstrom study was 0.06 mm, where macropores are roughly 2 to 11 mm in diameter.

Lindstrom et al. measured soil properties of Barnes loam plots in the third year of continuous corn (Table 27). Simulated rainfall was applied at 5.6 cm/hr <u>one to two weeks after planting.</u>

The three tillage treatments contrasted were (1) fall plow, spring disk, (2) fall chisel, spring disk, and (3) fall cultivation to make ridges for spring planting.

The water tensions in the Ap horizon at the start of the dry run were near 1 bar for all tillage treatments. The volumetric water contents for the no-till, conventional till, and reduced till systems were 28.8%, 18.8% and 22.0%, respectively. At the start of the wet run, water tension was near 0.1 bar. Volumetric water content was about 40% for all treatments.

Simulated rainfall was applied at a rate of 5.6 cm/hr with 55% the energy of natural rainfall.

Residue remaining on the three treatments were trace, 65%, and 100% for conventional, reduced, and no-till, respectively.

This paper is occasionally cited as an example of how no-till can diminish infiltration. This is justified, with the qualification that infiltration is reduced <u>one or two weeks after planting under moderate intensity rainfall</u>.

The total porosity and pore size distribution for the soil horizons are given in Table 28 (Lindstrom and Onstad, 1984). In the Ap horizon, total porosities differed with tillage. Conven-tionally tilled and chisel tillage treatments had significantly larger volumes of <60 μm diameter pores (at 1 to 2 weeks after planting) than did the ridge till system.

Volumes of the smaller pores in the Ap horizon were not significantly different. No differences in pore sizes were observed in the lower horizons.

Table 27
Physical Properties in the Ap Horizon (0-15 cm of Barnes Loam) Measured After Planting in the Third Year of Continuous Corn and Infiltration Rates at Two Antecedent Moisture Conditions
Lindstrom et al., 1984

Tillage	Physical Properties Bulk Density g/cm^3	Penetrometer Resistance kg/cm^2	Saturated Hydraulic Conductivity cm/hr	Macropore Volume %
Conventional	1.04	0.49	18.0	18.3
Chisel	1.17	0.35	9.9	12.9
Ridge till	1.34	16.04	0.3	7.6
LSD(0.05)	0.07	2.22	6.9	3.6

Tillage	Infiltration Rate[2] cm/hr Residue Not Harvested[3] Dry Run	 Wet Run	 Residue Harvested Dry Run	 Wet Run[1]
Conventional	5.4	2.2	5.4	2.3
Chisel	5.4	2.2	5.3	1.9
Ridge till	3.6	2.0	3.3	1.3
LSD(0.05)	0.5	0.3	0.5	0.5

Table 28
Total Porosity and Pore Size Distribution of Soil Horizons As Affected by Tillage During Seedbed Crop Stage
Lindstrom and Onstad, 1984

Treatment	Total Porosity	Pore Size Diameter[1] > 60 μm	 10 - 60 μm	 < 10 μm
		percent of total volume		
Ap Horizon, 0 - 15 cm				
Conventional	60.8 ± 2.7	18.3 ± 2.8	13.5 ± 2.2	29.6 ± 3.4
Chisel	54.7 ± 3.1	12.9 ± 1.8	18.7 ± 6.4	23.2 ± 5.0
Ridge till	49.4 ± 5.3	7.6 ± 2.1	13.3 ± 3.2	30.6 ± 7.5
LSD (0.05)	6.4	3.6	not significant	
B2 Horizon, 15 - 30 cm				
Conventional	58.1 ± 2.3	17.5 ± 4.5	15.6 ± 4.8	24.7 ± 4.4
Chisel	51.3 ± 6.3	10.3 ± 5.4	15.0 ± 5.9	26.6 ± 4.6
Ridge till	52.4 ± 5.7	9.7 ± 3.7	14.8 ± 3.9	28.2 ± 6.8
LSD (0.05)		not significant		
C1ca Horizon, 30 -75 cm				
Conventional	51.7 ± 2.3	11.9 ± 0.3	13.2 ± 1.6	28.3 ± 1.9
Chisel	48.7 ± 8.4	11.4 ± 1.5	10.5 ± 0.4	27.3 ± 6.4
Ridge till	48.3 ± 0.5	10.4 ± 1.7	9.4 ± 1.6	22.5 ± 7.2
LSD (0.05)		not significant		

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[1] 60 μm is the diameter of pores that will drain at 50 cm of water tension. 10 μm is the diameter of pores that will drain at 300 cm.

B. HYDROLOGIC PROPERTIES

1. INFILTRATION

The infiltration rate is the rate at which water is transmitted into the ground through the surface layer of soil, as distinct from percolation rate, which deals with water movement through deeper strata. Infiltration rates are influenced by many factors, such as vegetal cover, surface crusting, temperature, rainfall intensity, soil properties, and water quality.

At low rainfall intensity, the two main factors affecting the infiltration rate are antecedent soil moisture and soil bulk density. Soil bulk density is changed by various tillage practices through cultivation or artificial compaction. At higher rainfall intensities, the role of macropores dominates.

During most of the year, infiltration is greater under no-tillage, but during seedbed, the disrupted plow layer of conventional tillage may admit more water than does no-till.

a. ANTECEDENT MOISTURE

Shinde et al. (1982) studied infiltration rates in a moderately drained clay loam soil in India (33% clay, 27% silt, and 40% sand; containing 0.48% organic carbon and 9.06% calcium carbonate). They compared infiltration rates for seedbed conditions for three tillage practices under four initial soil moisture contents. The three tillage practices were

(1) zero-tillage, consisting of weed removal and bed preparation; bulk density of the first 20 cm of soil was 1.35 g/cm^3;

(2) conventional tillage, moldboard plowing to a depth of 20 cm followed by two harrowings; bulk density 1.24 g/cm^3; and

(3) conventional tillage with light compaction, compaction by wooden harrow beam twice after conventional tillage; bulk density 1.29 g/cm^3.

The antecedent moisture conditions in the top 15 cm were:

Condition	% Moisture	cm^3 water/cm^3 soil	Description
1	33	0.40	1/3 bar tension
2	20	0.25	5 bar tension
3	17	0.20	15 bar tension
4	10	0.12	air dry soil

Eight centimeters of water were applied (intensity not specified). The measured infiltration rates are given in Table 29. The infiltration rate drops quickly during the first 20 minutes after the onset of irrigation, but reaches a steady state, represented by the 180-minute rate. Graphs of the infiltration rates over time show very little change from the 30 minute measurement through the 180 minute measurement.

Equations predicting the cumulative infiltration (I) in cm to time elapsed (t) in minutes under initial moisture conditions of 0.12 cm^3/cm^3 were fitted:

Zero Tillage $I = 0.22 + 0.417\ t**0.62$ [6]
Conventional Tillage $I = 0.30 + 0.513\ t**0.71$ [7]
Light Compaction $I = 0.22 + 0.468\ t**0.67$ [8]

Table 29
Change in Infiltration Rates with Duration of Irrigation
Under Different Antecedent Moisture Conditions, Seedbed Stage
Shinde et al., 1983

Initial Moisture Condition, cm^3/cm^3	0.12	0.20	0.25	0.40
	Infiltration Rate, cm/hr, after 5 minutes			
Conventional Tillage,	22.0	17.4	13.5	9.8
Light Compaction	18.8	15.5	13.0	7.5
Zero Tillage	15.8	13.8	10.5	6.0
	Infiltration Rate, cm/hr, after 180 minutes			
Conventional Tillage	5.2	4.1	4.0	2.4
Light Compaction	3.7	3.0	2.7	2.0
Zero Tillage	2.8	2.7	2.3	1.4

b. MACROPORES

Water sometimes moves through undisturbed soil profiles much faster than can be explained by capillary water flow theory (Edwards et al., 1979). These anomalies of water and chemical movement have been attributed to the presence of large, continuous, surface-connected pores in the soil. The large pore flow occurs only during or soon after heavy rainfall, and it seems to bypass water held in the small pore soil matrix.

A system of large, surface-connected pores can result from the activity of soil animals such as worms or ants, by the rooting of plants, and by processes such as shrinkage, and freeze-thaw. These channels are obliterated by plowing, but redevelop as the crop grows.

Ehlers (1975) studied the difference in worm channels under no-till and conventional till winter wheat on a loess-derived, gray-brown podzolic soil (Hapludic Eutroboralf). After 4 years of no-tillage, the number and volume percentage of earthworm channels nearly doubled in the Ap horizon (0-25 cm). Altogether, the volume percentage occupied by channels was less than 1 percent, even in the B_t horizon (45-110 cm), which had the highest density of channels in both tilled and untilled plots.

Small, blocked, or horizontal pores (typical of a tilled Ap horizon) do little to conduct water downward; large diameter (8-11 mm), vertical macropores connected to the surface, responsible for much of the downward movement of water, are more typical of no-till plots. Unlike earthworm channels in the Ap horizon of the tilled plot, almost all of the channels reaching the untilled soil surface transmitted water deeply into the profile.

Ehlers estimated the maximum infiltration rate via earthworm channels in no-tilled podzolic soil to be 1.2 mm/min (7.2 cm/hr), compared to 0.6 mm/min (3.6 cm/hr) for tilled soil (infiltration in excess of that controlled by small scale porosity).

c. TRAFFIC PANS

Another reason for slow water intake of some soils can be the formation of a traffic pan at the 20 to 25 cm depth. This may occur on up to 25% of the typical cultivated field. Traffic compressed soils have low porosity.

Ehlers (1975) reported that the saturated hydraulic conductivity of the traffic pan in a silt loam to be 8×10^{-4} cm/sec, compared to the uncompressed cultivated soil above it at 5×10^{-3} cm/sec.

Lindstrom et al. (1984) measured runoff, infiltration, and soil properties in wheel-compressed and uncompressed areas of each of the three tillage systems (Table 30).

The 10-year study on continuous corn fields took place on four soils in Minnesota and South Dakota. Infiltration was measured on Nicollet clay loam, Forman clay loam, and Sinai silty clay loam in the first year and tenth year of the study. Rainfall was applied at 12.7 cm/hr immediately after planting.

Differences among infiltration rates for the three tillage treatments were generally not significant within compression variables, but tractor traffic reduced infiltration by a factor of 2 to 2.5 for clay loams and by 4 to 7 for the silty clay loam.

Table 30
Infiltration Rate (30 Minutes After the Initiation of Runoff)
After Corn Planting Operations on Three Soils
as Affected by Tillage
Lindstrom et al., 1984

	Infiltration Rate, cm/hr			
	First Crop Year		Tenth Crop Year	
	uncompressed	compressed	uncompressed	compressed
Nicollet clay loam				
Conventional	4.25	1.75	7.20	4.15
Chisel	5.00	2.00	7.33	4.15
Ridge till	2.88	1.38	4.36	4.41
LSD(0.05)	2.3		3.8	
Forman clay loam				
Conventional	6.00	3.38		
Chisel	5.75	2.75		
Ridge till	5.00	1.50		
LSD(0.05)	2.6			
Sinai silty clay loam				
Conventional	3.50	0.50		
Chisel	4.50	1.13		
Ridge till	4.25	1.88		
LSD(0.05)	2.4			

d. SURFACE CRUSTS

Many soils will form a dense surface crust upon exposure to relatively low rain intensities. Such soils often have high silt and low organic matter and calcium carbonate. During rain, silt and clay segregate, forming a surface seal. McIntyre (1958) determined saturated hydraulic conductivity of a crust formed by an Australian fine sandy loam as 5×10^{-7} cm/sec, where the conductivity of the soil below the crust was 10^{-3}cm/sec.

Ehlers (1975) observed that surface crusts did not form on no-till plots on podzolic soils (Typudalf) derived from loess, but were thick on tilled plots.

The increase of runoff from surface crusting has been observed with row crops under no-tillage on a well drained soil, though to a much lesser extent than on conventional till plots (Edwards, 1982). On soils with poor internal drainage the effects of crusting are overwhelmed by restrictions to infiltration from the subsoil (Edwards and Amerman, 1984).

Tillage related differences in cumulative infiltration may be short lived. Meyer and Mannering (1961) showed that surface crusts developed rapidly on conventionally tilled and minimum tilled soil surfaces on a Russell silt loam and significantly reduced infiltration amounts (Table 31).

Table 31
Changes in Infiltration Rate Over a Crop Year
For Conventional and Minimum Till Corn
Meyer and Mannering, 1961

Treatment	After Planting	After Cultivation	After Harvest
	Infiltration rate[1], cm/hr		
Dry Run			
Conventional tillage, cultivated	2.8	3.4	4.3
Minimum tillage, cultivated		4.1	4.7
Minimum tillage, uncultivated	4.1	2.5	3.1
Wet Run			
Conventional tillage, cultivated	2.4	2.2	3.4
Minimum tillage, cultivated		3.1	4.1
Minimum tillage, uncultivated	3.4	2.3	3.1
Very Wet Run			
Conventional tillage, cultivated	2.2	1.9	2.9
Minimum tillage, cultivated		2.6	3.7
Minimum tillage, uncultivated	3.0	2.0	3.0

---

[1] average rate during last 5 minutes of test

They compared infiltration rates during simulated rainfall under conventional tillage and five reduced tillages:

(1) Conventional: (plow, two diskings with trailing harrow, plant); two cultivations
(2) Plow-plant without smoothing; no cultivation
(3) Plow-plant with smoothing; no cultivation
(4) Plow-plant with smoothing; two cultivations
(5) Plow without smoothing, wheel-track planting; no cultivation
(6) Plow without smoothing, wheel-track planting; two cultivations.

Rainfall was applied first two to three weeks after planting (when corn plants were less than 1 foot tall) at an intensity of

6.6 cm/hr for 1 hour (dry run) followed the next day by 30 minutes more of the same intensity rainfall (wet run), a 15 minute rest, and another 30 minute application (very wet run).

The second test occurred after the first cultivation (same rainfall application). The corn was approximately 2 feet high. This operation eliminated surface crusts on the cultivated field so the effect of crusting on infiltration rate can be appreciated.

The last test was performed after harvest (same rainfall application.

e. PLANT AVAILABLE WATER

Tollner et al. (1984) compared water retention and soil texture for conventional till and no-till soybeans (six years of double cropped soybeans after small grain) grown on Cecil clay loam, an erosive Piedmont soil in Georgia.

Six years of data on the continuous soybean plots were used to determine soil properties. Plant available water (Table 32) was estimated by subtracting the residual moisture content from the water content at 300 cm suction. (The residual moisture content is the value at which moisture content ceases to change as suction increases).

Table 32
Soil Water and Textural Analysis by Tillage and Depth
Tollner et al., 1984

	surface (0 - 5 cm)	subsoil (30 cm)	surface (0 - 5cm)		subsoil (30 cm)	
	Plant available water, cm^3/cm^3		%sand	%clay	%sand	%clay
No-till	0.10	0.18	54	17	41	29
Conventional	0.17	0.23	44	29	44	29

Total Water (cm) in Top 60 cm of Soil

	21 May	23 June	8 July	29 July	12 August
No-till	14.3	16.4	22.5	12.5	16.8
Conventional	12.2	14.2	15.2	11.7	16.8

Soil in the no-till plots had significantly less plant available water near the surface than did the soil in the conventional tillage plots, though total water in no-till profiles exceeded that in conventional profiles.

It is not clear that Tollner's operational definition of plant available water is correct, in view of the observations that (1) that conventional tillage is used to warm and dry out wet soils for early planting, and (2) that no-till crops are more drought resistant than conventional till crops.

2. PERCOLATION

After water crosses the surface interface, its rate of downward movement will be controlled by the transmission characteristics of the underlying soil profile, including the volume of storage available below ground. Percolation rate is soil specific and varies with depth, time of year, as well as tillage.

Though the movement of water below the root zone is determined by a number of factors in addition to tillage, several authors have reported net percolation as a function of tillage. Net percolation refers to the volume of water leaving the root zone, not to the rate at which it leaves, which is determined by soil properties.

Alberts and Spomer (1985) demonstrated increased subsurface water loss with conservation tillage. They proposed several theories to account for this, though none of them were directly tested by their work. The two main suggested causes of the reduced surface runoff were enhanced infiltration and residue effects (including reduction of surface seal formation, runoff velocity, and evaporation).

They remark that the alteration of the water balance by conservation tillage can increase net percolation and the possibility of NO_3-N leaching below the root zone.

The test sites were in the deep loess hills of western Iowa. Three watersheds were instrumented:

(1) contour conventionally tilled corn,
(2) contour till-plant corn, and
(3) till-plant corn on terraces with underground pipe drainage.

Subsurface flows for these systems are reported in Table 33.

On their test plots for till-plant and conventional till corn, annual surface runoff from till-planted watershed generally was lower than that from the conventional tilled watersheds. Conservation tillage increased percolation below the root zone. The annual (10-year average) subsurface flow from the till-plant watershed/pasture was 18.6 cm compared to 13.6 cm for conventional tilled corn.

This subject will be revisited in the sections on nutrient fluxes and ground water contamination.

Table 33
Annual Subsurface Flows from Conventional Till and Till Plant Corn Watersheds
Alberts and Spomer, 1985

Year	Contour Conventional	Contour Till-Plant	Terraced Till-Plant
		cm[1]	
1974	22.0	21.1	19.2
1975	20.0	16.8	17.3
1976	12.3	12.8	10.7
1977	8.5	14.7	12.1
1978	13.3	21.3	16.0
1979	12.7	19.6	14.5
1980	9.5	18.7	15.1
1981	5.9	10.7	8.1
1982	10.8	22.1	16.0
1983	21.7	30.7	24.9

---

[1] Data read from a histogram. Reading error was approximately ± 0.5 cm (i.e., 0.5 cm is the amount of subsurface flow corresponding to the width of the smallest division on the ruler used to read the histogram.)

Lindstrom and Onstad (1984) reported saturated hydraulic conductivity with depth and tillage immediately after planting for continuous corn (third year) on a 2% slope Barnes loam (fine-loamy, mixed, Udic Haploboroll) (Table 34).

The authors compared three tillage systems:

(1) conventional: fall moldboard plow, spring disk,

(2) fall chisel plow, spring disk, and

(3) ridge till, consisting of growing season cultivation of the previous crop to establish ridges for planting.

Tillage was not done after harvest.

Their experiment involved two artificial storms immediately after planting, separated by a 24 hour rest period. Measurements were taken three times in two days:

(1) before the dry run (rainfall applied to dry ground)

(2) after the dry run, after 5.6 cm of simulated rainfall was applied in 1 hr, and

(3) after the wet run (24 hr after the first simulated rainfall a second rainfall of 4.2 cm in 45 minutes was applied to the wet soil).

Table 34
Saturated Hydraulic Conductivity of Soil Horizons
As Affected by Tillage During Seedbed Crop Stage
Lindstrom and Onstad, 1984

Treatment	Saturated Hydraulic Conductivity cm/hr
	Ap Horizon, 0 - 15 cm
Conventional	18.0 ± 9.7
Chisel till	9.9 ± 9.6
Ridge till	0.3 ± 0.3
LSD (0.05)	6.9
	B2 Horizon, 15 - 30 cm
Conventional	6.1 ± 8.4
Chisel till	2.4 ± 1.9
Ridge till	2.4 ± 1.7
LSD (0.05)	not significant
	C1ca Horizon, 30 - 75 cm
Conventional	4.5 ± 3.3
Chisel till	3.3 ± 1.2
Ridge till	4.0 ± 2.7
LSD (0.05)	not significant

Hydraulic conductivity varied greatly, particularly in the Ap horizon. The authors felt that an insufficient number of samples were collected to characterize the hydraulic conductivity of the Ap horizon. In general conductivity was high for the tilled systems and low for the no-till system, with no tillage effect detectable at depth.

V. EFFECTS OF TILLAGE ON MASS FLUXES

A. WATER BUDGET

1. RUNOFF

Occasionally in the literature a vacuous debate over the effects of conservation tillage on runoff quantity is encountered. Some authors are cited as finding decreased runoff with conservation tillage and others an increase (for an example of such a debate see Mueller et al., 1984a) implying more controversy than actually exists. Examination of the papers cited show their results are reported for <u>specific crop stages</u> or soil types and are therefore not commensurate.

For instance, no-tillage can result in increased runoff during the seedbed stage, relative to conventional tillage (Lindstrom and Onstad, 1984), though at other times runoff is reduced by no-tillage (McGregor and Greer, 1982). The amount of runoff reduction is not always dramatic or even evident, and is dependent on a number of factors in addition to tillage.

Another problem resulting in disinformation is insufficiently detailed presentation of conclusions. For example, in the abstract of Lindstrom and Onstad (1984) it is stated that

> "the no-tillage system produced the greatest runoff regardless of residue harvesting... The soil physical parameters measured indicated that the no-till system forms an undesirable surface condition characterized by high bulk density, high penetrometer resistance, low saturated hydraulic conductivity, and low volume of macropores, all of which can promote rapid water runoff under normal rainfall conditions."

Besides the problem that what those authors call no-till is actually ridge till, these conclusions are presented without the <u>essential</u> qualification that all measurements were taken during a period 6 to 17 days after planting, and are not representative of the rest of the year.

Mueller et al. (1984a) studied the effect of manure application on soil and water losses from corn fields under conventional, chisel, and no-till systems. Simulated rainfall was applied several times during the growing season.

Tillage and date interactions for runoff volume were reported (Table 38). Runoff was significantly greater for no-till plots relative to conventional and chisel sites immediately after planting, though the differences between no-till and conventional till were obscured in the other crop stages.

Consistently lower runoff volumes came from chisel sites during crop stage 3, but for seedbed and crop stage 1 the runoff volumes from the chisel sites resembled those from the conventional sites.

The importance of soil drainage characteristics in determining tillage related hydrologic response was documented by Edwards and Amerman (1984).

They contrasted a well drained soil (Rayne silt loam Typic Hapludult, parent material: medium grained resistant sandstone) with a soil having limiting internal drainage (Keene silt loam Typic Aquic Hapludalf, parent material: silty shale)

No-till and conventional till corn were grown for four years on the Ohio test site. During the summer months, no-tillage on either soil increased cumulative infiltration relative to conventional tillage.

Surface protection from residue was sufficient to reduce runoff during the winter dormant season on the well-drained soil, but on the poorly drained field the soil became saturated above the subsurface flow-restricting horizons, regardless of the surface cover conditions, producing increased runoff.

McGregor and Greer (1982) measured runoff and sediment from no-till, reduced till, and conventional till corn plots grown on highly erodible loess derived Providence (Typic Fagiudalfs) and Lexington (Typic Paleudalfs) silt loams in Mississippi. Reduced tillage consisted of no-till planting with two cultivations replacing herbicide application for weed control. Rows were up and down the 7 to 8% slopes. Monthly runoff and sediment loss are reported in Table 35.

Runoff was significantly reduced by conservation tillage, though strict no-till was not nearly as effective as no-till plus cultivations. The 3 year average runoff from no-till corn harvested for grain was 12% less than that from conventional till plots and the runoff from reduced till plots was 45% less (probably indicating surface seal development).

The expected reduction of runoff with residue cover was observed on the conventional plots from October through December, but did not occur at all on the no-till plots.

Soil loss from plots in conventional till corn harvested for grain was about 22 times greater than that for no-till harvested for grain plots, and 17 times greater than that from the reduced till plots.

2. EVAPOTRANSPIRATION

Crop residue at the soil surface shades the soil and serves as a vapor barrier for water losses from the soil. Blevins et al. (1983a) report estimated transpiration and evaporation values for conventional and no-till corn grown for four years on Maury silt loam, a deep, well-drained upland Kentucky soil (Typic Paleudalfs). They do not describe their experimental protocol. Monthly average values for the growing season are found in Table 36.

Table 35
Runoff and Erosion From Conventionally Tilled and No-till Corn Plots
McGregor and Greer, 1982

Date	Rainfall	Runoff Conventional[1]		Runoff No-Till		Runoff Reduced	Soil Loss Conventional		Soil Loss No-Till		Soil Loss Reduced
		silage	grain	silage	grain	grain	silage	grain	silage	grain	grain
	mm	mm	mm	mm	mm	mm	t/ha	t/ha	t/ha	t/ha	t/ha
October											
1975	67	24	7	4	13	4	0.24	0.02	-	0.01	0
1976	61	4	0	0	0	0	0.01	0	0	0	0
1977	135	41	19	-	14	11	1.37	0.08	-	0.01	0.02
Average	88	23	9	2	9	5	0.54	0.03	0	0.01	0.01
November											
1975	107	37	21	21	25	21	0.75	0.14	0.04	0.03	0.08
1976	172	51	44	18	45	21	0.13	0.13	0.01	0.02	0.04
1977	57	11	9	-	9	5	0.08	0.03	-	0.01	0.01
Average	112	33	25	20	26	16	0.32	0.10	0.02	0.02	0.04
December											
1975	139	69	37	25	40	27	0.43	0.13	0.04	0.04	0.06
1976	68	8	9	5	14	4	0.02	0.03	0.01	0.01	0.01
1977	46	0	0	-	0	0	0	0	-	0	0
Average	84	26	15	15	18	10	0.15	0.05	0.02	0.02	0.02
January											
1975	132	60	57	45	55	39	1.14	0.58	0.12	0.07	0.16
1976	97	22	24	12	27	10	0.07	0.11	0.02	0.03	0.03
1977	80	13	28	-	30	16	0.20	0.20	-	0.03	0.05
Average	103	32	36	28	37	22	0.47	0.30	0.07	0.04	0.08
February											
1975	167	77	67	52	74	50	1.43	0.92	0.13	0.11	0.47
1976	119	44	46	33	48	28	0.84	0.65	0.13	0.10	0.24
1977	48	4	1	-	3	2	0.43	0.04	-	0	0.02
Average	111	42	38	42	42	27	0.90	0.54	0.13	0.07	0.24
March											
1975	233	127	129	112	140	101	1.74	0.86	0.16	0.13	0.24
1976	208	88	78	52	76	51	1.09	0.58	0.18	0.13	0.34
1977	148	74	67	-	84	56	6.07	0.85	-	0.34	0.47
Average	196	96	91	82	100	69	2.97	0.76	0.17	0.20	0.35
April											
1975	86	15	10	8	20	5	0.87	0.46	0.07	0.02	0.02
1976	100	41	42	23	35	28	6.59	6.74	0.18	0.18	0.27
1977	104	31	29	-	32	15	2.36	0.45	-	0.09	0.12
Average	97	29	27	16	29	16	3.27	2.55	0.12	0.10	0.14

(Table 35 continued)

May											
1975	217	92	89	45	81	36	19.18	14.78	0.21	0.12	0.61
1976	132	52	58	25	42	28	3.76	4.53	0.02	0.07	0.06
1977	35	4	2	-	1	2	0.09	0.04	-	0	0
Average	128	49	50	35	41	22	7.68	6.45	0.12	0.06	0.22
June											
1975	42	0	0	0	0	0	0	0	0	0	0
1976	146	33	33	18	22	17	2.93	2.78	0.05	0.03	0.41
1977	99	7	8	-	7	3	0.64	0.59	-	0.05	0.07
Average	96	13	14	9	10	7	1.19	1.12	0.02	0.03	0.16
July											
1975	44	0	2	0	0	0	0	0.12	0	0	0
1976	83	13	16	4	6	2	0.65	0.98	0.01	0.01	0.02
1977	73	17	18	-	2	1	0.90	0.88	-	0.01	0.01
Average	67	10	12	2	3	1	0.52	0.66	0	0.01	0.01
August											
1975	124	18	27	5	9	3	0.95	1.47	0.02	0.01	0.05
1976	33	12	10	2	1	1	1.79	1.72	0.01	0	0.03
1977	59	4	4	-	0	1	0.32	0.24	-	0	0
Average	72	11	14	4	3	2	1.02	1.14	0.02	0	0.03
September											
1975	66	15	14	3	3	1	0.88	0.43	0.01	0.01	0.01
1976	80	32	30	5	4	7	4.63	3.94	0.02	0.01	0.13
1977	110	84	83	-	22	22	3.74	2.28	-	0.06	0.14
Average	85	44	42	4	10	10	3.08	2.22	0.02	0.03	0.09
Tankline Cleanout[2]											
1975							2.28	1.70	0.05	0.26	0.02
1976							2.25	1.62	0	0.23	0.04
1977							1.34	1.52	-	0.12	0.05
Average							1.96	1.61	0.02	0.20	0.04
Annual Totals											
1975	1424	534	460	320	460	287	29.89	21.61	0.85	0.81	1.72
1976	1299	400	390	197	320	197	24.76	23.81	0.64	0.82	1.62
1977	994	290	268	-	204	134	17.54	7.20	-	0.72	0.96
Average	1239	408	373	259	328	207	24.07	17.53	0.73	0.79	1.43

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[1] "Silage" and "Grain" refer to how the crop was harvested. Harvesting for silage means a portion of the stalks were taken. Harvesting for grain means stalks were left on the field. On the average 2190 and 4530 kg/ha residue were left on the conventional till corn plots for silage and grain, respectively, and 5045 and 9060 kg/ha on the no-till corn plots for silage and grain.

[2] Soil deposits in pipes leading to sample storage tanks were flushed out at least once each year.

Table 36
Soil Water Evaporation and Transpiration by Corn
Blevins et al., 1983a

Month	No-till		Conventional		
	Transpiration cm	Evaporation cm	Transpiration cm	Evaporation cm	Rain cm
May	0.0	2.1	0.0	6.3	17.9
June	7.6	1.0	6.4	6.8	9.7
July	12.4	0.3	9.5	2.1	10.1
August	9.2	0.2	7.2	1.4	4.1
September	1.5	0.5	1.1	2.5	9.1
Total	30.7	4.1	24.2	19.1	50.9

B. SOIL AND SEDIMENT FLUXES

Tillage systems utilizing the chisel plow or disk substantially reduce soil losses compared to conventional tillage (Siemens and Oschwald, 1976; Johnson and Modenhauer, 1979). Several studies have reported soil loss reductions for these systems were comparable to those from no-till (Römkens et al., 1973; Griffith et al., 1977; Lindstrom and Onstad, 1984). In many of these studies, in spite of higher sediment concentrations for chisel or disk systems relative to the no-till system, runoff was lower from chisel and disk, and as a result, soil losses were similar to those from no-till.

In contrast, Laflen et al. (1978) found chisel and disk systems less effective than no-till in reducing soil loss. Little difference in runoff was reported among tillage systems, but no-till plots produced lower sediment concentrations in runoff.

Römkens et al., 1973 contrasted the effects of five tillage practices on nitrogen and phosphorus in runoff from 2 simulated rainstorms occurring in the second to fourth week after planting of corn (when plant heights were 2 to 20 cm). The tillage systems were (1) conventional: plow, disk twice, plant, (2) ridge-till plant, (3) chisel till plant, (4) double disk: disk twice, coulter plant, and (5) coulter plant.

The test site soil was a Bedford silt loam (18.3% clay, 74.4% silt, 4.8% sand, and 2.5% organic matter in the plow layer). Slopes ranged from 8.2 to 12.4%, and tillage was across slope.

They concluded that the contour coulter and contour chisel systems controlled soil loss during the establishment phase of corn growth, but runoff water contained high levels of soluble N and P from surface applied fertilizer. The contour disk and contour ridge till systems were less effective in controlling soil erosion, but produced runoff with lower concentrations of soluble N and P. Contour conventional tillage, in which fertilizer was plowed under, had the highest losses of soil and water but the smallest losses of soluble N and P in runoff. All treatments lost large amounts of sediment bound nutrients.

The type of harvest can affect soil loss. From Table 34 (McGregor and Greer, 1982), we see an increase in soil loss from conventional plots when residue was removed, though the effect on no-till soil loss was questionable. This contrasts to the results of Lindstrom and Onstad (1984, Table 6, Section III.A.2.) who reported significantly higher soil losses from no-till compared to conventional or chisel systems during the seedbed crop stage when residue was removed, though no differences otherwise.

When corn is grown for cash grain, crop residue is usually left and animal wastes are seldom applied, but many dairy and livestock farmers remove a portion of crop residue for feed or bedding purposes. This practice may leave marginal residue coverage for erosion control.

Mueller et al., 1984a evaluated the effect of manure application and tillage on soil and water loss from corn fields. Their field plots were on Dresden silt loam (Mollic Hapludolls), a well to moderately well drained soil consisting of 61 to 76 cm of loess over calcareous gravel. The slope was 4 to 6%.

Runoff from simulated rainfall was collected during seedbed (May 1978 and June 1979), late crop stage 1 (July 1978), and crop stage 3 (September 1978 and August 1979).

In the first year some residue was removed, and in the second year no residue was removed. For the unmanured plots the residue breakdown cover was as follows:

	1978	1979
No-till	35%	45%
Chisel	12%	26%
Conventional	1%	3%

Manure added approximately 13% cover to no-till plots and 4% to chisel plots.

The conventional plots were spring moldboard plowed (mid-May) and disk harrowed once immediately prior to planting. Primary tillage was done to a depth of 20-25 cm and secondary tillage to a depth of 7 cm. All tillage was performed across slope. Chisel plowing and planting occurred in spring (mid-May). No-till plots were not tilled.

The results of this factorial experiment are summarized in Table 37.

Table 37
Significance of F Ratios in ANOVA on Runoff, Sediment Concentration, and Soil Loss
Mueller et al., 1984a

Effect	1978 Runoff	1978 Sed. Conc.	1978 Soil Loss	1979 Runoff	1979 Sed. Conc.	1979 Soil Loss
Tillage (T)	0.01	0.01	0.10	0.01	0.01	0.01
Manure (M)	-	0.05	-	0.05	0.01	0.01
Date (D)	0.01	-	-	0.01	-	-
T X M	-	0.10	0.05	-	-	-
T X D	0.05	-	-	0.01	-	-
M X D	0.10	-	-	-	-	-
T X M X D	-	-	-	-	0.10	-

Of interest are the Tillage-Date (T X D) interaction for runoff (Table 38) and the Tillage-Manure (T X M) interaction for soil loss (Table 39). In almost all cases, sediment concentration and soil losses (Table 39) were lower for manured contour chisel and contour no-till sites relative to unmanured contour chisel and contour no-till sites, though no such manure effect was observed for contour conventional sites. The effect of partial removal of residue in 1978 was evidenced by the inability to distinguish the unmanured tillage treatments by their soil loss. In 1979, when all residues were left, soil loss was less with contour chisel and contour no-till than with contour conventional, as was expected.

The use of manure on contour no-till corn did not enhance soil retention when the stalks were left on the field, apparently because the field had sufficient residue cover. However, manure application did reduce soil loss from contour chisel plots. Manure application reduces soil loss on no-till corn when the stalks are harvested for silage.

Table 38
Runoff: Tillage and Date Interaction
Mueller et al., 1984a

Treatment	1978 May	1978 July	1978 September	1979 June	1979 August
	liters of runoff				
Conventional	51d[1]	126bc	185a	47c	118ab
Chisel	50d	106c	148b	39c	20c
No-till	134b	146b	160a	112b	146a

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[1] Within each year, values followed by the same small letter are not significantly different at p = 0.10, as tested by the least significant difference test.

Table 39
Soil Loss: Tillage and Manure Interaction
Mueller et al., 1984a

Treatment	1978	1979
	kg/ha sediment	
Conventional, without manure	3040ab[1]	4490a
Conventional, with manure	4360a	2460ab
Chisel, without manure	2590ab	1380b
Chisel, with manure	1290c	50c
No-till, without manure	2480ab	750b
No-till, with manure	1080c	420b

---

[1] Within each year, values followed by the same small letter are not significantly different at p = 0.10, as tested by the least significant difference test.

Angle et al. (1983 and 1984) reported that, during 1980, the suspended sediment loss from their experimental conventional till corn watershed was more than 11 times greater than the loss from their no-till watershed. The soil was a Manor loam (coarse loamy, micaceous, mesic Typic Dystrochrept) in the middle Patuxent River drainage in Howard County, Maryland.

This result is consistent with results reported by McGregor and Greer (1982), who observed a 22-fold difference in sediment losses when comparing no-till and conventional till corn. The higher sediment losses of the McGregor and Greer study was attributed (by Angle et al.) to the steeper slopes of McGregor and Greer's test plots (6 to 8 %), the highly erodible, thin loess soils of their plots, and to the high rates of infiltration of the Manor loam soil under both tillages.

In Angle et al.'s study (see Table 44 section V.C.2.b.ii.) one or two rainfall events each year contributed the bulk of the yearly total sediment loss. For example, the runoff events of 13 June and 17 June 1982 were responsible for 78% of all sediment loss from the conventional till watershed for the entire year. During the same year, 99% of all sediment loss from the no-till watershed occurred on 17 June.

The work of Grumbs and Lindsay (1982) is sometimes misleadingly cited as a counter example of no-till's ability to reduce soil loss (e.g. Angle et al., 1983). Grumbs and Linsay's results are highly case specific. They found no significant differences in sediment loss between tillage systems in corn on steep (11-52% slope) mountain soils in Trinidad when the yearly rainfall total was low.

C. NUTRIENT FLUXES

1. NITROGEN CYCLE

In a rare field study of nutrient flux as affected by tillage, House et al. (1983) contrasted nitrogen cycling for conventional and no-till soybeans with rye winter cover and for sorghum/rye.

The study illustrates the influence of tillage on the mechanisms that transform nitrogen in agriculture. Ammonia volatilization loss estimates were not included in their nitrogen budgets.

No pesticides were applied. The soil type was Hiwassee loam (Typic Rodudults) on plots with relatively flat slopes (<3%).

Their results are presented schematically in Figure 1 and in tabular form in Table 40.

Soil tillage practices proved to be important regulators of N cycling processes. The seasonal dynamics of N differed between many conventional and no-tillage agroecosystem components.

Plowing increased the total annual amount and rate of N uptake by crop plants, litter decomposition and possibly crop consumptio n by insects. Each of these intercompartmental N transfers (kg/ha·yr) were greater in conventional tillage (plowed) than in no-tillage systems.

Standing stocks (kg/ha) of surface crop and weed residue where consistently higher in no-tillage systems than in conventional systems.

No-tillage litter was composed primarily of residue from the previous crop, while much of the conventional tillage litter, especially late in the season, was leaf-fall from the current crop.

At the termination of each cropping season, a large pulse of crop residue was added to the decomposition subsystem. The N in this organic matter was substantial (up to 100 kg/ha) and contributed significantly to the overall N economy of no-tillage agroecosystems.

Maximum rates of surface litter decomposition occurred in the summer, coinciding with peak populations of several taxa of soil biota important in decomposition.

Soil N under no-tillage often exceeded that in plowed soil. The upper soil layer (0-10 cm) under no-tillage contained as much as 20% more total N than the same soil layer in plowed soils.

Figure 1
Annual Nitrogen Budget Model
House et al., 1984

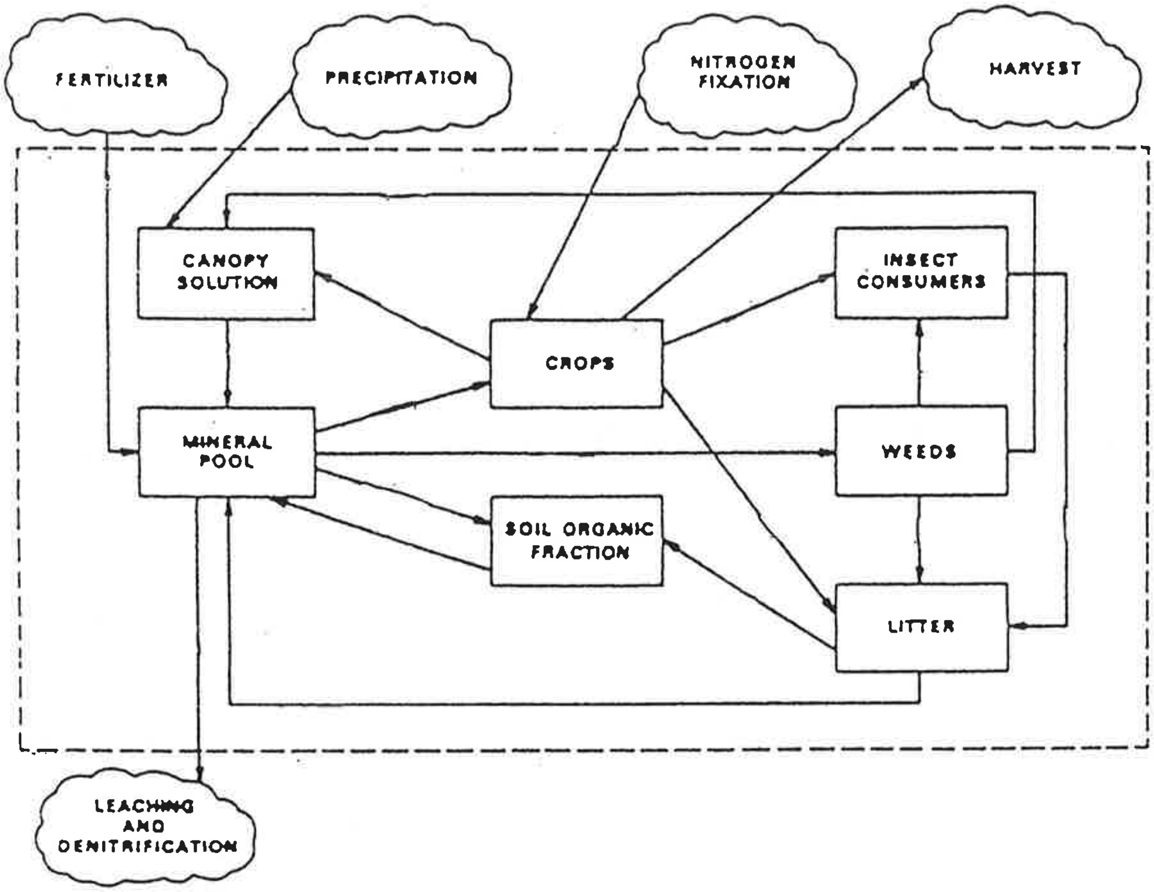

Table 40
Annual Nitrogen Fluxes, No-till versus Conventional Till Crops
House et al., 1984

Flux	Soybeans/Rye No-till	Soybeans/Rye Conventional	Sorghum/Rye No-till	Sorghum/Rye Conventional
	kg/ha yr			
Fertilizer to Mineral Pool	70	70	148	148
Mineral Pool to Crops	164.7	194.3	320	362.8
Mineral Pool to Weeds	41.2	28.5	29.2	23.1
Crops to Canopy Solution	5	5	0.5	1.0
Crops to Harvest	119.5	134.5	172.6	178
Crops to Litter	81.8	86	149.9	179.8
Crops to Insect Consumers	3.4	3.8	2	4
Litter to Mineral Pool		included with next entry		
Litter to Soil Organic Fraction	125.4	117.3	175.6	205.9
Soil Organic Fraction to Mineral Pool	42.5	73.5	85	94.4
Mineral Pool to Leaching and Denitrification	26	32	20	24
Mineral Pool to Soil Organic Fraction	21	21	44.4	44.4
Precipitation to Canopy Solution	9	9	4	4
Canopy Solution to Mineral Pool	15	15	5	6
Insect consumers to Litter	4.9	4.3	3	4.7
Weeds to Litter	38.7	27	27.7	21.4
Weeds to Canopy Solution	1	1	0.5	1.0
Weeds to Insect Consumers	1.5	0.5	1.0	0.7
Nitrogen Fixation to Crops	45	35	0	0
Net System Loss	21.5	52.5	40.6	50

The differences dropped sharply with increasing soil depth although at the lower soil depth (40-50 cm) nitrate-N concentrations were higher in conventionally tilled soils than in no-tillage soils.

The higher concentration of organic matter in the upper soil under no-tillage plays a major role in the immobilization of N, especially surface applied fertilizer. Rather than being a nitrogen loss, this large pool of immobilized N in no-tillage soils represents an enhancement of storage capacity and promotes long-term N retention within untilled agroecosystems.

In Maryland the typical amount of N fertilizer applied to soybeans is between 17 and 22 kg/ha (not 70 kg/ha). The winter cover crop, rye, would not be fertilized.

This study does not directly give rates or parameters for HSPF but it documents differences between no-till and conventional tillage nitrogen budgets that could be manifested in HSPF applications attempting to portray these practices. This paper also provides conceptual framework, similar to HSPF's, against which to interpret the more fragmentary studies.

2. NUTRIENT MOVEMENTS

a. NUTRIENT UPTAKE BY PLANTS

The availability and hence utilization of N by crops is affected by tillage practice (Kitur et al., 1984, Section VI.A.1.a.) especially at low N fertilization rates. Soil organic, mineralizable, and microbial biomass N are concentrated in surface few centimeters of soil under no-tillage, and not dispersed throughout the plow layer, as with conventional tillage (Doran, 1980, Section VI.C.).

Power et al. (1986) conducted an isotopic tracer study of nitrogen uptake by no-till soybeans and corn. This study quantified the different sources of nitrogen in the harvested crops, distinguishing residual nitrogen from first year fertilizer application, nitrogen from second ("current") year fertilizer application, nitrogen from indigenous soil nitrogen plus N fixed by soybeans, and nitrogen from crop residue.

Four levels of residue application were contrasted: 0, 0.5, 1.0 and 1.5 times the quantity of crop residue produced the previous year.

The sources nitrogen in the harvested plants as affected by residue treatment are given in Table 41. The experimental plots were on a Crete-Butler silty clay loam in Nebraska.

Table 41
Sources of N Taken up by No-till Soybeans and Corn
Power et al., 1986

Residue Rate	Crop Residues	Residual Fertilizer	Current Fertilizer	Native Soil N	Total
		Soybean, kg N ha^{-1}(percent)			
0.0	0.0 (0)	1.8 (2)	13.8 (14)	84.2 (84)[1]	99.8
0.5	1.0 (1)	1.6 (1)	20.6 (14)	124.0 (84)[1]	147.2
1.0	38.2 (22)	6.7 (4)	16.1 (9)	115.9 (66)[1]	176.9
1.5	63.2 (32)	5.7 (3)	20.0 (10)	105.5 (54)[1]	194.4
		Corn, kg N ha^{-1}(percent)			
0.0	0.0 (0)	4.8 (6)	3.7 (5)	72.8 (90)	81.3
0.5	0.0 (0)	5.8 (5)	7.3 (7)	96.7 (88)	109.8
1.0	1.8 (1)	5.8 (4)	7.1 (5)	114.4 (89)	129.1
1.5	0.0 (0)	5.7 (4)	10.6 (8)	124.5 (88)	140.8

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[1] plus biologically fixed N

Total and soil N uptake by soybean and corn tended to increase as residue rate increased. Almost all the N in soybean residues was mineralized and taken up by the second crop, with most of the uptake occurring after mid-July. Uptake by soybean of labeled N immobilized in soil organic matter the previous year increased from about 2 to 6 kg N ha^{-1} as residue rate increased. The authors speculated that the source of this N was N temporarily immobilized in microbial biomass.

Uptake by soybean of current-year fertilizer N also increased slightly with residue rate. The fraction of total N in soybean derived from indigenous soil N (plus fixed N) decreased with increased residue rate from about 85% for no residue to 55% for the 1.5 residue rate.

For corn, essentially none of the N from corn residue was recovered by the next crop at any residue rate. Uptake of labeled N immobilized in soil organic matter was not affected by residue rate, but uptake from second year fertilizer treatment increased about 6 kg/ha with increased residue rate. Over 80% of the total N uptake by corn in the second year came from indigenous soil N, regardless of residue rate.

Several short term studies of no-tillage versus conventional tillage have suggested that N availability is lower in no-tillage.

Reduced N uptake by crops under the first years of no-till systems usually is most evident at lower rates of N fertilizer. In the first year of their study Stinner et al. (1983) found reduced N uptake in no-till and conventional tilled sorghum, winter wheat, and weeds (293 versus 350 kg/ha biomass), but no difference was detected in the second year (221 versus 218 kg/ha). The soil of this Georgia piedmont study was a well drained sandy clay loam of the Hiwassee series (Typic Rhodudult).

Rice et al. (1986) proposed an explanation for the transient nature of the increased N uptake in the first years of conventional tillage relative to no-tillage. When the established sod was broken at the initiation of continuous corn cropping there was a rapid decline in soil onganic N. The loss of N from the surface 5 cm was almost threefold greater in the conventional tilled plot than in the no-till plot. The higher mineralization rate associated with tillage is offset by the reduced organic N reservoir under conventional tillage, resulting in roughly equal N availabilty in either tillage after 5 to 10 years.

The soil was a Maury silt loam (fine silty, mixed, mesic Typic Paleudalfs), a deep, well drained soil formed in residuum of phosphatic limestone (1 to 3% slope) in Kentucky. Corn was grown with a winter cover crop of rye at different fertilizer application rates. Total soil nitrogen over the 16 year experiment is reported in Table 42.

Table 42
Total Soil Nitrogen Changes Over Time and Tillage
For Corn with Winter Rye Cover
Rice et al., 1986

Year	1970		1974		1983	
Depth, cm	0-5	5-15	0-5	5-15	0-5	5-15
			mg N / g soil[1]			
No-till	2.5	1.5	2.2	1.5	2.2	1.5
Conventional	2.5	1.5	1.5	1.5	1.4	1.5

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[1] from a graph, graph reading error approximately ± 0.1 mg/g (i.e., the equivalent in mg/g of the smallest unit on the ruler used to read the graph)

Differences in bulk density for the two tillages were small. Soil N loss from the top 5 cm was rapid during the first 5 years. This loss was three times greater for conventionally tilled plots. After 1975 changes were not significant. There was little change in total N below 5 cm.

b. NUTRIENTS IN RUNOFF AS AFFECTED BY TILLAGE

Nitrogen and phosphorus losses are often reported together in the same papers, though this is not always true. For this review, papers discussing nitrogen and phosphorus will be followed by papers dealing only with phosphorus in runoff.

The earlier studies on the pollution consequences of crop production focused on dissolved N and P in runoff. Gradually an awareness of the importance of the particulate runoff load and the contamination of shallow ground water developed. Complete nutrient budget studies, accounting for dissolved, particulate, and gaseous, surface, subsurface, and atmospheric inputs and outputs, are rare (non-existent for P).

i. NITROGEN AND PHOSPHORUS IN RUNOFF

Conservation tillage methods have been shown to reduce erosion. Since about 75% of agricultural phosphorus loss is particulate, erosion reduction and phosphorus control are often considered to be synonymous. Phosphorus losses have been shown to decrease with conservation tillage (Mueller et al., 1984b; McDowell and McGregor, 1980; Siemens and Oschwald, 1976; and Römkens et al., 1973). However, some studies have shown that, at certain times of the crop year, soluble phosphorus losses may be greater with conservation tillage than with conventional tillage (Römkens et al., 1973; Barisas et al., 1978; McDowell and McGregor, 1980; Johnson et al., 1979).

These studies attributed the higher soluble phosphorus concentrations with conservation tillage to the localization of phosphorus on the field surface from broadcast fertilizer application and to a release of phosphorus from unincorporated crop residues. A third factor, apropos more to farm conditions than to these studies, is that many farmers apply more fertilizer to conservation tillage fields than to conventionally tilled fields to achieve comparable yields.

Alberts and Spomer (1985) measured ammonium (NH_4), nitrate (NO_3), and phosphate (PO_4) in runoff and leachate from corn grown on deep loess in Iowa, contrasting contour conventional tillage to contour till-plant (reduced tillage) systems. Fertilizer was applied at the recommended levels. Samples were taken monthly for ten years.

They showed that <u>dissolved</u> N and P pollution from contour reduced till corn exceeded that from contour conventional tilled corn. However, particulate nutrient loads were not measured. (Commonly the bulk of runoff N and P loss exits with the sediments. For example, Langdale et al. (1985) report 90% of the P loss from conventionally tilled corn to be sediment bound.)

Alberts and Spomer also showed that subsurface losses of dissolved nitrate greatly (by a factor of 10) exceeded surface losses. Ammonia losses in surface and subsurface flow were generally of the same magnitude. So were phosphorus losses (Table 43).

Reduced tillage delivered larger dissolved annual N and P load. Particulate annual load is not known.

Dissolved concentrations of N and P in surface runoff from soils under conservation tillage often are higher than concentrations in runoff from soils under conventional tillage. The authors list possible reasons including incomplete incorporation of surface-applied fertilizer, direct dissolved nutrient contributions from decaying plant residue, and higher dissolved N and P concentrations in the surface soil because of residue accumulation and decomposition.

P in surface water from the till-plant plot exceeded standard[4] 0.05 mg/l at least 13-fold. The conventionally tilled watershed produced lower surface and subsurface concentrations of PO4 and NO3.

Angle et al. (1983) compared the nutrients and sediment in runoff from conventional and no-till continuous corn fields in the Maryland Piedmont (the Middle Patuxent watershed.) The soil was Manor loam with a pH of 5.9, 27 g/kg organic matter, and a cation exchange capacity of 160.9 mmol (+) per kg. Extractable P and total N contents in the top 15 cm of soil prior to the study were 87 and 597 mg/kg, respectively. This study is notable for the care the authors took to demonstrate that the two watersheds were similar hydrologically before the experiment, and to allow a transition time of two years for the tillages to become established before contrasting their water quality impacts.

The conventional till watershed had a slope of 6% and a total area of 0.37 ha. Prior to planting, the conventionally tilled watershed was contour plowed, disc-harrowed, and cultipacked. The corn was planted in April and harvested in October. Following harvest,the stalks were mowed. The soil was not tilled in the fall, nor was a winter cover crop planted in the watershed.

The no-till watershed had an area of 0.26 ha and a slope of 7%. It was planted using no-tillage procedures, although following harvest, the stalks were mowed. A cover crop of barley was seeded immediately after mowing. Both watersheds received 4.02 g/m^2 of N (30% liquid solution) and 44.8 g/m^2 of surface applied 6-24-24 (N-P-K) dry fertilizer (67 kg/ha N and 108 kg/ha P). Table 44 presents the results of their runoff analyses.

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[4] the 1976 EPA recommended standard for water entering lakes or reservoirs. USEPA, 1976. Quality Citeria For Water. Washington, D.C.

Table 43
Dissolved Nutrient Losses on a Contour Conventional Corn Field Compared With Losses from a Contour Till-Plant Field on Hilly Deep Missouri Loess
Alberts and Spomer, 1985[1]

10 Year Average Dissolved Nutrient Losses (as atomic N or P)

Fertilizer Applied[2]		Contour Till-plant kg/ha yr						Contour Conventional kg/ha yr					
N kg/ha	P yr	Surface			Ground Water			Surface			Ground Water		
		NO3	NH4	PO4	NO3	NH4	PO4	NO3	NH4	PO4	NO3	NH4	PO4
188	36[3]	2.11	0.50	0.25	29.8	0.21	0.03						
188	40							1.36	0.36	0.10	9.00	0.20	0.03

Dissolved N Losses as Percent of Nitrogen Applied

Contour Till-plant %				Contour Conventional %			
Surface		Ground Water		Surface		Ground Water	
NO3	NH4	NO3	NH4	NO3	NH4	NO3	NH4
1.1	0.3	15.9	0.1	0.7	0.2	4.8	0.1

Dissolved P Losses as Percent of Phosphorus Applied

contour till-plant %		contour conventional %	
surface	ground water	surface	ground water
PO4	PO4	PO4	PO4
0.7	0.1	0.3	0.1

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[1] Data read from histograms

[2] 80% of the recommended level of N was applied as anhydrous ammonia, the other 20% was ammonium nitrate. P was applied as superphosphate, at the recommended dose for the crop.

[3] No P was applied in 1982 and 1983.

Table 44

Sediment and Nutrients in Runoff from Conventional and No-till Continuous Corn

Angle et al., 1983

Date	Rain	Runoff		Susp.Sed.		NH_4-N		NO_3+NO_4-N		Total N		Ortho-P		Total Sol.P		Total P	
		Conv.	No-till	Conv.	No-till	Conv.	No-till	Conv.	No-till	Conv.	No-till	Conv.	No-till	Conv.	No-till	Conv.	No-till
	cm	l/ha		g/ha		g/ha		g/ha		g/ha		g/ha		g/ha		g/ha	
1980																	
16 Jan	1.35	270	390	10	10	0.3	0.2	1.5	3.2	4.4	4.1	0.32	0.17	0.36	0.18	1.0	0.3
14 Mar	3.53	4050	850	296	82	120.8	0.9	2.4	2.8	141.1	8.2	0.04	0.31	1.91	0.32	2.6	0.8
21 Mar	3.23	12700	3850	10	10	22.8	0.4	77.5	10.7	209.6	36.1	3.56	6.15	3.68	6.62	54.2	16.6
1 Apr	3.76	1350	0	530	0	1.2	0.0	3.1	0.0	11.1	0.0	0.28	0.00	0.35	0.00	2.0	0.0
26 May	2.18	0	260	0	21	0.0	0.1	0.0	1.3	0.0	1.6	0.00	0.29	0.00	0.29	0.0	0.3
4 Jun	2.36	14050	1540	356	795	116.6	0.6	316.2	21.8	472.2	28.4	2.53	3.05	2.53	3.57	63.9	6.1
9 Jun	1.00	410	0	30	0	1.1	0.0	13.0	0.0	55.2	0.0	0.06	0.00	0.06	0.00	0.1	0.0
3 Jul	2.13	2160	0	7945	0	2.3	0.0	53.4	0.0	62.5	0.0	0.52	0.00	0.61	0.00	5.1	0.0
5 Aug	2.44	540	0	200	0	0.1	0.0	21.8	0.0	22.3	0.0	0.16	0.00	0.16	0.00	0.6	0.0
11 Aug	1.42	540	0	70	0	1.3	0.0	18.2	0.0	20.0	0.0	0.19	0.00	0.19	0.00	0.4	0.0
27 Oct	5.08	8110	230	868	24	3.2	0.0	64.0	3.5	108.6	4.5	6.49	0.14	6.50	0.15	24.3	0.5
25 Nov	3.18	11350	540	409	13	1.1	0.0	66.9	3.3	91.9	4.3	1.36	0.24	1.37	0.25	3.7	0.3
TOTAL	31.65	55530	7660	10723	955	270.8	2.2	638.0	46.6	1198.9	87.2	15.50	10.35	17.72	11.38	157.9	24.9
1981																	
1 Apr	0.07	0	270	0	68	0.0	5.6	0.0	2.2	0.0	18.9	0.00	0.89	0.00	0.87	0.0	1.0
16 May	2.74	L[1]	580	L	166	L	8.4	L	5.3	L	11.5	L	1.43	L	1.89	L	1.9
29 May	1.52	L	1160	L	321	L	16.5	L	22.7	L	33.9	L	2.77	L	2.71	L	3.6
2 Jun	1.09	1080	0	4740	0	0.0	0.0	7.8	0.0	18.8	0.0	0.11	0.00	0.08	0.00	2.8	0.0
8 Jun	1.70	1100	0	43738	0	1.9	0.0	19.1	0.0	95.7	0.0	0.18	0.00	0.15	0.00	16.2	0.0
15 Jun	4.11	L	8460	L	4095	L	1.0	L	37.3	L	169.2	L	18.78	L	18.36	L	0.0
22 Jun	1.02	5200	190	151952	25	14.8	0.0	85.9	0.9	244.5	2.5	0.56	0.52	0.56	0.49	28.4	0.6
6 Jul	3.38	L	190	L	10	L	0.0	L	1.1	L	1.9	L	0.47	L	0.52	L	0.6
TOTAL	5.63	24691[2]	10850	405358	4675	53.8[2]	30.6	289.0	69.5	913.3	237.9	23.74[2]	24.86	23.70	28.84	99.2	7.7
1982																	
18 Feb	1.42	270	580	5169	132	1.9	1.0	3.3	0.8	7.4	2.5	0.06	1.43	0.06	1.58	0.2	3.3
9 Mar	1.96	160	0	18	0	1.3	0.0	2.7	0.0	4.4	0.0	0.20	0.00	0.32	0.00	0.3	0.0
17 Apr	1.96	0	1540	0	147	0.0	3.6	0.0	3.1	0.0	10.7	0.00	0.80	0.00	1.80	0.0	2.8
20 May	1.09	2050	0	8031	0	36.1	0.0	23.0	0.0	66.8	0.0	0.42	0.00	0.51	0.00	8.6	0.0
21 May	1.63	2700	0	15530	0	38.4	0.0	27.8	0.0	107.8	0.0	0.26	0.00	0.28	0.00	12.7	0.0
24 May	1.08	5950	0	46010	0	48.5	0.0	20.6	0.0	105.9	0.0	0.55	0.00	1.79	0.00	2.8	0.0
30 May	3.91	1490	0	4344	0	11.3	0.0	7.2	0.0	20.4	0.0	0.26	0.00	1.20	0.00	5.2	0.0
7 Jun	1.55	540	0	807	0	0.6	0.0	5.3	0.0	9.4	0.0	0.05	0.00	0.01	0.00	0.0	0.0
13 Jun	5.41	14590	0	145479	0	24.3	0.0	30.2	0.0	154.4	0.0	7.13	0.00	0.39	0.00	71.3	0.0
17 Jun	2.26	13240	770	144722	93760	44.6	0.7	21.4	12.8	257.4	15.4	0.11	0.02	0.04	0.02	59.2	1.1
29 Jul	1.91	0	1540	0	173	0.0	0.0	0.0	2.2	0.0	3.9	0.00	0.32	0.00	0.30	0.0	0.5
6 Aug	3.51	810	0	323	0	0.8	0.0	2.7	0.0	5.2	0.0	0.08	0.00	0.06	0.00	0.2	0.0
TOTAL	25.69	41800	4430	370433	94212	207.8	5.3	144.2	18.9	739.1	32.5	9.12	2.57	4.66	3.70	160.5	7.7
GRAND TOTAL	72.98	*[3] 122021[2]	22940	* 786514	99842	* 532.4[2]	38.1	* 1071.2	135.0	* 2851.3	357.5	48.36[2]	37.78	46.08	39.92	* 417.6	40.3

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[1] Sample lost [2] Estimated [3] An * indicates a significant difference at $P \leq 0.05$ between tillage pairs

Frequently during the summer months, events were recorded where large amounts of runoff were collected from the conventional till watershed while no runoff was collected from the no-till watershed. Only four events were recorded over the entire three year study where runoff occurred from the no-till watershed but not from the conventional till watershed. Three of these events occurred in the early spring.

The authors explain that during this time the surface of the conventional till soil was relatively rough as a result of the recent cultivation. Wheel tracks and the rough, cloddy structure may have provided depression storage. The no-till soil surface, due to a lack of cultivation, remained smooth. Thus there existed a short period of time during the spring when greater amounts of runoff were expected from the no-till watershed.

The only runoff quality parameters not affected by tillage were ortho-phosphate and total soluble phosphate (see Grand Total, Table 44).

One or two rainfall events each year contributed the bulk of the yearly total sediment loss. For example, the runoff events of 13 June and 17 June 1982 were responsible for 78% of all sediment loss from the conventional till watershed for the entire year. During the same year, 99% of all sediment loss from the no-till watershed occurred on 17 June 1982. During 1980 the suspended sediment loss from the conventional till watershed was over 11 times greater than the loss from the no-till watershed.

See Angle et al. (1989) for a report on nitrate concentrations beneath these corn plots 1984-1987 (Table 68, Section V.C.2.c.i.).

A snapshot picture of runoff effects of no-till is given by Ross et al. (1987). They contrasted the runoff from conventional and no-till soybeans on three soils, conventional and chisel plow soybeans on one soil, and conventional and no-till corn on two soils in Virginia. The plots, soils, and dates of simulated rainfall tests were:

(1a) No-till corn on Goldsboro fine sandy loam, 2% slope, May 15-16, 1985

(1b) Conventional corn on same soil and dates

(2a) No-till soybeans on Slagle fine sandy loam, 5% slope, June 19-20, 1985

(2b) Conventional soybeans on same soil and dates

(3a) No-till soybeans on Suffolk loamy sand, 2% slope, July 26-27, 1985

(3b) Conventional soybeans on same soil and dates

(3c) Chisel plowed soybeans on same soil and dates

(4a) No-till corn on Tetotum fine sandy loam, 6% slope, August 6-7, 1986

(4b) Conventional corn on same soil and dates

(5a) No-till soybeans on Caroline fine sandy loam, 6% slope, August 11-12, 1986

(5b) Conventional soybeans on same soil and dates.

Simulated rainfall was applied to each plot over a two-day period. The application pattern was 60 minutes of rain (4 to 5 cm) applied the first day, followed 24 hours later by two 30 minute applications separated by half an hour. The rainfall delivered on the second day totaled about 5 cm. The authors characterized this schedule as consisting of a dry run, a wet run, and a very wet run.

The 5 cm of rainfall applied the first day approximated the one hour duration two year recurrence interval storm in Virginia, even though its kinetic energy for its intensity (5 cm/hr) was half that of natural rainfall.

The percent reduction in sediment, nutrients and surface water runoff are presented in Table 45. A seasonal effect is evident, but unfortunately, each entry is for a different soil (even though 4 of the five soils are fine sandy loams), so a soil effect may confound the seasonal effect.

Note that most of the negative entries (when the constituent in no-till runoff exceeded that from conventional plot runoff) are for dissolved species.

The experimental results expressed as areal yield are found in Table 46. The interpretation of these results is difficult because the lack of replication precludes significance testing and because the experiments were conducted on different soils.

Table 45
Percent Reduction in Sediment and Nutrients in Runoff From Conservation Tillage Test Plots
Ross et al., 1987

run	TSS	NH_4	NO_3	TKN	T-N	T-P	o-P	TKN-F	TP-F	FLOW[1]
			Percent Reduction in Yield[2]							
May corn, no-till versus conventional										
dry	99	76	80	91	89	94	-	66	88	85
wet	99	72	81	87	86	90	-	74	79	87
very wet	96	13	42	70	65	72	-	30	50	50
2-day[3]	98	62	70	85	83	88	-	57	72	76
August corn, no-till versus conventional										
dry	99	69	59	95	94	96	43	82	99	94
wet	87	74	**-863**	57	41	86	**-27**	38	91	64
very wet	64	84	**-4704**	57	35	70	**-174**	39	**-400**	24
2-day	82	80	**-674**	70	57	82	**-68**	53	89	62
June soybeans, no-till versus conventional										
dry	99	90	54	76	76	96	-	28	32	53
wet	99	92	76	90	90	95	-	82	50	70
very wet	99	83	62	78	78	81	-	84	68	37
2-day	99	89	63	80	80	92	-	66	45	53
July soybeans, no-till versus conventional										
dry	96	44	77	71	71	90	-	25	**-28**	40
wet	79	**-22**	40	34	34	62	-	**-56**	**-59**	0
very wet	94	**-359**	14	**-54**	**-51**	67	-	**-56**	**-143**	**-26**
2-day	92	**-166**	43	39	39	78	-	**-19**	**-61**	13
August soybeans, no-till versus conventional										
dry	100	90	76	98	98	100	**-36**	93	87	95
wet	100	64	**-21**	85	80	95	**-373**	69	62	75
very wet	99	27	3	75	67	92	**-589**	53	2	50
2-day	100	66	29	92	88	97	**-291**	74	55	75
July soybeans, chisel plow versus conventional										
dry	23	19	35	32	32	46	-	**-5**	1	21
wet	16	33	40	72	72	24	-	**-18**	2	12
very wet	42	44	44	3	5	34	-	18	**-17**	21
2-day	28	28	39	36	36	39	-	**-1**	**-2**	19

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[1] TSS = total suspended solids, NH_4 = ammonium, NO_3 = nitrate, TKN = total Kjeldahl nitrogen (particulate and dissolved species), T-N = total nitrogen, o-P = ortho-phosphate, TKN-F = filtered total Kjeldahl nitrogen (dissolved species only), TP-F = filtered total phosphorus.

[2] 100 * ([conventional]-[reduced tillage])/[conventional]

[3] total losses in runoff over two-day test period

The May corn experiment was intended to capture the most vulnerable season for soil erosion, 4 to 5 weeks after planting when crop canopy was not fully developed. Contrasted to the August experiment, it looks like the nitrate delivery from no-till stays constant May to August, but that the nitrate delivery drops considerably for conventional tillage after the early crop stages. With dissolved phosphorus (TP-F) it looks like conventional tilled corn delivers more than no-till in either season, with the exception of when the ground is soaked, then no-till delivers more dissolved phosphorus.

Table 46
Sediment and Nutrients from No-till, Chisel, and Conventional Tillage Plots Resulting from Simulated Rainfall Applications
Ross et al., 1987

tillage	TSS	NH_4	NO_3	TKN	T-N	T-P	o-P	TKN-F	TP-F
				kg/ha·experiment					
May Corn									
No-till	10	0.11	0.15	0.40	0.55	0.16	-	0.30	0.03
Conventional	558	0.30	0.51	2.65	3.17	1.25	-	0.71	0.10
August Corn									
No-till	42	0.06	0.22	0.48	0.70	0.08	0.02	0.15	0.02
Conventional	237	0.32	0.03	1.62	1.65	0.43	0.01	0.32	0.22
June Soybeans									
No-till	10	0.04	0.03	0.58	0.61	0.05	-	0.36	0.01
Conventional	914	0.32	0.08	2.93	3.00	0.67	-	1.07	0.02
July Soybeans									
No-till	101	0.13	0.03	1.45	1.48	0.28	-	0.53	0.09
Conventional	1225	0.08	0.05	2.36	2.42	1.29	-	0.44	0.06
Chisel	881	0.05	0.03	1.51	1.54	0.79	-	0.45	0.06
August Soybeans									
No-till	5	0.17	0.14	0.31	0.45	0.08	0.07	0.25	0.05
Conventional	1398	0.49	0.20	3.65	3.84	2.46	0.02	0.99	0.12

Laflen and Tabatabai (1984) studied the effects of a single simulated storm applied some unspecified time in May or June on plots on two soils.

Two soils were contrasted, Clarion sandy loam, a fine loamy mixed mesic Typic Hapludoll with a 5% slope, and Monona silt loam, a fine loamy mixed mesic Typic Hapludoll with slope of 11%. Before the simulation the top 15 cm of the soils had the following characteristics:

	%organic C	%total N	%clay	%sand	%moisture
Clarion	1.66	0.145	18	53	13
Monona	1.63	0.169	21	5	21

Fertilizer was applied the day before tillage and planting and 7 weeks before rainfall simulation for the Clarion soil. Therefore we assume the sampling occurred in June. The Monona soil was fertilized three weeks before rainfall simulation so the sampling could have taken place in either May or June.

Rain was applied at a rate of 6.3 cm/hr and samples of runoff were collected at five minute intervals for the first half hour and at ten minute intervals thereafter, for a total of one hour (so the results are not strictly comparable to studies using flow weighted sampling). This simulated storm resulted in 3.7 cm of runoff. In a typical year in central Iowa (the location of the study) about 7.5 cm of runoff is expected.

This study is interesting in that it shows how soil type is the most influential factor in runoff pollution during seedbed crop stage; more important than rotation, tillage, or type of standing row crop.

Three tillages and four rotations were compared for their effects on nutrient and sediment concentration and yield. All tillage and planting operations were oriented parallel to the surface slope. The conventional tillage included spring moldboard plowing, two diskings and planting. Chisel plow consisted of of spring chisel plowing, shallow disking, and planting. No-tillage received no tillage before or after planting.

The rotations consisted of continuous corn (CC), continuous soybeans (BB), corn following soybeans (BC), and soybeans following corn (CB). We infer from the authors' two-letter notation that the nutrient samples were taken in the second year of a two year rotation, though this is not clear from the text. The yields from a large single event occurring before canopy closure are reported in Table 47.

The biggest effect demonstrated was that due to soil type. Dissolved nutrient losses from the steeper, finer textured Monona soils were three times that from the Clarion soils and particulate nutrients were 20 times greater.

There were no significant tillage or rotation effects for runoff volume on the Monona soil, but on the Clarion soil a residue effect is apparent (the greater amount of cover from corn stubble than from soybean residue protects the soil from erosion and associated nutrient loss).

Table 47
Losses of Water, Sediment, and Nutrients During 60 Minutes of Simulated Rainfall In Early June By Tillage and Rotation on Two Iowan Soils
Laflen and Tabatabai, 1984

Tillage/ Rotation	Runoff	Erosion	Dissolved Nutrients			Sediment	Nutrients	Residue cover
			NH_4-N	NO_3-N	PO_4-P	Total N	Total P	
	cm	....	..kg/ha per	experiment	....			%
CLARION SOIL								
Conventional	1.19ab	2380a	0.024a	0.027a	0.009a	5.21a	1.65a	4.6a
Chisel	1.10a	1250b	0.076a	0.027a	0.020a	3.01b	1.02b	15.4b
No-till	1.52b	820b	0.192b	0.208b	0.089b	2.28b	0.74b	41.6c
LSD(0.05)	0.33	680	0.089	0.154	0.016	1.151	0.45	5.3
BB	1.79a	2300a	0.083a	0.029a	0.051a	5.40a	1.74a	14.4a
CB	0.92b	450b	0.050a	0.067ab	0.024b	1.33b	0.39b	23.9b
BC	1.51a	2550a	0.208b	0.068ab	0.042a	5.79a	1.94a	15.5a
CC	0.87b	630b	0.051a	0.223b	0.054a	1.48b	0.46b	28.6b
LSD(0.05)	0.41	790	0.103	0.198	0.018	1.74	0.52	6.1
MONONA SOIL								
Conventional	3.37a	46620a	0.078a	0.107a	0.051a	75.43a	35.87a	0.4a
Chisel	3.21a	31520b	0.201a	0.275a	0.101a	55.17a	25.48b	11.1b
No-till	3.47a	11640c	0.691b	0.668b	0.289b	23.14c	10.54c	28.5c
LSD(0.05)	0.68	10060	0.273	0.324	0.058	17.33	8.15	4.8
BB	2.91a	29890ab	0.138a	0.130a	0.115a	51.20a	24.71a	11.9a
CB	3.63a	36820b	0.124a	0.204a	0.145a	60.17a	28.58a	14.4a
BC	3.46a	27450ab	0.457b	0.430ab	0.166a	47.87a	21.67a	13.4a
CC	3.39a	25550a	0.574b	0.635b	0.162a	45.75a	20.88a	13.6a
LSD(0.05)	0.78	11610	0.315	0.374	0.067	20.01	9.41	5.6

Dissolved nutrients from no-till plots were also higher than for the other tillages, but these losses were dwarfed by the sediment associated losses, which were highest for conventional tillage, as opposed to either conservation tillage practice.

The same pattern is seen with the Monona soil. The dissolved nitrogen species are higher for the rotations ending in corn, but again, the distinction is trivial compared to the sediment associated nitrogen loss, which is not crop related.

Except in soil erosion, the effect of crop type, soybeans versus corn, on nutrient loss was insignificant.

McDowell and McGregor (1980) studied nutrient losses in runoff from soybeans under various no-till rotations and under conventional tillage on highly erodible Providence silt loams (Typic Fragiudalfs) of loessial origin underlain by coastal plain sands in northern Mississippi.

Table 48 records the fertilizer application, runoff and soil losses from each treatment. Conventional tillage consisted of moldboard plowing in the spring, disking, harrowing, bedding, planting,and follow up cultivation as needed to control grass and weeds. Fertilizer was placed about 8-10 cm deep and 7 cm to the side of the seed with a double-disk opener on the conventional planter.

No-till soybeans and corn were planted with a no-till planter by cutting through existing crop residues into the soil with a fluted coulter. The fluted coulter prepared a seedbed about 5 cm wide for seed placement. As with the conventional plots, fertilizer was placed about 7 cm to the side of the seed with a double-disk opener.

The authors reported that cutting through surface crop residues with the opener was difficult and an unquantified amount of fertilizer was left on the surface instead of below it.

The results of the second year of their 2-year project are reported in Table 49. Rainfall that year was 169 cm, above the average 133 cm for that area.

The authors reported that over 50 percent of the total nutrient loss occurred in the first month after fertilization and planting. Also relatively high P concentrations were observed in October and November, which were attributed to release of P from crop residues.

The values in Table 48 give an idea of the variability of nutrient export that can be observed with the same main crop and tillage practice due to previous crop in the rotation and fertilizer schedule.

Table 48
Annual Fertilizer Application, Runoff, and Soil Loss
McDowell and McGregor, 1980

Cropping System	Tillage	Fertilizer		Runof	Soil Loss
		N	P	cm	t/ha·yr
Continuous soybeans	Conven.	0	29	83.2	29.4
Continuous soybeans	No-till	0	29	42.8	0.4
Double cropped soybean-wheat	No-till	95[1]	49	31.3	0.2
Soybeans after corn rotation	No-till	0	29	88.2	3.6
Corn after soybeans rotation	No-till	136[2]	20	66.2	1.0

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[1] Ammonium nitrate top dressed on the wheat in the spring

[2] 45 kg N per ha applied to corn at planting, 91 kg N per ha applied as ammonium nitrate to the soil surface as a side dressing

Table 49
Annual Nutrient Loss from No-till and Conventional Soybean Plots
McDowell and McGregor, 1980

Cropping System	Tillage	N Losses		P Losses	
		sol.	sed.	sol.	sed.[1]
		kg/ha·yr			
Continuous soybeans	conventional	2.5	42.0	0.2	16.7
Continuous soybeans	no-till	2.5	2.5	2.0	1.0
Double cropped soybean-wheat	no-till	7.5	2.8	2.2	1.0
Soybeans after corn rotation	no-till	6.6	16.9	0.5	6.1
Corn after soybeans rotation	no-till	15.0	6.3	2.2	2.1

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[1] "solution" and "sediment" losses; read from graphs, graph reading error approximately ± 2.5 kg/ha·yr, i.e., the equivalent in hg/ha•yr of the smallest unit on the ruler used to read the graph.

Nutrient losses in solution exceeded particulate nutrient losses for some cropping systems, while the reverse relationship held for others.

Total runoff losses of N and P (solution plus sediment) from no-till soybeans were about 1/10 and 1/6, respectively, of losses from conventional tillage soybeans.

For the two no-till cropping systems where nitrogen fertilizer was surface applied (double cropped soybean/wheat and corn after soybeans), solution nitrogen (nitrate plus ammonium) runoff losses exceeded sediment total Kjeldahl nitrogen losses.

Following the typical pattern, most of the N and P was transported by sediments from conventional till soybeans. Solution P losses were significantly greater from no-till practices than from conventional tillage. Total (solution plus sediment) plant nutrient losses were smallest from no-tillage plots.

The authors proposed 5 mechanisms to account for the increased soluble P in runoff from no-till plots: (1) insufficient sediment to sorb P from solution, (2) the additional P fertilizer applied to the double cropping system, (3) the limited sorption of fertilizer P by the soil caused by partial fertilizer burial with no-till (4) the release of P from crop residues, and tentatively, (5) a greater phosphate supplying capacity of sediments in runoff from no-till.

In an often cited article, Beaulac and Reckhow (1982) reviewed the literature on the influence of land use on nutrient export to surface runoff. Table 50 is excerpted from this paper.

They concluded that the type of fertilizer (commercial or manure) used is not as important to nutrient flux as the time of application. Also incorporation of fertilizer after application reduces export. Excessive or insufficient fertilization can increase nutrient losses to runoff. Nutrient export is affected by tillage practice, with conservation tillage methods resulting in a smaller nutrient loss to surface runoff than conventional tillage.

The type of crop, non-row (grains) versus row crops, also influence nutrient loss to runoff. Though the median loss values reported were not statistically distinguishable, the highest reported loss from non-row crops was ten times smaller than the highest loss from row crops.

The issue of how manuring affects runoff from cropland is dealt with in more detail in the chapter on animal waste.

In a "208" study of the Occoquan watershed in northern Virginia (Metropolitan Washington Council of Governments, 1978) annual loading rates of nitrogen and phosphorus from small agricultural watersheds and urban land uses were determined. Table 51 presents average annual N and P concentrations in runoff for the agricultural and construction sites.

The conventional till corn on was grown on a dairy farm. Manure spreading observed at harvest and during winter months. 800 lbs of 10-10-10 fertilizer was applied in April. The minimum till corn had grass for an inter-season cover. The field was plowed after harvest. The grass winter cover was well established. Herbicides were applied 2 weeks before planting corn.

Table 50
Land Use and Total Nutrient Export in Runoff from Row Crops
Beaulac and Reckhow, 1982

Land Use	Fertilizer kg/ha yr N	Fertilizer kg/ha yr P	Soil	Precipitation cm/yr (range)	Runoff cm/yr (range)	Total Nitrogen Export kg/ha yr (range)	Total Phosphorus Export kg/ha yr (range)
Corn[1]	0	0	silt loam	77 (66-78)	11 (8-22)	3.96 (3.61-5.53)	1.22 (1.22-1.49)
Corn[1] fresh manure applied in winter	109	39	silt loam	77 (66-78)	12 (6-19)	7.97 (3.05-26.88)	2.00 (1.03-5.77)
Corn[1] fermented manure applied in spring	102	44	silt loam	77 (66-78)	12 (6-15)	3.38 (3.35-5.32)	0.75 (0.68-0.96)
Corn[1] fermented manure applied in spring	78	33	silt loam	77 (66-78)	12 (6-16)	2.88 (2.81-5.07)	0.95 (0.76-1.18)
Corn[2]	0	0	silt loam		12 (9-14)	4.33 (4.08-4.58)	1.30 (1.00-1.60)
Corn[2] fresh manure applied in winter	108	39	silt loam		9 (7-12)	15.25 (4.44-26.06)	3.40 (1.13-5.66)
Corn[2] fermented manure applied in spring	108	34	silt loam		9 (7-10)	4.22 (3.68-4.76)	0.81 (0.73-0.90)
Corn[2] liquid manure applied in spring	108	39	silt loam		9 (8-11)	3.88 (3.70-4.07)	0.94 (0.91-0.97)
Corn[3]	112	29	loam	63	9	79.6	18.6
Corn[4]	29	81	loam	66	10	44.2	14.0
Corn[4]-surface spread manure	268	124	loam	66	4	27.9	8.6
Corn[4] plowdown manure	268	123	loam	66	4	33.0	9.8
Corn[5]	56	29	loam	57	5	14.24	3.14
Corn[15]	112	29	loam	57	8	23.63	5.55
Corn[6] contour planting	448	64	deep loess	80 (63-106)	5 (1-13)	8.69 (2.2-72.47)	0.59 (0.09-2.12)
Corn[6] contour planting	168	39	deep loess	79 (62-104)	4 (2-10)	5.36 (1.69-43.71)	0.35 (0.08-1.29)
Corn[6] contour planting	280	64	deep loess	74 (53-102)	2 (<1-11)	2.1 (0.67-26.7)	0.26 (0.02-0.61)
Corn[7]	284	54	sdy cly loam	108	13	12.42	2.21
Corn[8]	100	35	silt loam	87		3.29	0.40
Soybeans[9] two crops per year, conventional tillage	0	29	silt loam	144	56	46.50	17.64
Soybeans[9] two crops per year, no-till	0	29	silt loam	144	28	5.1	2.6
Soybeans/Corn[9] two crops per year, no-till	0	29	silt loam	144	55	23.0	7.2
Corn/Soy[9] two crops per year, no-till	136	20	silt loam	144	50	19.3	3.7

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[1] 3-year median, Minshall et al., 1970
[2] 2-year mean, Hensler et al., 1970
[3] 10-year mean, Young and Holt, 1977
[4] 3-year mean, Young and Holt, 1977
[5] 6-year mean, Burwell et al., 1975
[6] 7-year median, Alberts et al., 1978
[7] Smith et al., 1978
[8] Bradford, 1974
[9] 2-year mean, McDowell et al., 1976

The 30 ac pasture supported 30 head of Charolais. There was a pond on site. No cattle were grazed in third the quarter (December - February) and limited grazing was observed in the fourth quarter (March - May). In the deciduous forest leaf fall began in mid-October and was complete by mid-November.

Table 51
Average Total N and P Concentrations in Runoff by Land Use
Metropolitan Washington Council of Governments, 1978

Land use	number of observations	mg/l total N	mg/l total P
construction	19	3.2 ± 0.7	0.18 ± 0.13
conventional till corn	9	15.3	3.34
minimum till corn	29	4.4 ± 0.8	2.09 ± 0.83
cow pasture	30	5.1 ± 1.7	0.82 ± 0.40
forest	23	1.5 ± 0.5	0.17 ± 0.07

Storm runoff samples were taken June 1, 1976 through May 30, 1977. The farm sites were sampled year round but the construction site sampling started in September, 1976. Samplers at the construction site were often clogged with sediment and rendered unusable. The worst sampling problems in the agricultural watershed was freezing, which made the winter samples under-represented. The construction site had several periods of excavation during the study (laying of sewers, landscaping, home building, etc.). Proper sediment management techniques were employed.

Table 52 gives the proportion of the different forms of N and P in the samples, averaged over all the samples. Total nitrogen and phosphorus were measured on unfiltered samples. Relatively high proportions of organic N and P were observed.

Table 52
Nutrient Species in Runoff, as Percent of Total N or P
Metropolitan Washington Council of Governments, 1978

Land use	Organic-N %TN	NH_4-N %TN	TKN %TN	NO_3-N %TN	Ortho-P %TP	Organic-P %TP
Construction	35 ± 15	43 ± 10	77 ± 6	23 ± 6	25 ± 20	75 ± 20
Conventional	64 ± 18	7 ± 5	71 ± 19	29 ± 19	30 ± 21	70 ± 21
Minimum till	68 ± 14	11 ± 9	79 ± 12	21 ± 12	54 ± 20	46 ± 20
Cow pasture	88 ± 7	3 ± 2	91 ± 6	9 ± 6	18 ± 16	82 ± 16
Forest	93 ± 6	5 ± 4	99 ± 2	1 ± 2	2 ± 3	98 ± 3

Another study of the effects of land uses on runoff quality

Another study of the effects of land uses on runoff quality was conducted by the USGS and the Susquehanna River Basin Commission (Lietman et al., 1983). The study site was Pequea Creek basin, Pennsylvania, in the southern part of the Conestoga Valley, a carbonate and shale section of the Appalachian Piedmont. The soils of the basin are classed as belonging to the Conestoga-Hollinger, Duffield-Hagerstown, and Chester-Glenelg associations. Table 53 summarizes the nutrient concentration measurements for the three year study. Chisel tillage with fall disking was the method of corn cultivation.

Hubbard et al. (1982) determined the losses and distributions of nitrogen and phosphorus in runoff and sediment from heavily fertilized conventionally tilled watersheds during both the growing season and nongrowing seasons. The study was done on two small watersheds in Michigan having well drained Spinks loamy fine sand (Psammentic Hapludalf). Slopes on the two plots varied from 0 to 12%, with erosion evident on the steepest parts of the plots. Corn was planted on one plot and soybeans on the other. Tillage consisted of spring moldboard plowing. Fertilizers supplying 68-93-172 kg/ha N-P-K in 1974 and 68-131-172 kg/ha in 1975 were incorporated to 7.5 cm before planting (mid-May). Corn received supplemental N as a banded (3 cm depth) sidedressing at 130 kg/ha N on 8 July 1974 and 64 kg/ha N on 25 June 1975. Rows were aligned across the main slope.

For N movement studies during winter and spring, broadcast surface applications of NH_4NO_3 (early November) supplied 130 kg/ha N in 1974 and 252 kg/ha in 1975. Totals applied over the 2 year study period were corn: 712 kg/ha N and soybeans: 518 kg/ha N. Both crops received 224 kg/ha P. In all cases N was supplied as NH_4NO_3 and P as $Ca(H_2PO_4)_2$. Corn was harvested for silage, soybeans for grain, and soybean straw was collected and removed from the field. No cover crop was planted and winter weed cover was minimal. Runoff was measured continuously and sampled during 50-60 events over a two year period.

Ninety nine percent of the sediment loss over the 24-month period occurred during only 11 months (Table 53); 64-86% occurred during a single storm in April when no crop canopy was present. This storm exceeded the 20-year expectancy in erosion potential. A second storm of equal magnitude occurred in August, 1975, illustrating the protective effect of canopy.

Seventy to ninety five percent of the N in runoff and 90-98% of the P were carried in sediments, except during winter months when up to 80% of the N and 33% of the P were in solution, due to snowmelt effects (when column 5 Table 54 exceeds 100%). The authors suggest that interflow from thawed soil or nutrients leached from plant residue may have contributed to the increase in soluble N and P during winter months.

Table 53
Nutrients and Sediment in Base Flow and Storm Samples from Pequea Creek Study
Lietman et al., 1983

Site		Flow cfs	Nitrogen		Organic Nitrogen		Ammonium		Nitrate		Phosphorus		Ortho-Phosphorus		Organic Carbon		Suspended Sediment	
			Tot	Diss	Tot	Diss	Tot	Diss	Tot	Diss	Tot	Diss	Tot	Diss	Tot	Diss	Tot mg/l	%<0.062mm
									mg/l									
BASE FLOW																		
Creek	Mean	147.0	6.5	6.1	0.4	0.2	0.0	0.0	6.1	5.9	0.1	0.1	0.1	0.1	3.1	2.6	27	76
Forest	Mean	0.3	3.6	3.2	0.2	0.1	0.0	0.0	3.3	3.0	0.0	0.0	0.0	0.0	2.9	2.3	12	68
Corn	Mean	0.1	21.0	20.0	0.2	0.1	0.1	0.0	21.0	20.0	0.2	0.1	0.1	0.1	2.8	2.4	73	51
Resid.	Mean	0.1	3.7	3.5	0.4	0.2	0.1	0.0	3.3	3.2	0.0	0.0	0.0	0.0	6.2	5.6	65	47
Pasture	Mean	0.3	7.7	7.1	1.2	0.8	0.2	0.1	6.3	6.1	0.5	0.4	0.3	0.3	4.6	3.6	49	69
STORM FLOW COMPOSITE SAMPLES																		
Pequea Creek	Max	2660	11.0	8.8	7.4	1.7	0.5	0.5	7.6	7.4	3.6	1.1	1.5	1.0	47.0	43.0	3540	98
	Min	140	2.2	1.0	0.1	0.0	0.0	0.0	1.3	0.3	0.0	0.0	0.0	0.0	2.3	1.6	25	71
	Mean	453	6.8	5.1	1.9	0.6	0.2	0.2	4.7	4.3	0.8	0.2	0.2	0.1	13.0	5.6	647	88
Forest	Max	1.7	5.8	4.4	3.1	1.8	0.1	0.1	3.6	3.2	0.4	0.1	0.1	0.0	27.0	15.0	734	96
	Min	0.1	1.8	1.2	0.2	0.0	0.0	0.0	1.1	1.0	0.0	0.0	0.0	0.0	2.5	2.2	17	43
	Mean	0.6	3.6	2.7	1.3	0.5	0.1	0.0	2.3	2.2	0.2	0.0	0.0	0.0	13.0	5.8	207	75
Corn Field	Max	3.8	70.0	43.0	54.0	6.1	3.4	2.9	41.0	39.0	19.0	4.1	4.5	3.3	148.0	28.0	16000	98
	Min	0.1	8.1	6.5	1.0	0.2	0.0	0.0	3.2	3.2	1.6	0.4	0.0	0.0	7.9	5.4	24	85
	Mean	1.4	25.0	17.0	8.1	2.0	0.6	0.5	16.0	14.0	4.9	2.0	1.8	1.6	33.0	12.0	2080	92
Residential	Max	11.0	6.3	5.7	4.7	1.9	2.8	2.8	2.9	2.8	2.6	0.7	1.9	0.4	114.0	104.0	1140	100
	Min	0.2	0.8	0.6	0.2	0.0	0.0	0.0	0.3	0.2	0.1	0.0	0.0	0.0	5.0	3.9	13	60
	Mean	1.8	3.1	2.1	1.7	0.6	0.3	0.3	1.3	1.2	0.4	0.1	0.1	0.1	19.0	12.0	189	85
Pasture	Max	52.0	38.0	9.4	32.0	1.6	1.8	0.8	8.5	8.8	4.2	2.0	1.8	1.6	86.0	22.0	3840	99
	Min	0.6	1.4	1.5	0.8	0.3	0.0	0.0	0.2	0.4	0.1	0.1	0.0	0.0	4.4	0.3	56	67
	Mean	3.9	7.8	3.6	4.9	0.8	0.4	0.3	2.5	2.4	1.4	0.5	0.4	0.4	23.0	9.0	845	92

Table 54
Nitrogen and Phosphorus in Runoff from Conventionally Tilled Corn and Soybeans For the Eleven Months During Which 99% of the Sediment Loss Occurred
Hubbard et al., 1982

Date	Sediment Loss kg/ha	Sediment Loss mg/ml	Runoff cm	Runoff %[1]	Sediment Nutrients %N	Sediment Nutrients %P[2]	N:P Ratio Water	N:P Ratio Solid	Dissolved Mineral N %TDN[3]	Dissolved Mineral N $\frac{NH_4\text{-}N}{NO_3\text{-}N}$[4]	Dissolved Phosphate as %Total Diss. P	%Available P in Total Sediment P	%NH_4-N in Total Sed. N
CORN													
APR 75	12314	13.9	8.9	73	92	96	4.6	1.9	53	1.2	96	8	7.8
AUG 75	1628	3.3	5.0	40	81	87	3.1	1.9	51	1.3	92	5	0.9
MAR 76	1236	10.6	1.2	29	93	97	3.8	1.4	65	4.2	91	4	0.5
FEB 76	809	1.4	5.6	103	63	89	7.5	1.5	53	2.3	77	3	1.3
AUG 74	797	3.1	2.6	41	73	92	7.4	1.6	35	0.6	94	16	2.5
MAR 75	723	2.7	2.7	100	57	91	12.1	1.5	55	1.8	92	4	9.1
DEC 75	416	3.2	1.3	24	78	94	6.7	1.6	60	2.5	83	3	5.0
JUN 75	409	7.0	0.6	19	83	92	4.0	1.6	46	1.5	90	6	1.2
JUL 74	388	12.7	0.3	13	78	93	7.6	1.9	40	0.2	100	22	1.3
JAN 75	248	0.7	3.3	71	35	83	11.4	1.3	52	1.6	79	4	1.2
FEB 75	244	0.7	3.3	108	27	71	9.9	1.5	56	1.5	83	4	1.0
18 APR 75	12232	15.1	8.1	74	93	96	4.0	1.9	53	1.1	96	8	7.7
56 events[5]	6981	2.6	26.8	43	68	91	7.6	1.6	52	1.6	87	6	1.4
SOYBEANS													
APR 75	31688	34.3	9.3	76	94	98	4.0	1.5	51	1.6	86	11	1.1
AUG 75	2106	4.6	4.6	37	80	92	4.5	1.5	59	1.4	96	6	1.9
FEB 76	571	6.7	6.7	123	37	78	11.3	1.9	58	3.5	88	4	0.7
MAR 76	532	11.9	0.4	10	93	97	4.9	1.8	65	7.9	93	3	1.2
MAR 75	523	2.0	2.7	100	45	90	16.7	1.5	63	2.3	82	3	0.9
AUG 74	520	3.6	1.4	22	78	93	7.4	2.0	31	0.5	100	19	1.3
JUN 75	412	8.3	0.5	16	84	94	4.1	1.4	43	0.3	92	4	1.3
JUL 74	247	17.4	0.1	4	78	95	10.9	1.9	43	0.3	75	22	1.2
FEB 75	134	0.4	3.4	108	20	67	15.9	2.0	61	2.3	74	5	0.8
JAN 75	74	0.3	2.5	54	21	67	14.4	1.9	59	2.8	68	4	0.9
DEC 75	14	0.3	0.5	9	83	94	6.6	2.1	65	3.0	83	2	1.0
18 APR 75	31666	37.0	8.6	78	95	98	3.5	1.5	49	1.6	86	11	1.1
54 EVENTS[5]	5165	2.2	23.3	38	57	89	10.5	1.6	58	2.2	88	7	1.3

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[1] Percent of event precipitation, where event precipitation is the rain or snow immediately preceding and concurrent with runoff

[2] Sediment N or P as percent of total N or P loss

[3] Dissolved Mineral N (ammonium plus nitrate) as a percent of total dissolved N

[4] Dissolved Ammonium nitrogen to dissolved nitrate nitrogen ratio

[5] Excluding 18 April 1975

Table 55
Surface Runoff Quality from Conventionally Tilled Plots
Over a Two Year Period
Hubbard et al., 1982

Parameter	Period or Event	Corn	Soybeans
sediment, kg/ha	total	19,213	36,831
	all but 4/18/75	6,981	5,165
	4/18/75	12,232	31,666
	annual average	9,607	18,416
water, cm	total	34.9	31.9
	all but 4/18/75	26.8	23.3
	4/18/75	8.1	8.6
	annual average	17.5	16.0
dissolved P, kg/ha	total	1.90	1.70
	all but 4/18/75	1.33	1.04
	4/18/75	0.57	0.66
	annual average	0.95	0.85
sediment P, kg/ha	total	29.6	39.6
	all but 4/18/75	13.3	8.7
	4/18/75	16.3	30.9
	annual average	14.8	19.8
dissolved N, kg/ha	total	12.5	13.1
	all but 4/18/75	10.1	10.9
	4/18/75	2.4	2.2
	annual average	6.3	6.6
sediment N, kg/ha	total	52.4	59.7
	all but 4/18/75	21.0	14.3
	4/18/75	31.4	45.5
	annual average	26.2	29.9

Atmospheric deposition contributed to N and P during all seasons of the year. A snow sample from December 1975 contained 3.5 µg/g Kjeldahl N, 0.9 µg/g NO_3-N and 0.2 µg/g ortho-phosphate. Half (2.2 µg/g) of the N was organic.

Sediments were enriched, relative to soils, in total N, but not in extractable P. In sediment samples from the April 18 storm there was 5.6 times more N in sediment than in the soil (top 7.5 cm--the depth of fertilizer incorporation). The enrichment ratio for the soybean plot was 2.0. Enrichment ratios for available P were 0.9 and 1.2 for the corn and soybean plots, respectively.

Changing patterns of detachment and transport, and of sorted sedimentation, were observed during the course of individual events and from event to event. These patterns varied with runoff stage and duration and with rainfall intensity, and were reflected in widely varying sediment loadings, nutrient content, and nutrient distributions within events and between events of varying magnitude.

Particulates in early increments of runoff were comprised mainly of plant debris and dark colored fines in relatively stable suspension. In later increments, denser, light colored fine and very fine sand of lower N and P content appeared in proportions directly related to flow rate.

Table 55 compares the total loads generated during two years of storm water runoff monitoring, excluding the April 18, 1985 storm, to the loads resulting from that storm. For sediment and sediment associated nitrogen and phosphorus the single event loads exceeded the sum of loads from the remaining events.

iii. PHOSPHORUS IN RUNOFF

Langdale et al. (1985) studied the phosphorus losses from six cropping and tillage systems on a Cecil sandy loam soil (clayey, kaolinitic, thermic, Typic Hapludults) in the Georgia Piedmont. The systems were contour conventional till corn (winter fallow), contour conventional corn on terraces with a rye winter cover, contour conventional wheat and soybeans, two methods of conservation till barley/wheat and soybeans, and conservation tillage clover and sorghum.

The study ran from May 1974 through September 1975. The following table does not give annual yields or concentrations, but yields for the period represented by the crop stage.

To allow comparison with other papers in this report, the annual average yield was estimated (Table 56) by counting crop stage 4 entries twice, to get 24 months, with the exception of average annual PO4-P and sediment-P yields, for which Langdale et al. provided average annual estimates in the text. Comparing the two estimates on the entries for which there were both kinds of estimate, our estimates were from 10 and 20 percent too low. The annual average concentrations were estimated by us by runoff volume-weighted average of the 17-month period (75%) with stage 4's concentrations (25%).

The sediment and phosphorus species (dissolved phosphate as phosphorus, total dissolved phosphorus, and sediment associated phosphorus) concentrations are discharge weighted.

Table 56
P in Runoff from Contour Conventionally Tilled Corn on Sandy Loam of Upland Watersheds in the Georgia Piedmont
Langdale et al., 1985

	SEDIMENT		PO4-P		TOTAL DIS-P		SEDIMENT-P		RUNOFF
	yield kg/ha	conc g/l	yield g/ha	conc mg/l	yield g/ha	conc mg/l	yield g/ha	conc µg/g	cm
Crop stage[1]									
fallow thru 1	1750	1.74	101	0.11	132	0.13	1430	820	10.1
2 thru 3	105	0.53	11	0.06	14	0.07	60	570	2.0
4 NOV-JAN	3	0.15	5	0.25	5	0.26	4	1330	0.2
4 FEB-APR	95	0.12	144	0.20	160	0.21	120	1240	7.7
fallow thru 1	5270	6.00	68	0.08	78	0.09	3930	750	8.8
2 thru 3	102	0.35	20	0.07	23	0.08	240	2330	2.9
WHOLE 17 MO.	7325	2.31	349	0.11	412	0.13	5784	790	31.7
EST. ANN. AV.	3712	1.77	280	0.14	288	0.16	3800	914	19.8

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[1] the crop stages are
Fallow, inversion plowing to secondary tillage
1, plant establishment, SB to 50% canopy cover
2, development, 50% to 75% canopy cover
3, maturing crop, 75% cover to harvest
4, residue and stubble, harvest to plowing or new seeding

Around 90% of the annual phosphorus runoff loss from contour conventional corn occurs during the fallow, seedbed, and establishment (crop stage 1) periods (May through July), when the disturbed and unprotected land is most vulnerable to erosion forces.

Ortho-phosphate was the dominant species of dissolved phosphorus. The difference between total dissolved P and PO4-P was attributed to the presence of acid hydrolyzable organic forms of P. An estimate of this form of phosphorus for the fallow-corn sequence is total dissolved P (288 g/ha) minus dissolved phosphate P (249 g/ha), that is, 39 g/ha dissolved organic phosphorus (13.5% of the total dissolved phosphorus load).

Sediment phosphorus yield on the whole was more than 10 times higher than dissolved phosphorus. On an annual basis, 93% of the phosphorus lost was particulate.

The study also looked at conventional tilled corn with a rye cover crop and terraces. This management practice did not reduce dissolved phosphorus in runoff, but did reduce sediment phosphorus yields from 5.8 kg/ha to 1.6 kg/ha (71%).

Both terraced and unterraced watersheds received roughly the same amount of rainfall (206 and 196 cm), but more water infiltrated in the terraced watershed so that it produced 12.4 cm less runoff than did the other field.

Table 57 gives phosphorus loads normalized by centimeters of runoff because of differences in rainfall on the plots as well as yields and concentrations for the six different crop and tillage practices.

Table 57
Phosphorus Losses in Runoff from Contour Conventionally Tilled and Conservation Tilled Georgia Piedmont Watersheds
Langdale et al., 1985

	# OF STORMS	RUNOFF	PO4-P	TOTAL-P	DIS.PO4-P		TOTAL-P	
		cm	kg/ha·cm ro[1]		YIELD g/ha	CONC. mg/l	YIELD g/ha	CONC. mg/l
CONTOUR CONVENTIONAL WATERSHEDS								
FALLOW AND CORN								
FALLOW	15	7.9	0.02	0.05	158	0.20	395	0.50
CORN	10	11.9	0.01	0.31	119	0.10	3689	3.10
AVE. ANN. YIELD					280		4080	
RYE COVER AND CORN ON TERRACES								
RYE COVER	16	10.8	0.02	0.07	216	0.20	756	0.70
CORN	6	4.2	0.02	0.15	84	0.20	63	1.50
AVE. ANN. YIELD					300		1390	
WHEAT AND SOYBEANS								
WHEAT	7	3.4	0.06	0.07	204	0.60	238	0.70
SOYBEANS	1	0.1	0.03	0.04	3	0.30	4	0.40
AVE. ANN. YIELD					210		240	
CONSERVATION TILLAGE WATERSHEDS								
BARLEY/WHEAT AND SOYBEANS USING FLUTED COULTER								
BARLEY/WHEAT	8	3.6	0.10	0.16	360	1.00	576	1.60
SOYBEANS	2	1.7	0.02	0.09	34	0.20	153	0.90
AVE. ANN. YIELD					390		730	
BARLEY/WHEAT AND SOYBEANS USING IN-ROW CHISEL								
BARLEY/WHEAT	6	1.4	0.10	0.15	140	1.00	210	1.50
SOYBEANS	1	0.8	0.10	0.12	80	1.00	96	1.20
AVE. ANN. YIELD					220		310	
CLOVER AND SORGHUM USING IN-ROW CHISEL								
CLOVER	3	0.5	0.12	0.16	60	1.20	80	1.60
SORGHUM	0	0.0	-	-	-	-	-	-
AVE. ANN. YIELD					60		80	

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[1] Concentration (ppm) = kg/ha·cm runoff X 10

Total P runoff losses in all crop and tillage practices ranged from 0.1 to 4 kg/ha·yr and consistently related to soil loss, irrespective of watershed landscape or conservation practice imposed.

Mueller et al. (1984b) measured P losses from corn plots grown by three tillage methods, with and without manure application. The three tillages were conventional, chisel, and no-till. All tillage was performed across slope. There were four plots for each tillage system, two receiving 8 metric tons of manure per hectare (dry weight) prior to tillage. At planting 250 kg/ha of 10-26-26 was banded, and in June 125 kg/ha N was as NH_4NO_3 was sidedressed.

The test plots were on a Dresden silt loam (Mollic Hapludolls).

Runoff generated by simulated rainfall was collected during late May, mid-July and early September 1978 and early June and late August 1979. Rainfall was applied for 1 hour at 14.5 cm/hr. Total phosphorus, dissolved molybdate-reactive P, and algal available P were measured.

An analysis of variance was performed to find if the tillage type, manure application, of date of sampling, or interactions of these variables could be used to explain phosphorus concentration or losses. Total phosphorus and algal-available phosphorus data were log transformed.

Details of the most important effects found by their linear regression analysis are presented in Table 58, which gives the relative delivery of each phosphorus species (relative to conventional till without manure = 1.00).

In this table phosphorus concentration and flow weighted losses are indicated. Full year studies, rather than discontinuous sampling (as in this study) may show different conclusions regarding the phosphorus delivery from the three tillages.

In 1978 the first year of the study, tillage was not a predictor of phosphorus loss. This time-delay in establishment of typical no-till conditions is in agreement with results of previously mentioned studies. The tillage effect on runoff nutrient loss is discernable in the next year.

This paper can also give an idea of the proportion of the different phosphorus species in runoff for the described experimental conditions.

Table 58
Relative Phosphorus Delivery by Tillage and Manure Application
Mueller et al., 1984b

Total Phosphorus Concentration and Flow Weighted Losses, 1979

	Concentration	Losses
Conventional	1.00a[1]	1.00a
Conventional + Manure	0.94a	0.67ab
Chisel	0.63b	0.39bc
Chisel + Manure	0.48bc	0.04d
No-till	0.20d	0.30cd
No-till + Manure	0.42c	0.56bc

Dissolved Molybdate-Reactive P Concentration and Losses, 1978-9

	Conc. all dates in 1978 (residue removed)	Losses all dates in 1978 (residue removed)	Concentration 6/79	Concentration 8/79	Losses 6/79	Losses 8/79
Conventional	1.00c	1.00b	1.00d	1.00d	1.00c	1.00c
Conventional + Manure	1.19c	1.25b	0.94d	2.00cd	1.00c	1.67c
Chisel	1.16c	1.00b	1.44cd	1.09d	2.00c	0.33c
Chisel + Manure	3.13b	2.50b	5.56b	3.59c	2.00c	0.33c
No-till	1.41c	1.75b	1.06d	1.28d	2.50c	1.67c
No-till + Manure	6.70a	7.00a	18.72a	8.06b	40.00a	8.67b

Algal Available P Concentration and Losses, 1978 and 1979

	Losses, all dates in 1978, Residue Removed
Conventional	1.00ab
Chisel	0.73b
No-till	1.24a

	Concentration	Losses, all dates in 1979
Conventional	1.00ab	1.00b
Conventional + Manure	1.04ab	0.75bc
Chisel	0.67b	0.38cd
Chisel + Manure	1.03ab	0.13e
No-till	0.32c	0.46cd
No-till + Manure	1.57a	1.88a

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[1] Within each year and column category (concentration or losses), values followed by the same letter are not significantly different at p = 0.10 as tested by the least significant difference test.

The average concentrations for 1979 (1978 was incompletely reported in the paper) are recorded in Table 59. Residue was partially removed in 1978, but not in 1979. There is a lot of scatter in these data and dissolved phosphorus (DP) is averaged in an ad hoc fashion, due to how these data were reported. This table is only meant to give an idea of the breakdown of phosphorus species.

Table 59
Concentration of Phosphorus Species in Runoff from Corn Fields, 1979
Mueller et al., 1984b

Treatment	TP mg/l	DP mg/l	AAP[1] mg/l	AAPs[2] %[4]	DP %[4]	AAP %[4]	PP[3] %[4]
Conventional	3.56	0.041	0.726	19	1	20	99
Conventional + Manure	3.36	0.056	0.758	21	2	23	98
Chisel	2.23	0.054	0.488	19	2	20	98
Chisel + Manure	1.70	0.197	0.746	32	12	44	88
No-till	0.70	0.047	0.235	27	7	34	93
No-Till + Manure	1.50	0.597	1.140	36	40	76	60
average (all)	2.18	0.165	0.682	26	11	36	
standard deviation	1.11	0.219	0.303	7	15	22	
average with manure	2.19	0.283	0.881	30	18	48	
standard deviation	1.02	0.281	0.224	8	20	27	
average without	2.16	0.047	0.483	22	3	25	
standard deviation	1.43	0.006	0.245	5	3	8	

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[1] Total phosphorus, dissolved phosphorus, and algal-available phosphorus

[2] 100(AAP-DP)/TP, the portion of total P that is algal available and sediment derived

[3] Particulate P = TP-DP

[4] Percent of total P

When manure is surface applied, dissolved phosphorus concentrations increase significantly. This effect is observed most acutely in the spring and early summer, right after manure application.

Andraski et al. (1985) investigated the differences in P losses among the conventional till, chisel till, till-plant, and no-till systems when P fertilizer is subsurface banded. The reason for this fertilization practice was to isolate the effects of tillage distinct from the effects of surface application of fertilizer versus plowed down fertilizer.

The study was performed on Griswold silt loam plots, with tillage 2% off contour. Corn was grown for the four years of the study. Rainfall was simulated, and all runoff was collected and analyzed.

The conservation tillages reduced total P concentration and losses by controlling soil loss. No-till, chisel till, and till-planting reduced total P losses by an average of 81, 70 and 59%, respectively, relative to conventional tillage (Table 60).

Table 60
Total P Concentrations and Losses as Affected by Tillage When Fertilizer Was Banded Beneath the Surface
Andraski et al., 1985

	9/80	6/81	7/81	10/82	6/83	7/83
			Total P concentration, mg/l			
Conventional	2.34a[1]	3.42	7.89a	3.22a	2.90a	1.10a
Chisel	2.14a	-	4.33a	1.66bc	1.73b	0.88a
Till-plant	2.28a	3.49	8.94a	1.99b	1.42b	0.92a
No-till	0.99b	-	6.44a	1.14c	0.51c	0.48b
			Total P loss, kg/ha			
Conventional	1.33a	0.08a	2.30a	2.20a	1.75a	0.22a
Chisel	0.21c	trace,b	0.20c	0.37b	0.67b	0.11b
Till-plant	0.83a	0.04b	0.87b	0.40b	0.72b	0.10b
No-till	0.39b	trace,b	0.20c	0.24b	0.21c	0.05c

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[1] Values for each measurement month and parameter that are followed by the same letter are not significantly different at $p = 0.1$ as determined by Fisher's least significant difference test.

The no-till, chisel till and till-plant treatments reduced algal available P losses (dissolved and particulate) in runoff by an average of 63, 58 and 27%, respectively, relative to conventional tillage. Dissolved phosphorus concentrations and loads are reported in Table 61.

Table 61
Algal-available P Concentrations and Losses as Affected by Tillage
Andraski et al., 1985

Tillage	1980	1981		1982	1983	
	Sept.	June	July	Oct.	June	July
	Algal-available P concentration, mg/l					
Conventional	0.118b[1]	1.56	2.53a	0.450a	0.300a	0.231a
Chisel	0.148b	-	0.86b	0.241a	0.263a	0.257a
Till-plant	0.220a	0.54	0.62b	0.288a	0.247a	0.309a
No-till	0.085b	-	0.84b	0.220a	0.150b	0.246a
	Algal-available P loss, kg/ha					
Conventional	0.070a	0.017a	0.661a	0.299a	0.178a	0.046a
Chisel	0.015c	trace,b	0.059c	0.054b	0.103bc	0.033a
Till-plant	0.079a	0.007b	0.156b	0.066b	0.129b	0.039a
No-till	0.032b	trace,b	0.032c	0.045b	0.066c	0.023a

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[1] Values for each measurement month and parameter that are followed by the same letter are not significantly different at p = 0.1 as determined by Fisher's least significant difference test.

Andraski et al. claim that phosphorus losses would have been roughly twice those reported if the P fertilizer had been surface broadcast.

This conjecture is (loosely) based on their observation that for each sampling period, the comparative magnitude of total P loss and soil loss reductions afforded by each conservation tillage treatment relative to conventional tillage were similar, and that the reduction of total P loss was on the average only about 5% less than soil loss reduction.

They couple this result to the observation of Römkens et al. (1973) that total P loss reduction was about one half that of soil loss reduction for a no-till treatment where fertilizer P was broadcast.

Table 62
Dissolved P Concentrations and Losses as Affected by Tillage When Fertilizer Phosphorus Was Banded Beneath the Surface
Andraski et al., 1985

Tillage method	1980	1981		1982	1983	
	Sept.	June	July	Oct.	June	July
	Dissolved P concentration, mg/l					
Conventional	0.042b[1]	0.183	0.120a	0.052a	0.050a	0.039a
Chisel	0.083a	-	0.130a	0.062a	0.043a	0.061a
Till-plant	0.073a	0.134	0.148a	0.086a	0.070a	0.081a
No-till	0.072a	-	0.136a	0.076a	0.045a	0.079a
	Dissolved P loss, kg/ha					
Conventional	0.025a	0.003a	0.028a	0.035a	0.029a	0.008a
Chisel	0.008b	trace,a	0.007c	0.014b	0.015a	0.008a
Till-plant	0.028a	0.001a	0.016b	0.020b	0.034a	0.010a
No-till	0.027a	trace,a	0.006c	0.015b	0.023a	0.008a

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[1] Values for each measurement month and parameter that are followed by the same letter are not significantly different at $p = 0.1$ as determined by Fisher's least significant difference test.

c. PERCOLATION

i. SOLUTE LEACHING

In December, 1981 Tollner et al. applied potassium chloride to the surface of their experimental tillage plots (double cropped soybean/small grain on Cecil sandy loam), then sampled the profiles in February and July, 1982 and January, 1983. The chloride profiles, corrected for background level, are presented in Table 63. The chloride leaching front under no-till was usually deeper than that under conventional tillage, indicating a higher water throughput in the no-till profiles.

This was consistent with previous work on well-structured soils (McMahon and Thomas, 1976; and Tyler and Thomas, 1977 (on Maury silt loam and Lowell silty clay loam)) where larger amounts of nitrate were found to move more deeply in the soil in no-till relative to conventional till plots.

Table 63
Chloride Leaching Study by Depth and Tillage and Date After Surface KCl Application in December, 1981
Tollner et al., 1984

Depth	February, 1982		July, 1982		January, 1983	
	No-till	Conv.	No-till	Conv.	No-till	Conv.
cm	net chloride concentration, ppm					
8	33.6	48.0	21.6	26.4	4.8	7.2
22	33.6	84.0	33.6	60.0	19.2	21.6
38	84.0	196.8	33.6	112.8	16.8	48.0
50	134.4	175.2	81.6	153.6	7.2	129.6
68	105.6	153.6	43.2	79.2	7.2	93.6
82	64.8	67.2	38.4	40.8	10.6	98.4

Walter et al. (1979) generalize that the subsurface losses of nitrate and other substances with low affinity for sediment binding are at least five times the surface losses in humid regions.

In a release of preliminary results, Staver et al. (1987) compared surface and ground water chemical characteristics from no-till and conventional till continuous corn watersheds on silty, moderately well-drained, nearly level (0-3% slopes) silt loam soils of the Mattapex association of the Maryland coastal plain physiographic region.

"Conventional" tillage consisted of <u>chisel</u> plowing in conjunction with disking and use of a field cultivator. (Chisel plowing is often associated with reduced tillage, but since only 10% residue cover was left at planting, they did not call this conservation tillage.)[5] Fertilizer was applied in two increments: 40 to 50 kg/ha nitrogen at planting followed by a sidedressing of 100 kg/ha nitrogen 30 to 50 days after planting.

No-till reduced surface runoff relative to the "chisel-conventional" till system after spring tillage, but not before it. Little recharge of ground water occurred during the growing season, therefore, nitrate concentrations remained stable. When recharge occurred after the completion of the growing season, nitrate was flushed from the root zone into the shallow ground water.

Staver et al. speculate that no-till practices may actually promote elevated nitrate levels in ground water (though supporting

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[5] Apparently most of the farmers in the area chisel corn unless they intend to follow it by small grain, in which case they would moldboard plow for weed control.

data were not given in this paper). The most extreme case of this pattern would occur during years having dry growing seasons when both surface water export and crop uptake of nitrogen are limited. Dry conditions also produce an unfavorable environment for denitrification, another pathway for removal of nitrogen from the system.

Some studies show equal N fertilizer losses (leaching and denitrification) under no-till and conventional tillage systems for corn (Kitur et al., 1984, Section VI.A.1.a.).

Kanwar et al. (1985) found higher nitrate leaching losses under conventional tillage seedbed conditions when they applied large amounts of artificial rainfall comparable to the 100 year 24-hour storm to test plots. The rainfall was delivered in two increments, equivalent to the 100 year 6-hour storm and the 24-hour storm (cumulatively) on three treatments, no-till seedbed preparation (no crop planted) with 150 kg/ha NO_3-N liquid KNO_3 surface applied, moldboard plow with surface broadcast fertilizer (same rate), and moldboard plow with fertilizer incorporated by disking. The plots were on Clarion-Nicollet loam soil. Results are found in Table 64.

Table 64
Average Dissolved Nitrate-Nitrogen Content in the Soil Profile After Two Large Simulated Rainfall Events on Prepared Seedbeds
Kanwar et al., 1985

	No-till Before Rainfall	No-till Surface[1]		Plow Surface		Plow Incorporated[2]		LSD(0.05)	
Rainfall	0cm	13cm	19cm	13cm	19cm	13cm	19cm	13cm	19cm
Soil Depth cm					kg/ha NO_3-N				
0-15	229.6	97.3	72.5	19.9	6.4	9.2	4.7	25.3	17.3
15-30	73.4	83.1	74.9	68.6	33.5	48.0	29.6	68.1	22.5
30-45	53.4	63.3	59.9	71.2	53.0	68.2	35.7	29.2	28.6
45-60	30.3	50.1	52.5	53.9	49.8	75.9	41.2	29.9	40.1
60-90	25.3	56.4	72.0	63.4	77.5	117.4	82.0	42.6	34.9
90-120	19.5	42.4	56.9	35.0	58.7	44.8	70.6	16.0	14.5
120-150	20.3	28.3	39.6	30.3	38.6	32.3	41.5	5.9	15.6
total nitrate-N in the 0-150 cm profile									
0-150	451.8	420.9	428.3	342.3	317.5	396.6	305.3	91.9	86.2
net nitrate-N leaching loss below the 150 cm depth (calculated)									
>150		28.9	21.5	121.5	146.3	44.2	135.5	46.6	77.4

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[1] Fertilizer surface applied
[2] Fertilizer incorporated into top layer

Increases in NO_3-N amounts were found below the 45 cm depth in both plowed and no-till plots after the first rainfall. No significant difference was found in the amount of nitrogen in the top 45 cm of soil profile between the two tillage treatments after the 13 cm rainfall when N was surface applied.

After an additional 6 cm of rainfall on the same plots, total N increased below the 60 cm depth under both tillage systems. The nitrate concentrations at the 150 cm depth remained relatively constant and equal to the initial N concentration after 13 cm of rain on both tillage treatments.

Kanwar et al. (1987) conducted a field experiment to determine the effects of tillage and split N applications on nitrate leaching. The plots were on a Clarion-Webster silt loam soil in Iowa with a maximum slope of 2%.

Tile drains were installed at a depth of 1.2 m. Piezometers were installed at depths ranging from 1.2 to 3.6 meters for collection of ground water samples. Samples were taken April through October.

Six plots were established, two replications of two tillage systems, no-till and conventional continuous corn, each receiving 175 kg/ha nitrogen fertilizer at planting; and two plots of no-till corn receiving N fertilizer three times during the growing season (25 kg/ha at planting in early May, 50 kg/ha in June, and 50 kg/ha in July). Corn was harvested in October. Table 65 presents the nitrogen losses in subsurface drain flow for the three treatments.

Table 65
Average Growing Season Subsurface Drain Flows and NO_3-N Losses with Drain Water by Tillage and Fertilizer Management
Kanwar et al., 1987

Year	Precip.	Single Application 175 kg/ha·yr N				Split Fertilizer 125 kg/ha·yr	
		Conventional		No-till		No-till	
		Tile Flow	NO_3-N Loss	Tile Flow	NO_3-N Loss	Tile Flow	NO_3-N Loss
	cm	cm	kg/ha	cm	kg/ha	cm	kg/ha
1984	91.6	17.4	17.9	18.3	21.0	18.2	20.7
1985	64.3	0	0	0.1	0.1	0	0
1986	108.9	21.2	43.2	36.1	58.7	32.4	32.2
Ave.	88.3	12.9	20.4	18.2	26.6	16.9	17.6

In the first year of the study, when equal amounts of fertilizer were applied, there was no differnce in the N concentrations with tillage (both averaging around 11 mg/l NO_3-N). In the second year nitrogen loss to tile drains was higher with no-till than with conventional tillage (58.7 kg/ha, no-till to 43.2 kg/ha, conventional), even though the N concentration in tile flow from conventional plots were on the average 1.6 times those from no-till (conventional, 24 mg/l and no-till, 15 mg/l).

Fertilizer management (split and reduced application) did however reduce nitrogen losses. (There was no detectable difference in yield between the three treatments.) Table 66 lists the average NO_3-N concentration measurements over depth for the three year study.

Table 66
Average Growing Season NO_3-N Concentrations in Water Samples from Piezometers and Tile Drains
Kanwar et al., 1987

Year	Depth	Tillage and Fertilizer Applied No-till 175 kg N/ha	Conventional 175 kg N/ha	No-till 125 kg N/ha
	m		mg/l NO_3-N	
1984	1.2TD[1]	10.9	10.8	11.8
	1.2	12.4	10.8	
	1.8	10.1	6.9	
	2.4	10.9	16.8	
	3.0	11.4	9.3	
	3.6	8.1	11.5	
1985	1.2TD[1]	11.6	--	--
	1.2	--	--	
	1.8	12.1	--	
	2.4	14.1	10.1	
	3.0	9.5	11.5	
	3.6	6.8	4.7	
1986	1.2TD[1]	15.1	23.8	11.4
	1.2	17.7	17.9	
	1.8	16.5	12.9	
	2.4	14.3	16.1	
	3.0	8.3	15.1	
	3.6	3.7	5.3	
Ave.	1.2TD[1]	12.5	17.3	11.6
	1.2	15.1	14.4	
	1.8	12.9	9.9	
	2.4	13.1	14.3	
	3.0	9.7	16.9	
	3.6	6.2	7.2	

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[1] These 1.2m samples were from tile drains. All other entries from piezometer.

Tylor and Thomas (1977) found that leachate losses were higher under no-till than under conventional till corn, though the difference was great only during the month following fertilizer application (Table 67).

Table 67
Tillage Effects on Leachate Quantity and Nitrate-N Loads[1]
Tyler and Thomas, 1977

Month	Rainfall	Leachate		Nitrate-N Load		Chloride Load	
	cm/mo	No-till	Conven.	No-till	Conven.	No-till	Conven.
		cm/mo		kg/ha·mo		kg/ha·mo	
Nov72	9.0	0.0	0.0	0.3	0.3		
Dec72	14.4	6.6	8.4	0.2	0.2		
Jan73	3.0	1.2	2.2	0.8	1.0		
Feb73	6.0	0.3	1.2	0.4	2.0		
Mar73	11.7	1.0	2.2	1.0	1.8		
Apr73	13.4	3.0	3.6	3.9	3.1		
May73	15.0	3.9	3.9	2.6	4.4		
168 kg/ha ammonium nitrate and 396 kg/ha KCl fertilizer at planting, mid-May							
Jun73	14.4	4.8	1.2	14.7	4.0		
Jul73	10.8	0.6	0.6	1.2	0.5		
Aug73	4.2	0.0	0.0	0.0	0.0		
Sep73	4.0	0.0	0.0	0.0	0.0		
Oct73	3.6	0.0	0.0	0.0	0.0		
Nov73	15.8	0.9	0.7	2.7	0.3	4.0	2.0
Dec73	5.4	2.4	1.0	2.5	2.9	6.5	4.2
Jan74	15.3	6.9	7.8	8.6	11.0	30.0	19.7
Feb74	4.1	0.6	0.7	0.8	0.8	4.0	1.3
Mar74	10.0	5.4	6.0	13.5	9.7	44.0	17.0
Apr74	7.7	3.0	3.0	7.2	4.2	15.0	6.9
May74	15.9	2.0	1.4	8.1	3.4	9.2	6.8
168 kg/ha ammonium nitrate fertilizer at planting, mid-May							
Jun74	20.4	5.7	3.8	17.9	6.1	14.8	9.2
Jul74	7.2	0.3	0.3	0.1	0.1	0.2	0.2
Aug74	27.0	2.7	2.4	2.1	7.8	0.2	8.3
Sep74	7.5	2.4	2.4	1.4	6.6	0.2	4.2

The soil was a Maury silt loam (Typic Paleudalfs, clayey,

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[1] data from graphs; graph reading errors are approximately ± 0.3 and 3.0 kg/ha·mo for nitrate and chloride, respectively

mixed, mesic), a well-drained upland Kentucky soil formed in material weathered from phosphatic limestone. The soil shrank only moderately on drying so that any cracks formed were small (1-2 mm across). Prior to this study the site had been under bluegrass sod for about 50 years.

In this kind of study collection error can be significant.

Typically nitrate concentrations ranged from 5 to 17 ppm, except after heavy rains, which produced concentrations from 24 to 81 ppm. Unlike the results of other studies (e.g., Kanwar et al., 1987), differences in the amount of leachate collected from the two tillage treatments were slight, except for in June 1973, when 3.6 cm more leachate was collected from the no-till watershed than from the conventional till watershed.

The chloride tracer part of the study offers some insights into the movement of solutes through the soil. More chloride was collected under no-till plots than under conventional plots until August and September, a trend that did not become clear with nitrate until the second year of the study.

Under no-till 37% of the applied chloride (396 kg/ha KCl broadcast in May, 1973) was collected from May 1973 to September 1974. During that time 23% of the chloride applied to conventional till plots was recovered. The remainder is assumed to be stored in the soil profile. Chloride movement was not identical to water movement. The authors explained the discrepancies as resulting from water flowing through cracks and channels, bypassing chloride held by soil.

Angle et al. (1989) conducted a three year study of the nitrate nitrogen concentrations under soils planted to conventional and no-till corn. The soil was a Manor loam (coarse loamy, micaceous, mesic Typic Dystrochrept) in Maryland. Watersheds were instrumented with wells (10 m depth) and suction lysimeters (at 0.75 m and 1.5 m). Grassed "control" areas uphill of the tillage sites were also instrumented.

They analyzed ground water and percolate samples monthly for nitrate-N. Ammonium-N was also measured, but was not reported because the concentrations were frequently below detection limits.

Percolate from the grassed areas were consistently low in nitrate-N, averaging 2 mg/l.

Nitrate concentrations from the 0.75 m depth lysimeter exceeded those from the 1.5 m lysimeter.

Little significant difference was observed between the percolate samples for the two tillages except for during the first 5 months of 1987, the last year of the three-year study (Table 68).

Table 68
Monthly Nitrate Concentrations of Percolate and Ground Water Beneath Conventional and No-till Corn
Angle et al., 1989

mg/l Nitrate-N[1]

	0.75 m Lysimeter			1.5 m Lysimeter			10 m Well		
	Conven.	No-till	Grass	Conven.	No-till	Grass	Conven.	No-till	Grass
Apr84	10	8	1	7	4	1			
May84	11	8	1	7	7	1	17	8	
Jun84	10	10	1	8	8	1	17	7	
Jul84	14	14	1	10	7	1	17	7	
Aug84	39	14	1	10	10	1	16	7	
Sep84	19	11	1	10	8	1	10	7	
Oct84	18	10	3	11	8	3	9	7	
Nov84	18	10	4	11	8	3	13	7	
Dec84	12	14	3	11	8	3	13	7	
Jan85							10	7	
Feb85	3	36	3	8	9	1	7	10	
Mar85	12	14	4	10	10	1	10	8	
Apr85	16	24	5	17	11	2	4	4	
May85							6	6	
Jun85	52	0.5	0.4	18	0.4	1	8	1	1
Jul85	0.5	16	3	8	13	6	3	6	1
Aug85	0.5	39	2	1	11	5			
Sep85		33	2	8	7	2	13	9	1
Oct85	35	29	1	8	8	4	12	8	2
Nov85	22	25	1	8	8	1	9	7	2
Dec85	20	23	0.4	18	8	1	3	6	1
Jan86	13	28	1	8	9	1	4	8	1
Feb86	13	27	1	9	8	1	14	6	1
Mar86	15	18	1	10	15	1	10	8	2
Apr86	17	18	1	16	17	1	16	7	4
May86	25	22	1	15	13	1	9	12	3
Jun86	22	24	2	26	22	3	8	7	3
Jul86	15	8	2	18	9	2	5	7	3
Aug86	30	7	2	20	19	2	4	10	4
Sep86	3	22	1	10	14	2	4	8	6
Oct86	9	27	3	20	17	2	4	10	5
Nov86	7	24	2	19	18	2	3	6	5
Dec86	24	24	1	32	14	1	7	3	2
Jan87	5	13	0.4	16	6	1			
Feb87									
Mar87	5	19	1	28	19	1	9	7	6
Apr87	7	15	1	36	16	2	8	8	5
May87	10	19	1	40	14	2	9	6	5

---

[1] Read from graphs; reading error ≈ ± 1.5 mg/l.

During these months the percolate nitrate-N concentrations from a depth of 1.5 m were 15 and 30 mg/l for the no-till and conventional till watersheds, respectively.

The ground water nitrate concentrations under the tillage plots were consistently lower than the concentrations in the percolate. Under the grassed plot the ground water nitrate-N concentration averaged 2 mg/l. For most of the months reported, the ground water nitrate-N concentrations below the tillage plots exceeded that of the grassed control. The average ground water nitrate-N concentration of the two watersheds over the entire study was approximately 9 mg/l. The only significant difference in ground water nitrate-N concentration detected during the study occurred in 1984 (year 1), when the concentration under the conventional till watershed exceeded that under the no-till watershed.

One problem with the data in Table 68 is that taking monthly grab samples probably confuses the nitrate concentration short and long term responses (see Gerhart, 1986) that occur with storm events. If there were a tillage effect, Angle's sampling design would obscure it.

ii. NUTRIENTS IN TILED DRAINAGE

Drainage pipes are installed in fields which would otherwise be too wet to grow crops. The quality of the effluent from this drainage water will, of course, be greatly influenced by the nutrient input, soil characteristics, and plant uptake.

In studies reporting water quality parameters for both tiled drainage and lysimeter collections, there is often some discrepancy (e.g., Table 66, above). For this reason, papers reporting the chemical quality of tile drainage water are presented separately.

Instances where macropores directed soluble nutrients around tile drains into the shallow ground water have been reported (Bergström and Brink, 1986), implying that tile drainage waters do not accurately represent the waters leaving the root zone.

Please refer to the Waste Management chapter for information on drainage water quality from fields spray irrigated with manure slurry.

Kanwar et al. (1986) have summarized the results of various field experiments on the quality of subsurface drainage from cropland. They found that, on the average, 20 to 40 percent of the applied N fertilizers is being discharged through subsurface drainage waters.

In their ten year study, Schwab et al. (1980) analyzed drainage water below corn, oats, soybeans, alfalfa, and bare soil in rotation on a poorly drained, fine textured, Toledo silty clay, Mollic Haplaquept soil in Ohio (Table 69). Plots were drained at one of two depths. Dissolved nitrate-N and dissolved plus particulate phosphorus were measured.

Table 69
Average Dissolved N and Total P Delivery from Agricultural Drainage Water
Schwab et al., 1980

	Fertilizer Input kg/ha·yr	Rainwater Input kg/ha·yr	Drainage (1 m depth) kg/ha·yr	Drainage (0.5 m depth) kg/ha·yr	% Inputs Exiting with 1 m Drainage
NO_3-N	93	21.1	18.7	11.2	16%
P	89	4.6	1.2	0.8	1%

Ignoring the rainwater input would only make the subsurface nitrogen loss 20% of the applied fertilizer. These numbers are not strictly comparable to other studies on single crops.

Ellis et al. (1985) contrasted the losses of water, sediment, phosphorus, and nitrogen from test plots under conservation (chisel plow, 50-60% corn residue left) and conventional tillage (moldboard plow, all residue buried) in Michigan. The soils were fine textured, somewhat poorly drained Londo loam (an alfisol, classified as an Aeric Glossaqualf, fine loam, mixed, mesic soil) with slopes of about one percent.

The plots were tile drained, but the depth of the drains was not reported. Corn was grown in 1980 and 1981 and beans were grown in 1982.

Table 70 presents results of chemical analyses and flow monitoring over a two year period, April, 1981 to April, 1983. During this time there were 19 events, 16 resulting from rainfall and three from snowmelt. The table groups 18 of the events together and singles out an intense storm (6/20/82) falling on an emerging bean crop which resulted in most of the difference in sediment and phosphorus losses between the two tillage practices.

The authors reported an overall reduction in nutrient loads in runoff and subsurface drainage with chisel tillage. For what it's worth, 40 and 58% of the soluble P exiting conventional and chisel tillage left with the drainage water (assuming 100% collection efficiency); 30 and 48% of the total P; 85 and 82% of the nitrate; and 27 and 44% of the total Kjeldahl nitrogen.

Table 70
Two-Year Loads and Concentrations of Water Sediment, Phosphorus and Nitrogen from Conventional and Chisel Tillage on Tiled Fields
Ellis et al., 1985

Parameter	Period	Conventional Tillage		Chisel Tillage	
		Runoff	Tile Flow	Runoff	Tile Flow
Water	All 19 events	19.08	25.47	11.40	24.17
cm	18 events	18.54	24.41	11.40	23.31
	6/20/82	0.54	1.06	0.00	0.86
Sediment	All events	771	220	233	146
kg/ha	18 events	507	206	233	133
	6/20/82	264	14	0	7
mg/l	All events	404	61	204	51
	6/20/82	4890	94	-	59
Soluble P					
	estimated total[1]	0.300	0.200	0.170	0.230
kg/ha	18 events	0.268	0.190	0.170	0.220
	6/20/82	0.032	0.007	0.000	0.010
mg/l	All events	0.28	0.07	0.21	0.12
	6/20/82	0.59	0.05	-	0.09
Total P	All events	0.96	0.42	0.48	0.45
kg/ha	18 events	0.66	0.38	0.48	0.42
	6/20/82	0.30	0.04	0.00	0.03
mg/l	All events	0.50	0.12	0.42	0.17
	6/20/82	5.55	0.28	-	0.35
Nitrate-N	All events	5.96	35.0	4.8	21.7
kg/ha	18 events	5.89	33.8	4.8	20.9
	6/20/82	0.07	1.2	0.0	0.8
mg/l	All events	3.1	9.8	4.2	7.9
	6/20/82	1.29	7.9	-	8.1
TKN	All events	5.02	1.89	2.54	1.97
kg/ha	18 events	4.26	1.82	2.54	1.92
	6/20/82	0.76	0.07	0.00	0.05
mg/l	All events	2.63	0.53	2.23	0.72
	6/20/82	14.1	0.47	-	0.47

---

[1] estimated from approximately 65% of the events

d. NUTRIENT MANAGEMENT EFFECTS (FERTILIZER PLACEMENT, AMOUNT OF FERTILIZER APPLIED, TIMING OF FERTILIZER APPLICATION)

Kanwar et al. (1987, Table 65 above, Section V.C.2.c.i.) showed that cutting fertilizer N by 1/3 reduced nitrate leaving the root zone of corn by from 8 to 45%.

Whitaker et al. (1978) examined some of the nutrient management alternatives available within the context of conventional and no-tillage. These were

(1) fall fertilization at plowing versus spring fertilization by plowing or disking immediately before planting

(2) surface application of fertilizer versus banding at a depth of 10-13 cm

(3) fertilizing 5 to 6 weeks before planting versus fertilizing 25% at planting and 75% 4 to 6 weeks after planting by sidedressing, and

(4) applying various amounts of fertilizer (5 rates, from 15 to 364 kg/ha·yr)

Corn was planted parallel to slope on a slowly permeable Missouri clay pan soil (Mexico silt loam, Typic Udollic Albaqualf) with 3% slope. This soil has a clay subsoil beginning at a depth of 18-46 cm which restricts air and water movement and greatly retards plant root development during wet periods. Evaporation is a problem during draughty seasons.

The advantage of fall plowing in such a soil is not having to wait for the field to dry out in spring before plowing. This permits earlier planting of corn and increases the crops' ability to withstand late summer drought. The disadvantage is that it makes fields more vulnerable to erosion during the winter.

Table 71 shows the effect of time of fertilizer application on soluble nitrate and ammonium in runoff. The form of fertilizer applied was NH_4NO_3.

Soluble NO_3 and NH_4 concentrations were measured with ion sensitive electrodes, but other forms of dissolved N were not measured. Neither was ground water N loss monitored. The sampling protocol was not specified beyond "all runoff events were sampled", so the validity of the load estimates can not be evaluated.

Table 71
Effect of Fertilization Schedule on the Amount of Dissolved N in Runoff
Whitaker et al., 1978

Fertilization Schedule	Nitrogen Applications (kg/ha) Conventional Fall Plow	 Spring Plow	 No-till
Plowed down	121	121	--
Banded at planting	15	15	136
Banded at sidedressing	76	76	76
Annual total	212	212	212

Dissolved Nitrate and Ammonium in Runoff (kg/ha)	Fall Plow		Spring Plow		No-till	
Year of study	1st	2nd	1st	2nd	1st	2nd
Fall plow (11/7) to spring plow (4/2)	23.1	6.4	2.8	2.4	3.6	2.5
Spring plow (4/3) to planting (5/29)	4.1	1.8	2.9	7.5	0.6	0.6
Planting (5/30) to sidedress (6/25)	1.9	0.1	3.2	0.1	10.3	0.0
Sidedress (6/26) to fall plow (11/6)	0.0	4.5	0.0	13.2	0.0	14.2
Total	29.1	12.8	9.3	23.2	14.5	17.3
Annual Average	21.0		16.5		15.9	
Average corn yield (bu/ac)	106		103		97	

Table 72
Fertilizer Dose Effects on Annual Nitrogen Loss in Runoff and Sediment From No-till and Conventional Till Corn on a Clay Pan Soil
Whitaker et al., 1978[1]

N-fertilizer kg/ha·yr[4]	No-till runoff-N[2] kg/ha·yr	 TKN(sed)[3] kg/ha·yr	Conventional runoff-N kg/ha·yr	 TKN(sed) kg/ha·yr	Total N Ratio No-till to Conventional
14.6	7.8	4.5	7.8	25.8	0.4
97.5	20.2	2.2	13.4	24.7	0.6
219.75	17.9	2.2	13.4	21.3	0.6
211.8	17.9	3.4	14.6	30.3	0.5
249.9	22.4	3.4	39.2	19.1	0.4
364.3	33.6	2.2	34.7	21.3	0.6

---

[1] Data read from histogram
[2] dissolved nitrate and ammonium nitrogen
[3] particulate ammonium-N plus organic nitrogen
[4] precipitation supplied an additional 5 kg/ha·yr N
[5] considered to be optimum fertilization level

The nutrient uptake capacity of corn is low until 5 or 6 weeks after planting, so fertilizer applied before that time is more vulnerable to loss, especially, as is often the case in spring, when the soil is near field capacity.

The success of delayed plowing in controling dissolved N loss is largely determined by the weather, as evidenced by the variation in Table 72, and possibly by the first-year-of-tillage effect (the depletion of N in the plow layer of conventional till fields). Averaging the two years gives the impression of lowered dissolved nitrogen losses with spring tillage and no-tillage, but this conclusion, though true in the first year, was false in the second year of the study. In any case, the real issue with fall versus spring tillage is particulate not dissolved N loss.

In the fertilization rate study, over-fertilization results in similar dissolved N losses from no-till and conventional till corn. Only for the low fertilization rates did the dissolved losses fit the typical pattern of higher dissolved N from no-till relative to conventional till fields. Interestingly, the sediment bound N loads did not increase with fertilizer dose. As expected, the sediment bound TKN was about 8 to 10 times higher with conventional tillage than with no-till.

No-till plots delivered 40 to 60 percent of the total nitrogen leaving conventional plots in runoff. This was not particularly sensitive to the degree of over-fertilization.

Though the runoff-N category omits dissolved organic-N and the TKN(sed) category omits particulate nitrate-N, the authors generalized about the particulate/solid nitrogen split as related to tillages. They reported that, on the average, 13% of the nitrogen transported annually from the no-till plot was sediment bound, but 54% of the total N leaving the conventional plot was particulate.

Banding (reported for two fertilization rates, 212 and 364 kg/ha·yr) decreased the annual dissolved nitrate and ammonium loss by 50% at both rates. (Recall, from Table 71, 30% of the total fertilizer application of the broadcast plots was banded at sidedress.)

Timmons et al. (1973) studied N and P losses from a variety of conventional tillage fertilizer placement options. This paper is often incorrectly cited as support for the theory that the surface application of fertilizer in no-till is responsible for a springtime nutrient flush from no-till fields, since all of their plots were plowed.

Timmons et al. established experimental plots on a Barnes loam soil (Udic Haploborolls) in Minnesota with a 7% slope on which oats had been grown and harvested. They contrasted the following treatments by simulated rainfall tests:

1) Plot plowed and disked, no fertilizer applied

2) Fertilizer broadcast on oat stubble, plot plowed and disked

3) Plot plowed, fertilizer broadcast and disked in

4) Plot plowed, fertilizer broadcast on plowed surface

5) Plot plowed and disked, fertilizer broadcast on disked surface

For each of two years the fertilized plots received 168 kg/ha nitrogen as granular 33.5-0-0 and 39 kg/ha phosphorus as 0-46-0. All tillage operations were upslope.

Simulated rainfall was applied in two runs, separated by 24 hours. For each run 6.35 cm was applied at a rate of 6.35 cm/hr (equivalent to the 30 year storm) soon after cultivation. The antecedent soil moisture for the two runs was as follows:

depth	dry runs % moisture	wet runs % moisture
0-15 cm	16	27
15-30 cm	13	25

Table 73 indicates that there was not much difference in the losses from the five treatments (which would all be considered variants of conventional tillage).

Plowing down the fertilizer reduced the nutrient losses relative to surface broadcasting on top of plowed or disked surfaces. Disking down the fertilizer was inferior in nutrient retention to the deep placement achieved with inversion plowing.

The most striking feature of Table 73 is that for this simulated rainfall test immediately after fertilization with inorganic fertilizer, the bulk of the nitrogen loss is in the particulate organic form.

Baker and Laflen (1982) compared the dissolved nutrients in runoff from plots with no fertilization, injected fertilizer, and surface application. The runoff from plots with fertilizer injected to a depth of 5 cm was of the same quality, in terms of soluble nutrients, as runoff from the plots where no fertilizer had been applied.

Table 73
Losses of N and P in Surface Runoff as Affected by Broadcast Fertilizer Placement
Timmons et al., 1973

Treatment	Ammonium-N kg/ha test			Nitrate-N kg/ha test			Organic-N kg/ha test			Phosphorus[1] kg/ha test		
	Sediment	Dissolved	Total	Sediment	Dissolved	Total	Sediment	Dissolved	Total	Sediment	Dissolved	Total
1968--Low Antecedent Soil Moisture												
1	0.01a	<0.01a	0.01a	<0.01a	<0.01ab	0.01a	3.98a	0.01a	3.99a	0.01a	<0.01a	0.01a
2	0.01a	<0.01a	0.02a	<0.01a	0.01a	0.02a	3.92a	0.01a	3.93a	0.01a	<0.01a	0.02a
3	0.03b	0.06b	0.09b	<0.01a	0.01a	0.02b	4.24a	0.01a	4.29a	0.02b	<0.01a	0.03b
4	<0.01c	<0.01a	<0.01a	0.01b	<0.01b	<0.01c	0.02b	<0.01b	0.02b	<0.01c	<0.01a	<0.01c
p (%)	1	1	1	1	5	1	1	1	1	1	1	1
1968--High Antecedent Soil Moisture												
1	0.09	0.02a	0.11a	0.06ab	0.10	0.16a	36.76a	0.08a	36.84a	0.15ac	0.03	0.18a
2	0.11	0.02a	0.13a	0.08a	0.12	0.20a	38.63a	0.09a	38.72a	0.20a	0.03	0.24a
3	0.24	0.20b	0.44b	0.10a	0.16	0.26b	38.66a	0.13b	38.79a	0.27b	0.04	0.31b
4	0.06	0.16a	0.21a	0.02b	0.04	0.07c	6.32b	0.03c	6.35b	0.07c	0.04	0.11a
p (%)	NS	5	5	5	NS	1	1	5	1	5	NS	5
1969--Low Antecedent Moisture												
1	0.01a	<0.01a	0.02a	<0.01	0.02	0.03	4.49	0.03	4.53	0.02a	<0.01a	0.02a
2	0.02a	<0.01a	0.02a	<0.01	0.02	0.03	4.48	0.02	4.51	0.02a	<0.01a	0.02a
3	0.02a	0.02a	0.04a	<0.01	0.03	0.03	4.55	0.02	4.57	0.03a	0.01a	0.04a
5	0.03b	0.11b	0.15b	0.01	0.03	0.04	4.56	0.02	4.58	0.09b	0.08b	0.17b
p (%)	5	1	1	NS	NS	NS	NS	NS	NS	1	5	1

---

[1] Available P in sediment and total soluble P in water

VI. SOIL CHEMISTRY EFFECTS OF TILLAGE

A. RATES OF CHEMICAL TRANSFORMATIONS

1. NITROGEN TRANSFORMATIONS AND TILLAGE

For maximum crop yields, more N fertilizer is often recommended for no-till than for conventional till. One partial explanation for no-till's increased N requirement is that tillage influences the rates of nitrogen's chemical transformations, namely mineralization, immobilization, nitrification, and denitrification.

a. FERTILIZER NITROGEN IMMOBILIZATION AND TILLAGE

Kitur et al. (1984) examined the nitrogen balance under conventional and no-tillage. They fertilized plots (Maury fine-silty, mixed, mesic Typic Paleudalfs in Kentucky) growing corn and winter rye with ^{15}N-depleted ammonium nitrate for three years and partitioned the added N into crop recovery and soil immobilization categories.

Two levels of fertilization were used, N deficient (84 kg/ha) and high N (168 kg/ha) in order to highlight the N transformations that would be masked by abundant N availability.

After three years, 71-75% of the fertilizer was accounted for in the harvested grain, stover (corn stalks, excluding grain), or soil. Neither tillage nor fertilization rate had a large effect on the size of the lost fraction (denitrified or leached N).

Nitrogen fertilizer remaining in the no-till soil was 42% of that applied at the low fertilizer rate, and 39% of that applied at the high N rate. In the conventionally tilled plots the amounts remaining were 28 and 37% for the low and the high N rate, respectively.

Approximately one half of this immobilized soil N was located in the surface 5 cm of the no-till soil, though for conventional tillage it was more uniformly distributed in the top 30 cm. They concluded that this localization of the bulk of the organic soil N pool in the top 5 cm of no-till plots is the most likely explanation for the reduced crop recovery of fertilizer at low N fertilization rates.

The N balance from this study is outlined in Table 74, and Table 75 gives the N content of the soils by depth. Virtually all the recovered soil N was immobilized organic N (rather than ammonium or nitrate).

Table 74
Nitrogen Balance for No-Till and Conventional Till Corn
Kitur et al., 1984

	Total N applied (3 yr)	Fertilizer N in Grain 1980	1981	1982	(3 yr)	Stover N (1982)	Soil N	Lost N
	kg/ha[1]							
No-till	252	20a	19a	18a	57a	13a	107a	29a
Conven.	252	38b	33b	28b	100b	17a	70b	26a
No-till	504	49b	50c	46c	145c	36b	199c	25a
Conven.	504	43b	49c	49c	141c	37b	188c	27a

---

[1] Values within a column followed by the same letter are not significantly different at p = 0.05 by t-tests.

The measurements presented in the following table were made immediately after corn harvest on the third year of the study.

Table 75
Fertilizer N Recovered in Soil After 3 Years of a Tagged Fertilizer Study on No-till and Conventional Till Corn
Kitur et al., 1984

	Total N Applied (3 yr)	0-5cm	5-15cm	15-30cm	30-90cm	Total N Recovered
	N kg/ha[1]					
No-till	252	58	17	15	16	107 ± 8 a
Conven.	252	23	34	13	0	70 ± 11b
No-till	504	90	40	30	38	199 ± 38c
Conven.	504	35	72	54	28	189 ± 11c

---

[1] Values within a column followed by the same letter are not significantly different at p = 0.05 by t-tests.

b. NITRIFICATION RATES AND TILLAGE

Rice and Smith (1983) measured nitrification rates in the top 15 cm of Maury silt loam soil under long-term conventional and no-till corn cultivation in Kentucky. The nitrification rates (describing the oxidation of ammonium to nitrate) were dependent on soil moisture.

Two methods were used, one with mixed soil samples and the other with intact cores. The rates were roughly 3 to 4 times higher with the mixed samples (Table 76). With intact cores no-till mineralization rates were generally slightly higher than conventional rates. The opposite is true of mixed samples.

Their interpretation of the data was that nitrification rates of mineralized ammonium are limited primarily by ammonium supply and distribution. Ammonium limitations are more severe in no-till soils because of reduced mineralization rates and the absence of plow-mixing. After fertilization by large amounts of ammonium, other factors (moisture and pH) control nitrification rates.

The intact core results may be closer to reality than the mixed sample because mixing the samples results in an artificially raised redox potential, which would affect the metabolism of the organisms responsible for the ammonium consumption. It is also worth remembering that no-till soil moisture is usually higher than that of conventional till, and that field studies of nitrification would reflect this. Furthermore, the rates in Table 76 are expressed per gram of soil. If no-till soils have greater bulk density than conventionally tilled soils, then the nitrification rate of the plow layer will increase proportionately.

Table 76
Nitrification (Rate of Ammonium Consumption) and Tillage
Rice and Smith, 1983

Soil moisture g/100g soil	Mixed Sieved Samples No-till	Mixed Sieved Samples Conventional	Intact Cores No-till	Intact Cores Conventional
	μg of N per gram of soil per day			
34	-	-	3.6	4.2
31	15.7	11.5	4.4	3.7
26	14.1	15.3	4.0	2.7
22	-	-	3.2	2.6
20	10.1	13.4	-	-
18	-	-	2.9	2.7
15	8.1	11.5	-	-
14	-	-	0.8	1.7
10	3.0	5.4		

c. DENITRIFICATION RATES AND TILLAGE

Rice and Smith (1982) observed that denitrification was linked to moisture level, and concluded that the soils under no-till with a well developed mulch cover would be expected to lose more N by denitrification than they would under conventional tillage because of moisture retention by the mulch cover. Their experiment was not amenable to determining denitrification rates.

Colbourn (1985) studied in situ denitrification and other N losses from no-till and conventional till winter cereal in southern England on a mole-drained (50 cm depth) Drenchworth clay soil (Table 77).

Table 77
Denitrification Losses from Winter Grain: Four Year Study
Colbourn, 1985

Parameter	No-till mean	No-till standard error	Conventional
N loss, kg/ha per year	27	11	35
Denitrification N loss, kg/ha N[1] (summed October through June)	0	5	<5

	Monthly Denitrification N Losses, kg/ha mean	Monthly Denitrification N Losses, kg/ha standard error	Monthly Leached N Losses, kg/ha[1] mean	Monthly Leached N Losses, kg/ha[1] standard error
October	1.7	1.3	3.3	3.0
November	2.0	1.3	3.5	3.3
December	1.3	1.3	5.0	4.0
January	0.7	0.7	3.7	2.3
February	0.3	0.3	1.2	0.8
March	0.3	0.3	2.3	2.0
April	0.8	1.0	4.2	4.2
May	2.2	2.7	4.7	3.5
June	0.8	0.3	0.5	1.2

---

[1] read from a graph; composite of 4 years and both tillages

Fertilizer (20 to 30 kg/ha ammonium nitrate as N) was applied to the autumn seedbed and the crop was top dressed 3 times each spring (receiving from 109 to 223 kg/ha).

It was estimated that the crop took up about half of the N applied annually. A quarter was retained in the soil and the rest was lost to leaching or denitrification (to gaseous nitrogen or nitrous oxide).

Denitrification rates in no-till fields averaged 5-10 kg N per hectare per year with wide seasonal variation. Most of the N loss occurred in fall and spring. Plowing and drainage both diminished denitrification losses.

Parkin and Meisinger (1989) found a higher denitrification maximum potential (as indicated by enzyme activity) in the top 30 cm of no-till soil than in conventional till soil in a well drained Matapeake silt loam (fine-silty mixed, mesic Typic Hapludult) near Queenstown, Maryland. Little tillage-related difference in denitrification below the (unnamed) crop rooting zone was found. Their single sample was taken in spring before tillage, in order to catch the alleged denitrification maximum. The maximum rates were 2497 ± 1161 μg N/kg per day and 9633 ± 2933 μg N/kg per day for the top 30 cm of no-till and conventional till soil, respectively. Differences below that layer were not significant. Denitrification maximum rates dropped to less than 0.02 μg N/kg per day (0.3 kg N/ha per year) below 120 cm.

d. N MINERALIZATION AND TILLAGE

El-Haris et al. (1983) studied the effects of tillage and crop rotations on soil N mineralization potential on a Palouse silt loam (fine silty, mixed, mesic Pachic Ultic Haploxerolls) in Washington state. Fertilizer management plots were on a Ritzville silt loam (coarse-silty, mixed, mesic Calciorthidic Haploxerolls).

Experimental plots contrasted three tillage systems (moldboard, chisel, and no-till) and four crop rotations (continuous winter wheat (ww), winter wheat-pea (ww-pea), winter wheat-spring wheat (ww-sw), and green manure (3 years) and spring wheat-winter wheat (GM&SW-WW)). The crops were grown for 9 years before soil sampling for nitrogen.

The study also included the effects of increasing fertilizer application on soil N. Nitrogen fertilizer (NH_4NO_3 was applied in the fall prior to planting at 0, 34, 67, 135 and 270 kg of N per hectare (randomized complete block experimental design).

Soil cores (15 cm depth) were collected in mid-September and mid-March for the tillage experiment, and in mid-March for fertilizer experiment.

They found that conventional (moldboard) tillage of grains and legumes resulted in higher mineral N, lower organic C, and a lower C:N ratio in the first 15 cm relative to the no-till treatment, but did not affect total N (Table 78). No responses to chisel plowing treatments were detected. Crop rotation did not significantly influence soil total N, but organic C was higher for the GM&SW-WW rotation and the WW-Pea rotation.

Table 78
The Influence of Tillage and Crop Rotations on N and P in the top 15 cm of Palouse Silt Loam, Fall Sampling and the Influence of Fertilizer Application Rate on N and P in Ritzville Silt Loam
El-Harris et al., 1983

Treatment	Mineral N mg/kg	Total N mg/kg	Organic C g/kg	C:N
Tillage				
Moldboard	50	1680	14.0	8.3
Chisel	39	1610	14.2	8.8
No-till	29	1623	14.7	9.1
$LSD_{0.05}$[1]	14	124	0.6	
Crop Rotation				
Continuous winter wheat	49	1655	13.5	8.2
Winter wheat-pea	23	1487	12.9	8.7
Winter wheat-spring wheat	56	1621	14.5	8.9
GM&SW-WW	29	1789	16.2	9.1
$LSD_{0.05}$	14	374	0.5	
Nitrogen applied kg/ha				
0	8	581	4.3	7.4
34	12	592	4.6	7.7
67	15	629	4.7	7.5
135	20	630	4.5	7.1
270	51	674	4.9	7.3
$LSD_{0.05}$	17	79	1.0	

---

[1] Least significant difference at the 0.05 confidence level

Table 79
The Influence of Tillage and Season on the Nitrogen Mineralization Potential (N_0) and First Order Decay Rate (k)
El-Haris et al., 1983

Treatment	Fall sampling		Spring sampling	
	N_0 mg/kg	k day^{-1}	N_0 mg/kg	k day^{-1}
Moldboard	136	0.013	68	0.048
Chisel	131	0.013	74	0.049
No-till	125	0.015	73	0.054
$LSD_{0.05}$[1]	33	0.003	4	0.009

---

[1] Least significant difference at the 0.05 confidence level

Soils from crop rotations receiving the most fertilizer N (continuous winter wheat and winter wheat - spring wheat) were highest in mineral N. Those receiving the least fertilizer N (winter wheat - pea and green manure and spring wheat - winter wheat rotation) were lowest in mineral N, which reflects differences in residual mineral N from the previous crop.

The average nitrogen mineralization potential, N_0, (assuming first order decay) in the top 15 cm was not affected by tillage treatment in the fall (Table 79). However, in the spring N_0 for the moldboard-plowed treatment was significantly lower than for both the other tillages.

Compare these spring mineral pool figures to those of Blevins et al. (1983) in Table 87, below. Blevins reports 24 mg/kg for no-till and 16 mg/kg for conventional tillage on a Maury silt loam.

N_0 in the first 0 to 15 cm of soil was also found to vary with depth on a finer scale (Table 80).

Table 80
The Effect of Tillage and Sampling Time on Estimated N_0 with Respect to Soil Depth
El-Haris et al., 1983

Nitrogen Mineralization Potential, N_0 in mg/kg[1]

	Fall sampling			Spring sampling		
Depth, cm	0-5	5-10	10-15	0-5	5-10	10-15
Moldboard	153	126	129	71	65	68
Chisel	165	126	97	96	71	59
No-till	185	107	79	109	59	53

---

[1] read from a graph

Cumulative mineralized N in the top 5 cm of soil averaged over the fall and spring samples was significantly higher for no-till than for moldboard plow, but not for chisel plow (Table 81). The greater mineralized N with no-till was attributed to increased organic matter and microbial activity near the surface when little fertilizer incorporation takes place.

Rotations did not affect the average N_0 of the top 15 cm in the fall sampling, though in the spring N_0 for GM&SW-WW was higher than N_0 for winter wheat. No other differences emerged (Table 82). The k for continuous WW was significantly lower in the fall than for both the GM&SW-ww and WW-SW rotations, but there were no significant differences among the k values for the crop rotations in the spring sampling.

Table 81
The Effect of Tillage on Cumulative Nitrogen Mineralization in the Top 5 cm of a Palouse Silt Loam
El-Haris et al., 1983

	Nitrogen Mineralized, mg N per kg soil[1]		
Week	No-till	Chisel	Moldboard
2	41	39	28
4	70	59	47
6	77	69	53
8	88	78	63
10	98	89	69
12	109	100	78

---

[1] read from a graph

Table 82
The Influence of Crop Rotations and Time of Sampling on the Depth Averaged N_0 and k Values for Palouse Silt Loam
El-Haris et al., 1983

Rotation	Fall sampling		Spring sampling	
	N_0 mg/kg	k day^{-1}	N_0 mg/kg	k day^{-1}
Continuous WW	153	0.010	68	0.049
WW-pea	140	0.013	69	0.050
WW-SW	113	0.015	73	0.045
GM&SW-WW	118	0.016	77	0.052
$LSD_{0.05}$[1]	47	0.004	8	0.013

---

[1] Least significant difference at the 0.05 confidence level

Although no-till methods may alter the pathway of nitrogen movement from agricultural systems, their implementation probably has little bearing on the total amount of nitrogen lost from these systems. No-till methods could prove valuable in reducing nitrogen losses in situations where enhanced water retention increases crop uptake efficiencies of applied nitrogen.

B. ACIDITY AND ACID SOLUBLE CATIONS

In no-tillage the soil remains largely undisturbed for several years and fertilizers are surface applied. Nutrients and organic matter accumulate at the soil surface.

Blevins et al. (1983a) examined cation abundance in Maury silt loam (Kentucky) after ten years of continuous corn and winter rye under conventional and no-tillage. Their results are summarized in Table 83.

Table 83
Major Cation Concentration in Maury Silt Loam After Ten Years of Conventional and No-Tillage Corn Production at Two Nitrogen Fertilization Rates
Blevins et al., 1983a

Depth cm	Fertilizer Nitrogen	pH in water		Calcium meq/100g		Magnesium meq/100g		Potassium meq/100g		Aluminum meq/100g		Manganese meq/100g	
	kg/ha	No-till	Conv.	No-till	Conv.	No-till	Conv.	No-till	Conv.	No-till	Conv.	No-till	Conv.
Unlimed													
0-5	0	5.8	6.4	5.48	7.84	0.33	0.70	0.68	0.39	0.06	0.03	0.62	0.01
0-5	168	4.8	5.8	2.61	6.13	0.47	0.59	0.44	0.34	1.22	0.04	0.10	0.02
5-15	0	6.0	6.4	5.96	7.48	0.62	0.72	0.34	0.43	0.05	0.04	0.01	0.01
5-15	168	5.6	5.9	4.74	6.19	0.55	0.55	0.34	0.37	0.26	0.05	0.04	0.01
15-30	0	6.2	6.6	6.61	7.49	0.58	0.71	0.21	0.29	0.05	0.01	0.01	0.00
15-30	168	6.2	6.2	6.10	6.91	0.54	0.63	0.20	0.21	0.06	0.05	0.01	0.01
Limed													
0-5	0	7.3	6.9	9.95	8.49	0.72	0.67	0.69	0.38	0.03	0.01	0.00	0.00
0-5	168	7.0	6.6	9.77	8.03	0.76	0.63	0.61	0.34	0.03	0.00	0.00	0.01
5-15	0	6.9	7.0	7.80	8.17	0.62	0.68	0.38	0.40	0.05	0.01	0.00	0.00
5-15	168	6.6	6.6	7.49	7.97	0.57	0.64	0.32	0.36	0.04	0.01	0.01	0.01
15-30	0	6.6	7.0	6.85	8.33	0.56	0.69	0.18	0.25	0.05	0.03	0.00	0.00
15-30	168	6.6	6.8	7.14	8.13	0.58	0.69	0.20	0.23	0.06	0.01	0.00	0.00

Nitrogen fertilizer application in particular is linked with the induction of acid conditions especially on no-till fields (Table 84). Phosphorus fertilizer was not added because the Maury silt loams have abundant phosphorus from their limestone parent material.

As soil pH declined, a decrease in exchangeable calcium and an increase in exchangeable aluminum and manganese occurred, accompanied by a reduction in yield (relative to limed plots at the same fertilization level). The difference in yield between limed and unlimed plots was 0.5 t/ha less for the unlimed conventional tillage plots and 2 t/ha less for the unlimed no-till plots.

Also acidic surface soil (pH less than 5.5) results in rapid deactivation of the triazine herbicides commonly used in no-till corn production. Reduced weed control was evident on the more acid plots in the Blevins study.

Table 84
Soil pH at Three Depths After Ten Years of Corn Production by No-till and Conventional Tillage as Related to N Fertilization
Blevins et al., 1983a,b

Depth	0 - 5 cm				5 - 15 cm				15 - 30 cm			
Fertilizer kg/ha N (Unlimed soil)	0	84	168	336	0	84	168	336	0	84	168	336
					pH							
Conventional	6.4	6.4	5.9	5.6	6.4	6.4	5.9	5.4	6.5	6.4	6.1	5.8
No-till	5.8	5.2	4.8	4.4	6.0	6.3	5.6	4.9	6.2	6.1	6.2	5.7

The work of Baldwin et al. (1985) supports the contention that surface fertilizer broadcasting is responsible for the pH differences attributed to tillage. They found little difference in runoff pH from no-till, chisel till, and conventional till plots to which fertilizer had been surface applied to all plots <u>after</u> tillage (Table 85). (Had fertilizer been applied <u>before</u> tillage it would have been buried during conventional tillage.) The soil was a well buffered Maury silt loam in Kentucky in the first year of a tillage study. Before the study, the plots had been in bluegrass sod.

Table 85
Acidity of Runoff as Affected by Tillage
Baldwin et al., 1985

Date (1984)	6/27	7/4	7/11	7/26	8/1	10/28	11/1	11/4	AVE
				pH					
Conventional	7.16	5.78	6.17	6.48	6.17	6.12	6.16	5.86	6.24
Chisel plow	7.19	6.57	6.42	6.69	6.40	6.18	6.74	6.30	6.56
No-till	7.30	6.19	7.25	6.37	6.23	5.86	6.35	5.45	6.38

C. ORGANIC MATTER

Blevins et al. (1983a) also reported the average (over all fertilization rates) percent organic matter with depth after their ten year experiment (Table 86). See the above section (Acidity) for a description of the experiment.

Table 86
Soil Organic Matter Distribution Over Depth For Continuous No-till and Conventional Till Corn
Blevins et al., 1983a

Soil Depth cm	No-till	Conventional Till
	percent organic matter[1]	
0-5	4.6	2.3
5-15	2.3	2.1
15-30	1.0	1.2

---

[1] From a graph

A greater amount of N is immobilized into organic forms under no-tillage, reducing nitrogen availability (Tables 87 and 88) and influencing other aspects of the nitrogen budget.

Table 87
Soil Organic Nitrogen in 0-5 cm of Limed Maury Silt Loam After Five and Ten Years of
Continuous Corn Production Under Conventional and No-tillage
Blevins et al., 1983a

Fertilizer	No-till		Conventional	
kg/ha	1975	1980	1975	1980
	percent organic nitrogen			
0	0.21	0.23	0.15	0.14
84	0.23	0.25	0.15	0.15
168	0.23	0.24	0.17	0.15
336	0.24	0.30	0.16	0.15
average	0.23	0.26	0.16	0.15

Table 88
Organic Carbon and Nitrogen as a Function of Tillage and Depth After Ten Years of Continuous Corn Production Under Conventional and No-Tillage
Blevins et al., 1983b

Depth cm	Fertilizer kg N / ha	Unlimed Plots Organic Carbon No-till	Unlimed Plots Organic Carbon Conv.	Unlimed Plots Organic Nitrogen No-till	Unlimed Plots Organic Nitrogen Conv.	Limed Plots Organic Carbon No-till	Limed Plots Organic Carbon Conv.	Limed Plots Organic Nitrogen No-till	Limed Plots Organic Nitrogen Conv.
		%, w/w		%, w/w		%, w/w		%, w/w	
0-5	0	2.15	1.25	0.21	0.14	2.24	1.38	0.23	0.14
	84	2.95	1.40	0.26	0.15	2.77	1.29	0.25	0.15
	168	2.80	1.39	0.26	0.16	2.48	1.38	0.24	0.15
	336	2.93	1.46	0.23	0.15	3.21	1.37	0.30	0.15
5-15	0	1.09	1.38	0.15	0.15	1.23	1.30	0.15	0.14
	84	1.28	1.34	0.17	0.15	1.30	1.28	0.15	0.14
	168	1.36	1.34	0.15	0.15	1.25	1.38	0.15	0.16
	336	1.15	1.49	0.15	0.16	1.46	1.46	0.16	0.16
15-30	0	0.57	0.78	0.10	0.11	0.57	0.65	0.10	0.10
	84	0.94	1.01	0.11	0.12	0.70	0.71	0.10	0.10
	168	0.66	0.70	0.10	0.10	0.72	0.76	0.10	0.10
	336	0.90	0.94	0.12	0.12	0.82	0.89	0.11	0.12

Table 89
Total Organic Carbon and Kjeldahl Nitrogen Content Under Two Tillages
Doran, 1980

Depth,cm	Total Organic Carbon 0-7.5	Total Organic Carbon 7.5-15	Total Organic Carbon 15-30	Kjeldahl Nitrogen 0-7.5	Kjeldahl Nitrogen 7.5-15	Kjeldahl Nitrogen 15-30	Soil Water Content 0-7.5	Soil Water Content 7.5-15	Soil Water Content 15-30	Mineralizable N[5] 0-7.5	Mineralizable N[5] 7.5-15	Mineralizable N[5] 15-30
				No-till to Conventional Till Ratio								
Corn												
Site 1	1.55^{4}	0.81^{3}	0.90^{1}	1.48^{3}	0.95	0.94	1.27^{1}	1.80	1.03	1.52^{4}	0.94	0.86
Site 2	1.07^{1}	1.00	0.89	1.13^{2}	1.02	0.85^{2}	1.01	0.85^{3}	0.87^{2}	1.45^{4}	1.00	0.75^{2}
Site 3	1.20^{2}	0.97	1.02	1.08	1.02	0.95	0.99	0.98	0.97	1.13	0.95	0.92
Site 4	1.10^{1}	1.05	0.97	1.00	0.99	0.96	0.86^{1}	0.82^{1}	0.90	1.19^{2}	0.93	0.97
Wheat												
Site 5	1.50^{2}	1.00	0.98	1.45^{3}	0.98	1.01	1.22^{2}	0.97	0.89	1.58^{2}	0.97	1.07
Site 6	1.25	0.98	0.93	1.12^{1}	1.07^{1}	0.92^{2}	1.08	1.10	1.07	1.34^{3}	1.12	0.97
Site 7	1.10	0.88^{3}	0.82^{2}	1.12	1.01	0.95	3.83^{3}	1.05	1.01	1.25^{2}	0.81^{2}	0.83
Average	1.25^{4}	0.96^{2}	0.93^{3}	1.20^{4}	1.01	0.94^{4}	1.47^{3}	0.98	0.96	1.35^{4}	0.96	0.91^{3}

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[1] $p < 0.10$ [2] $p < 0.05$ [3] $p < 0.01$ [4] $p < 0.001$

[5] potentially mineralizable nitrogen, estimated by the amount of ammonium-N released during 16 hr of autoclaving in 0.01 M $CaCl_2$

Doran (1980) gathered soil cores from several long term tillage experiments and compared organic matter content over depth (Table 88). He reported no-till:conventional-till ratios for various parameters. From Blevins' work, above, (Doran's Site 1), it is clear that the ratios would have been different had Doran chosen to isolate the top 5 cm rather than the top 7.5 cm.

A brief site description follows:

Site	Soil	Years of continuous crop
1 Lexington, Kentucky	Maury silt loam	9 years corn
2 Waseca, Minnesota	Nicollet clay loam	9 years corn
3 Morgantown, WV	Wharton-Cookport silt loam	5 years corn
4 Lincoln, Nebraska	Crete-Butler silty clay loam	3 years corn
5 Sidney, Nebraska	Duroc loam	9 years wheat
6 Sidney, Nebraska	Alliance silt loam	10 yrs wheat-fallow
7 Pendleton, Oregon	Walla Walla silt loam	9 yrs wheat-fallow

The organic C and Kjeldahl N contents of 0-7.5 cm no-till soil averaged 1.25 and 1.20 times higher, respectively, than conventionally tilled soil. On the average, the organic content of the 7.5-15 cm layer was the same for both tillage treatments. Below the 15 cm depth the organic C and N contents of the no-till soils were roughly 6 or 7 percent lower than those of conventionally tilled soils.

Similar results were reported by Dick (1983) for an 18 year study. He found elevated organic matter concentrations in the top 7.5 cm of no-till plots, but little difference deeper in the soil profile. Organic matter concentrations were similar for minimum and no-till plots.

VII. CHAPTER 1 CONCLUSIONS

In the introduction various issues were presented. The organization of this section is reiteration of the issues with discussion of the following questions

- Are the claims from the introduction supported by the literature review?
- Do the properties vary over time? What are the time variant parameters?
- What are the reported parameter ranges?
- Are there other important issues or processes to consider?

The first four claims are from the Maryland State Soil Conservation Committee. The rest are from Donigian et al., 1983.

Claims

1. The amounts of dissolved P leaving a field in runoff and leachate are small relative to the amounts adsorbed on eroded sediments.

Alberts and Spomer (1985) reported that the quantity of leachate P from conventional and till-plant corn was about the same, namely one tenth of a percent of the amount of fertilizer P applied annually. Three tenths of a percent and 0.7% of the annually applied P fertilizer was lost as dissolved phosphorus in runoff from conventional and till-plant corn, respectively. Unfortunately, particulate P in runoff was not reported in this study, so partitioning the remaining P into biomass, soil P pool, and particulate P in runoff can only be speculative. However, in view of the relationships between dissolved and particulate P loads in runoff reported in other studies (below), Claim 1 seems to be generally true.

Schwab et al. (1980) analyzed drainage water below corn, oats, soybeans, alfalfa, and bare soil in rotation on a poorly drained, fine textured, Toledo silty clay, Mollic Haplaquept soil. They reported an average annual 0.3% loss of dissolved phosphorus to leachate, 1.6% loss to particulate P in runoff, and 0.5% loss to dissolved P in runoff.

Dissolved P losses in runoff, when expressed as percent of total P loss in runoff tend to be low for conventional tillage during the growing season (Hubbard et al., 1982). From May through September 80-98% of the runoff total phosphorus loss is particulate. More intense storms move higher proportions of particulate P. In the winter (November through April) particulate

P losses in runoff from conventionally tilled fields can drop to 40-70% of the total runoff P loss. Snowmelt and the accompanying increase in interflow are important factors in this change.

The proportion of particulate P in runoff from no-till fields is less than that from conventionally tilled fields, generally 30-80% of the total P in runoff is particulate, with different studies reporting widely different dissolved-particulate splits. The time of highest dissolved P loss is immediately after fertilization.

2. Runoff and erosion reduction reduce phosphorus loss.

This is undisputed. If 3% of the fertilizer P applied annually to conventionally tilled corn fields is lost as particulate P in runoff (a back-calculation using Alberts and Spomers figures and an assumed 90% particulate P loss in runoff), runoff and erosion reduction can theoretically reduce phosphorus loss up to this limit.

3. Nitrogen is lost primarily through downward movement in the soluble nitrate form.

Walter et al. (1979) generalize that the subsurface losses of nitrate and other substances with low affinity for sediment binding are at least five times the surface losses in humid regions.

Kanwar et al. (1987) cite studies indicating that on the average, 20 to 40% of the applied N fertilizers are being discharged to the surface water supplies through subsurface drainage waters.

Alberts and Spomer (1985) reported that 16 percent of the nitrogen applied as fertilizer appeared as ground water nitrate under till-plant corn, and 5 percent leached into ground water below conventional till corn. One percent of the applied N appeared as runoff nitrate (for both tillage treatments).

Ground water ammonium accounted for 0.1% of the applied N (both tillages), and ammonium in runoff was about 0.2% of the fertilizer N applied. Particulate and organic nitrogen species were not measured in this study.

House et al. (1984) found 14 and 15% of the fertilizer N applied to sorghum lost annually in leachate from no-till and conventional till plots. For soybeans, 37 and 46% of fertilizer N was leached from no-till and conventional till plots, respectively. In this study runoff losses were not reported.

(Though I don't recommend that anyone else combine the results of unrelated studies in this manner, for the sake of argument) using Alberts and Spomers' figures for nitrate losses and assuming non-nitrate runoff N losses to be 44% of total nitrogen runoff losses (Leitman et al., 1983), and non-nitrate N leachate losses to be zero (which is probably not true), then 3 times more N is lost in leachate from conventional tillage than is lost in runoff, and 8 times more is lost with no-till leachate.

In their ten year multi-crop study, Schwab et al. (1980) reported that 20% of the annually applied fertilizer was lost as leachate nitrate and 9% as runoff nitrate.

4. Other forms of nitrogen loss are negligible relative to dissolved nitrate subsurface losses.

This is not clear (because nobody measures organic N in leachate), but probably true.

5. Interception is increased by no-till, minimum-till, spring plow, and winter cover.

As interception storage is a function of cover, certain concepts should be kept in mind.

The times that percent cover varies with tillage is from harvest through seedbed. After plant establishment the developing canopy provides most of the surface cover so percent cover can be considered roughly the same and equal to canopy cover for the different tillages, as long as plant development is synchronous.

After planting, the percent cover for no-till corn or soybeans is about 60% (reported range 45-76%). Only when no-tillage is performed on killed sod does percent cover approach 100.

Harvesting type (taking grain only or grain plus stalks for silage) affects cover. Harvesting no-till crops for silage defeats one of the purposes of that tillage, but is practiced by some dairy farmers. If modeling a crop harvested for silage, reduce the cover figure by 70% (e.g.% cover for no-till corn for silage would be 20%).

There are quite a few different reduced tillages, each leaving a distinctive surface signature. Reports of percent cover after planting range from 8 to 85%, but average about 30%. Of course, this is reduced if the crop is harvested for silage.

After planting by conventional tillage about 2% cover remains (0 to 5%).

6. Depression storage is increased by contouring and contour strips, chemical till-in, and maybe (0,+) by no-till, minimum-till, and spring plowing.

Agricultural soils have varying degrees of micro-relief surface storage. This storage is dependent on the recent agronomic history of the soil surface, which is modified by the action of rain, wind, tillage and cultivation practices.

Both random roughness and maximum storage are decreased significantly by the action of rainfall. On the average, the maximum storage was increased 3.5 times by plowing. Typically, rainfall decreased storage on plowed surfaces by 45%, resulting in storages roughly twice the volume of the original (before rain) unplowed plots. Unplowed plots also lose a substantial part (40%) of their surface storage during rainfall (Moore and Larson, 1979).

7. Soil moisture storage is may be increased (0,+) by no-till and minimum-till.

Conventional tillage dries out the soil profile, relative to no-tillage. House et al. (1984) reported significantly higher (usually around 2 percentage points higher) moisture content in the first 20 cm of no-till soybeans in January, June, and September. There was no tillage related difference in moisture content at 40-50 cm.

Tollner et al. (1984) found that, despite the greater moisture content in a no-till sandy loam soil, there was actually less "plant available water", operationally defined as the difference between the residual moisture content and the water content at 300 cm suction. (The residual moisture content is the value at which moisture content ceases to change as suction increases). The entity, plant available water, gives an indication of how well different soils would supply water to plants under <u>dry</u> conditions with all other factors (such as residue and canopy cover) being equal.

Soil in the no-till plots had about 0.07 cm^3/cm^3 less plant available water near the surface than did the soil in the conventional tillage plots. This does not conflict with the observation that the no-till soil was generally wetter than the conventional till soil, because soil moisture is a transient condition of the soil where plant available water is a permanent characteristic of the soil.

The no-till profile (0-60 cm) contained 15 percent more water in May and June and 30 percent more water in July than did conventional plots. There was no difference in water content in August.

8. Overland flow is decreased by no-till, minimum-till, contouring, contour strips, spring plowing, winter cover, and chemical till-in.

Angle et al. (1983) reported significantly (five times) more runoff coming from contour conventional till corn than from contour no-till corn watersheds. In their study, frequently during the summer months, events were recorded where large amounts of runoff were collected from the conventional till watershed while no runoff was collected from the no-till watershed.

Only four events were recorded over the entire three year study where runoff occurred from the no-till watershed but not from the conventional till watershed. Three of these events occurred in the early spring.

No-tillage can result in increased runoff during the seedbed stage, relative to conventional tillage (Lindstrom and Onstad, 1982), though at other times runoff is reduced by no-tillage (McGregor and Greer, 1982). The difference in runoff volume is not always dramatic or even evident, and is dependent on a number of factors in addition to tillage.

Runoff was significantly greater for no-till plots relative to conventional and chisel sites immediately after planting, though the differences between no-till and conventional till were obscured in the other crop stages (Mueller et al., 1984a).

Consistently lower runoff volumes came from chisel sites during crop stage 3, but for seedbed and crop stage 1 the runoff volumes from the chisel sites resembled those from the conventional sites.

The amount of overland flow seems to be highly correlated to antecedent soil moisture (Lindstrom and Onstad, 1984). In the week after planting, the increased runoff observed from no-till fields seems to be explained by the higher soil moisture content of their soils. When, during seedbed, the soils under conventional or chisel tillage systems become saturated, they produce the same amount of overland flow as do no-till plots.

The importance of soil drainage characteristics in determining tillage related hydrologic response was documented by Edwards and Amerman (1984). They contrasted a well drained soil (Rayne silt loam Typic Hapludult, medium grained resistant sandstone parent material) with a soil having limiting internal drainage (Keene silt loam Typic Aquic Hapludalf, silty shale parent material).

They found that during the summer months no-tillage on either soil increased cumulative infiltration relative to conventional tillage. Surface protection from residue was sufficient to reduce runoff during the winter dormant season on the well-drained soil, but on the poorly drained field the soil became saturated above the subsurface flow-restricting horizons, regardless of the surface cover conditions, producing increased runoff.

McGregor and Greer (1982) measured runoff from no-till, reduced till, and conventional till corn plots grown on thin, highly erodible loess soils. Reduced tillage consisted of no-till planting with two cultivations replacing herbicide application for weed control.

Runoff was significantly reduced by conservation tillage, though strict no-till was not nearly as effective as no-till plus cultivations. The 3 year average runoff from no-till corn harvested for grain was 12% less than that from conventional till plots, and the runoff from reduced till plots was 45% less (probably indicating surface seal development on the no-till plots).

9. Infiltration is increased by no-till, minimum-till, contour strips, and chemical till-in, and maybe increased (+,0 and +,? in Table 3) by spring plowing and winter cover.

Plowing often, but not always, increases the infiltration rate of the plow layer immediately after planting. Lindstrom et al. (1984) showed no significant effect of tillage (conventional, ridge, and chisel) on the infiltration rate of two clay loams and a silty clay loam after planting.

Soil texture appears to be the significant variable here. They were able to demonstrate an elevated infiltration rate with conventional tillage on an unsaturated loam (also after planting), though the tillage effect ceased when the Ap horizon was saturated. Chisel tillage did not reduce infiltration (relative to conventional tillage) under any moisture or residue condition.

Tillage related differences in infiltration may be short lived. Meyer and Mannering (1961) showed that surface crusts developed rapidly on conventionally tilled and minimum tilled soil surfaces which significantly reduced infiltration amounts. They reported that infiltration rates on minimum till corn plots on a podzolic soil with a tendency to crust (Russell silt loam, Indiana) were reduced by surface crusting 21 to 36% (averaging 27%), depending on antecedent moisture by surface crusting.

Many soils will form a dense surface crust upon exposure to relatively low rain intensities. Such soils often have high silt and low organic matter and calcium carbonate. During rain, silt and clay segregate, forming a surface seal. McIntyre (1958) determined saturated hydraulic conductivity of a crust formed by and Australian fine sandy loam as 5×10^{-7} cm/sec, where the conductivity of the soil below the crust was 10^{-3}cm/sec.

Ehlers (1975) observed that surface crusts did not form on no-till plots on podzolic soils (Typudalf) derived from loess, but were thick on tilled plots on the same soils.

However, increased of runoff due to surface crusting has been observed with row crops under no-tillage on a well drained soil, though to a much lesser extent than on conventional till plots (Edwards, 1982).

On soils with poor internal drainage the effects of crusting are overwhelmed by restrictions to infiltration from the subsoil (Edwards and Amerman, 1984).

With no surface crust present, at low rainfall intensity, the two main factors affecting the infiltration rate are antecedent soil moisture and soil bulk density. Soil bulk density is changed by various tillage practices through cultivation or artificial compaction.

At higher rainfall intensities, the role of macropores becomes important, resulting in significantly greater infiltration rates for no-till than for conventional tillage. These rates are higher than what would be accounted for by capillary water flow theory.

The large pore flow occurs only during or soon after heavy rainfall, and it seems to bypass water (and chemicals) held in the small pore soil matrix. Though macropores are obliterated by plowing, they redevelop as the crop grows.

Ehlers estimated the maximum additional infiltration rate mediated by earthworm channels in no-tilled podzolic soil to be 1.2 mm/min (7.2 cm/hr), compared to 0.6 mm/min (3.6 cm/hr) for tilled soil (infiltration in excess of that controlled by small scale porosity).

The overall effect of these factors is that during most of the year, net infiltration is greater under no-tillage, but during seedbed, for low intensity storms, the roughened plow layer of conventional tillage may admit more water than does that of no-till (Shinde et al., 1983).

10. Subsurface flow is increased by no-till, minimum-till, spring plowing, and winter cover, and maybe (0,+) increased by contouring and contour strips. The effect of chemical till-in on subsurface flow in unknown (?).

Tollner et al. (1984) applied potassium chloride to the surface of their experimental no-till and conventional tillage plots (double cropped soybean/small grain on Cecil sandy loam in Georgia), then sampled the profiles in February and July, 1982 and January, 1983. The chloride leaching front under no-till was usually deeper than that under conventional tillage, indicating a higher water throughput in the no-till profiles.

This was consistent with previous work on well-structured soils (McMahon and Thomas, 1976; and Tyler and Thomas, 1977 (on Maury silt loam and Lowell silty clay loam)) where larger amounts of nitrate were found to move more deeply in the soil in no-till relative to conventional till plots.

Alberts and Spomer (1985) demonstrated increased subsurface water loss from conservation tillage. Their test sites were in the deep loess hills of western Iowa. Three watersheds were instrumented: (1) contour conventionally tilled corn, (2) contour till-plant corn, and (3) till-plant corn on terraces with underground pipe drainage.

Conservation tillage increased percolation below the root zone, though this effect did not manifest until the fourth year of the ten year study. The annual (10-year average) subsurface flow from the plant-till and pasture watershed was 18.6 cm compared to 13.6 cm for conventional tilled corn (a 28% overall difference, a 40% difference if only considering the last 7 years of the study, when the tillage effect was detected).

A partial explanatory mechanism is the macropore development associated with no-tillage. After 4 years, no-till soils showed twice the number of large surface-connecting macropores as did tilled soils (Ehlers, 1975).

The B_t horizon (45-110 cm), had the highest density of channels in both tilled and untilled plots, but it is the Ap horizon which is critical. Small, blocked, or horizontal pores (typical of a conventionally tilled Ap horizon) do little to conduct water downward; whereas large diameter (8-11 mm), vertical macropores connected to the surface, responsible for much of the downward movement of water, are more typical of no-till plots.

Unlike earthworm channels in the Ap horizon of the tilled plot, almost all of the channels reaching the untilled soil surface transmitted water deeply into the profile (Ehlers, 1975).

Tyler and Thomas (1977) found that leachate losses were higher under no-till than under conventional till corn, though the difference was great only during the month following fertilizer application. Typically nitrate concentrations ranged from 5 to 17 ppm, except after heavy rains, which produced concentrations from 24 to 81 ppm.

The chloride tracer part of the study offers some insights into the movement of solutes through the soil. More chloride was collected under no-till plots than under conventional plots until August and September, a trend that did not become clear with nitrate until the second year of the study.

Under no-till 37% of the applied chloride (396 kg/ha KCl broadcast in May, 1973) was collected from May 1973 to September 1974. During that time 23% of the chloride applied to conventional till plots was recovered. The remainder is assumed to be stored in the soil profile.

Chloride movement was not identical to water movement. The authors explained the discrepancies as resulting from water flowing through cracks and channels, bypassing chloride held by soil.

On silty soils the operation of farm equipment can result in the development of a traffic pan. Ehlers (1975) reported that the saturated hydraulic conductivity of a traffic pan in a silt loam was an order of magnitude less than that of the tilled soil above it. Percolation under conventionally tilled fields could be restricted by such a pan.

11. Evapotranspiration is decreased by no-till, maybe (0,+) increased by winter cover and chemical till-in, and is not affected by minimum-till, contouring, or contour strips.

Crop residue at the soil surface shades the soil and serves as a vapor barrier for water losses from the soil. Blevins et al. (1983a) reported estimated transpiration and evaporation values from May through September for conventional and no-till corn grown for four years on Maury silt loam, a deep, well-drained upland Kentucky soil (Typic Paleudalfs).

During these months no-tillage increased transpiration (relative to conventional tillage) by 30% and decreased evaporation by 80%, for a net decrease in evapotranspiration of 20%.

12. Sediment detachment by rainfall is decreased by no-till, minimum-till, spring plowing, and winter cover, but contour strips and contouring have no effect on detachment. Sediment detachment by rainfall is increased by chemical till-in.

Sediment detachment by rainfall varies with the surface protection afforded seasonally by cover (McGregor and Greer, 1982, Lindstrom and Onstad, 1984; Mueller et al., 1984a). The breaking of surface crusts by cultivation can temporarily increase soil detachment.

13. Sediment detachment by overland flow is decreased by no-till, minimum-till, contouring, strips, winter cover, and spring plowing and increased by chemical till-in.

No papers specifically addressing sediment detachment by overland flow as an isolated entity were found. More often runoff and sediment loss are reported as a function of tillage practice.

Tillage systems utilizing the chisel plow or disk substantially reduce soil losses compared to conventional tillage (Siemens and Oschwald, 1976; Johnson and Modenhauer, 1979). Several studies have reported soil loss reductions for these systems were comparable to those from no-till (Römkens et al., 1973; Griffith et al., 1977; Lindstrom and Onstad, 1984).

In many of these studies, in spite of higher sediment concentrations for chisel or disk systems relative to the no-till system, runoff was lower from chisel and disk, and as a result, soil losses were similar to those from no-till.

In contrast, Laflen et al. (1978) found chisel and disk systems less effective than no-till in reducing soil loss. Little difference in runoff was reported among tillage systems, but no-till plots produced lower sediment concentrations in runoff.

Römkens et al., 1973 compared the effects of five tillage practices on nitrogen and phosphorus in runoff from 2 simulated rainstorms occurring in the second to fourth week after planting of corn (when plant heights were 2 to 20 cm). The tillage systems were (1) conventional: plow, disk twice, plant, (2) ridge-till plant, (3) chisel till plant, (4) double disk: disk twice, coulter plant, and (5) coulter plant.

The test site soil was a Bedford silt loam (18.3% clay, 74.4% silt, 4.8% sand, and 2.5% organic matter in the plow layer). Slopes ranged from 8.2 to 12.4%, and tillage was across slope.

They concluded that the contour coulter and contour chisel systems controlled soil loss during the establishment phase of corn growth, but runoff water contained high levels of soluble N and P from surface applied fertilizer.

The contour disk and contour ridge till systems were less effective in controlling soil erosion, but produced runoff with lower concentrations of soluble N and P. Contour conventional tillage, in which fertilizer was plowed under, had the highest losses of soil and water but the smallest losses of soluble N and P in runoff. All treatments lost large amounts of sediment bound nutrients.

14. Sediment transport by overland flow is decreased by no-till, minimum-till, contouring, strips, winter cover, and spring plowing and increased by chemical till-in.

    Not addressed.

15. Sediment delivery to streams is not affected by any tillage BMP.

    Not addressed.

16. Gully erosion and scour is reduced by no-till, minimum-till, contouring, contour strips, and may be reduced (0,-) by spring plow and winter cover.

    Not addressed.

17. The production of sediment fines is decreased by no-till and minimum-till, perhaps decreased by contour strips and spring plow, and either increased or decreased (±) by winter cover. Contouring has no effect on the production of sediment fines. Chemical till-in increases fines.

None of the studies reviewed directly addressed the tillage effects on sediment particle size. Some provided theoretical discussions or measurements from a single tillage practice.

Storm flow and base flow runoff samples from chisel till corn were collected by Lietman et al. (1983). During base flow the amount of total suspended sediment finer than 0.062 mm averaged 51 percent. During storms this percentage ranged from 85 to 98%, averaging 92%.

Hubbard et al., (1982) studied runoff from conventional till corn and soybeans. Changing patterns of detachment and transport, and of sorted sedimentation, were observed during the course of individual events and from event to event.

These patterns varied with runoff stage and duration and with rainfall intensity, and were reflected in widely varying sediment loadings, nutrient content, and nutrient distributions within events and between events of varying magnitude. Particle size was not measured.

Particulates in early increments of runoff were comprised mainly of plant debris and dark colored fines in relatively stable suspension. In later increments, denser, light colored fine and very fine sand of lower N and P content appeared in proportions directly related to flow rate.

Römkens et al. (1973) described the process of selective soil erosion, in which small colloidal particles are transported preferentially to the larger silt or sand size particles.

Features contributing to roughness, such as ridges across slope and corn residue on the soil surface act as sediment traps. Systems that induce these surface conditions, such as chisel and coulter, yield sediments with a relatively higher clay fraction than systems with less surface roughness. Römkens et al. found that clay content of sediment correlated well with total N and P content.

At different times the contrast between tillages may or may not be pronounced. Immediately after moldboard plowing, but before disking soil surfaces are very rough. This roughness is dramatically reduced by the first good rainfall (Cogo et al., 1984).

In general roughness values are low in planted fields (Lindstrom and Onstad, 1984). After fall primary tillage, surface elevations may be quite variable, giving high random roughness with the capacity for considerable surface storage of water, but after a winter of weathering, secondary tillage, and planting, random roughness declines.

During the major part of the growing season (May through July), the random roughness is determined by the cultivation schedule, not tillage (Allmaras et al., 1966).

Cultivation roughens the soil surface. If no cultivation is performed, the roughness declines to a smooth base level (the roughness existing before tillage). If no-till fields are cultivated, they show the same pattern as do conventionally tilled cultivated fields, otherwise no-till field roughness is at the base level throughout this period.

After harvest there is not much difference in roughness between tillages (Siemens and Oschwald, 1978). Also after a winter of weathering, roughness differences are slight, between tillages, as well as before and after rainfall.

How, when, or if selective erosion modifies the particle size distribution of sediment in runoff remains to be shown.

18. Aggregation and compaction are decreased by no-till and minimum-till, chemical till-in, and maybe (-,0) by winter cover. Contour strips, contouring, and spring plowing (0,?) have no effect on soil compaction.

The movement of heavy farm equipment compacts about 25% of a conventionally tilled field. Since reduced tillage often means less field traffic, the amount of compacted soil should be less.

On a finer scale, ignoring the effect of heavy equipment, if soil bulk density and plow layer porosity are proper indicators of aggregation and compaction, no-till increases compaction, rather than decreases it relative to conventional tillage.

Factors influencing porosity include soil type, soil management history, and moisture content at the time of tillage. Plowing increases the surface layer porosity. Seedbed preparation and post-planting cultivation decreases porosity on friable soils, but weathering is responsible for the bulk of the annual decrease in porosity in these soils (Allmaras et al., 1966). On wet soils disking and harrowing can increase porosity.

Edwards (1982) observed that with continuous no-tillage management, plow layer porosity decreased from 50 to 40%, but the preserved macropores in the soil and crop residue on the surface reduced runoff to 1/20 of that from a nearby conventionally tilled watershed. Conventional tillage increased the immediately available pore space near the surface, but decreased the amount of infiltrating water moving deeper into the profile.

19. Surface and subsurface nutrient availability are increased by no-till and minimum-till, maybe increased (0,+) by spring plow and winter cover, and decreased by chemical till-in and contour strips.

The most important determinant of nutrient availability in soil is fertilizer rate, the more fertilizer applied, the more nitrogen is available (Kitur et al., 1984).

According to Kitur et al., 40 to 50% of the fertilizer applied will be immobilized in the first 90 cm of soil. No-till immobilizes roughly 20 to 25 percent more N in the plow layer than does conventional tillage. The plow layer to deeper layer N split (of total soil N) is roughly 2 to 1. Sixty percent of no-till plow layer N is in the first 5 cm. Closer to 30% of the plow layer N for conventional tillage is in the first 5 cm.

Soil N under no-tillage often exceeded that in plowed soil (House et al., 1984). Annually the soil organic N fraction was 40 to 50 percent higher with no-tillage. The upper soil layer (0-10 cm) under no-tillage contained as much as 20% more total N than the same soil layer in plowed soils.

The differences dropped sharply with increasing soil depth although at the lower soil depth (40-50 cm) nitrate-N concentrations were higher in conventionally tilled soils than in no-tillage soils. Leaching losses from conventional tillage were about 20 percent greater than those from no-tillage.

The higher concentration of organic matter in the upper soil under no-tillage plays a major role in the immobilization of N, especially surface applied fertilizer. Rather than being a nitrogen sink, this large pool of immobilized N in no-tillage soils represents and enhancement of storage capacity and promotes long-term N retention within untilled agroecosystems.

Timmons et al. (1973) studied the following fertilizer placement options:

1) Plot plowed and disked, no fertilizer applied
2) Fertilizer broadcast on oat stubble, plot plowed and disked
3) Plot plowed, fertilizer broadcast and disked in
4) Plot plowed, fertilizer broadcast on plowed surface
5) Plot plowed and disked, fertilizer broadcast on disked surface

They did not find much difference in the surface losses from the five treatments. Plowing down the fertilizer reduced the nutrient losses relative to surface broadcasting on top of plowed or disked surfaces. Disking down the fertilizer was inferior in nutrient retention to the deep placement achieved with inversion plowing.

Andraski et al. (1985) designed an experiment to show the effect of surface broadcasting fertilizer (a practice common in no-tillage, though also done with other tillages) on P losses in runoff for conventional till, chisel till, till-plant, and no-till systems. P fertilizer was subsurface banded and runoff was generated by simulated rainfall several times in two years.

There was little difference in dissolved phosphorus concentrations in runoff between the tillages, but particulate P concentrations from conventional tillage were consistently higher than no-till and often higher than other conservation tillages.

Thus the conservation tillages reduced total P concentration and losses by controlling soil loss. When fertilizer was buried, no-till, chisel till, and till-planting reduced total P losses by an average of 81, 70 and 59%, respectively, relative to conventional tillage.

The no-till, chisel till and till-plant treatments reduced algal available P losses (dissolved and particulate) in runoff by an average of 63, 58 and 27%, respectively, relative to conventional tillage.

Andraski et al. claim that phosphorus losses would have been roughly twice those reported if the P fertilizer had been surface broadcast.

This conjecture was (loosely) based on their observation that for each sampling period, the comparative magnitude of total P loss and soil loss reductions afforded by each conservation tillage treatment relative to conventional tillage were similar, and that the reduction of total P loss was on the average only about 5% less than soil loss reduction.

They couple this result to the observation of Römkens et al. (1973) that total P loss reduction was about one half that of soil loss reduction for a no-till treatment where fertilizer P was broadcast.

The conjecture seems to hold for nitrate. Whitaker et al. (1978) reported an annual 50% reduction in dissolved N in runoff from no-till corn when fertilizer was subsurface banded. (They found that 87% of the runoff N loss from no-till was in dissolved forms.)

A feature of conservation tillage is the maintenance of residue cover. The potential of this surface pool of organic matter to supply dissolved mineral nutrients to runoff has been studied by leaching experiments. The additional residue typical of conservation tillage does not significantly increase dissolved inorganic N and P concentrations in runoff (Baker and Laflen, 1982).

20. Chemical transformation may (0,-) be decreased by no-till and minimum-till. Contouring, contour strips, and winter cover have no effect. Spring plowing may increase transformation.

Tillage does not affect N mineralization rate, but season does (El-Haris et al., 1983). Fall mineralization rates (when mineralizable pool is measured in mg/kg soil) for chisel, conventional, and no-till averaged 0.014 day^{-1}. Spring rates averaged 0.050 day^{-1}. A temperature correction factor was not used.

The plow layer nitrification rates (describing the oxidation of ammonium to nitrate) increase with soil moisture (Rice and Smith, 1983). No-till nitrification rates are slightly higher than conventional till rates.

Denitrification rates in no-till fields averaged 5-10 kg N per hectare per year with wide seasonal variation. Most of the N loss occurred in fall and spring. Plowing and drainage both diminished denitrification losses (Colbourn, 1985).

Rice and Smith (1982) observed that denitrification rate was linked to moisture level, and concluded that the soils under no-till with a well developed mulch cover would be expected to lose more N by denitrification than they would under conventional tillage because of moisture retention by the mulch cover.

Soil tillage practices proved to be important regulators of N cycling processes (House et al., 1983). The seasonal N fluxes differed between many conventional and no-tillage agroecosystem components.

Plowing increased the total annual amount and rate of N uptake by crop plants (about 15% more nitrogen was taken up by conventional till crops) and increased the amount of sorghum litter decomposition (which was about 20% higher with conventional tillage).

In House's study, standing stocks of surface residue where consistently higher in no-tillage systems than in conventional systems. No-tillage litter was composed primarily of residue from the previous crop, while much of the conventional tillage litter, especially late in the season, was leaf-fall from the current crop.

At the termination of each cropping season, a large pulse of crop residue was added to the decomposition subsystem. The N in this organic matter was substantial (up to 100 kg/ha) and contributed significantly to the overall N economy of no-tillage agroecosystems.

Maximum rates of surface litter decomposition occurred in the summer.

21. Nutrient uptake/degradation is increased by winter cover, but has questionable if any relationship to the other BMPs.

At low rates of N fertilization no-till corn is generally more N deficient and yields less than corn under conventional tillage (Bandel et al., 1975; Blevins et al., 1977). Yield reduction has been attributed to lowered in N uptake (because of lowered N availability) as well as decreased soil pH from surface localization of fertilizer.

Kitur et al. (1984) found that about 75% of the N applied as fertilizer is recovered in corn as grain and stover, regardless of tillage.

Conventional tillage increased the total annual amount and rate of N uptake by sorghum and soybeans by about 15% relative to no-till crops (House et al., 1984).

Reduced N uptake by crops under the first years of no-till systems usually is most evident at lower rates of N fertilizer. In the first year of their study Stinner et al. (1983) found reduced N uptake in no-till and conventional tilled sorghum, winter wheat, and weeds (293 versus 350 kg/ha biomass), but no difference was detected in the second year (221 versus 218 kg/ha).

Rice et al. (1986) proposed an explanation for the transient nature of the increased N uptake in the first years of convention al tillage relative to no-tillage. When the established sod was broken at the initiation of continuous corn cropping there was a rapid decline in soil organic N.

The loss of N from the surface 5 cm was almost threefold greater in their conventional tilled plot than in the no-till plot. The higher mineralization rate associated with tillage is offset by the reduced organic N reservoir under conventional tillage, resulting in roughly equal N availability in either tillage after 5 to 10 years.

Surprisingly little of the N taken up in a year by a no-till crop comes from the current year's fertilizer. Much more comes from the soil nitrogen pool.

About 89% of the N taken up by no-till corn in a year is from soil reservoirs, 6% from current fertilizer, 5% from the previous year's fertilizer, and less than 1% from decayed crop residue (Power et al., 1986).

For soybeans (all residue left) these percentages are soil pool plus N fixation, 66%, 9% current fertilizer, 4% last year's fertilizer, and 22% from decayed residue. Parallel studies for conventional tillage would shed light on the uptake issue.

22. Nutrient adsorption/desorption is probably (?) increased by no-till, minimum-till, contour strips, and winter cover. Contouring and spring plowing have no effect on nutrient adsorption/desorption.

The BMP-related factors possibly affecting the adsorption/desorption coefficients of phosphate and ammonium would be soil pH in the top 7 cm and soil temperature.

23. Soil temperature can be either increased or decreased by no-till, spring plow, and minimum-till. Winter cover and chemical till-in increase soil temperature, and contouring and contour strips have no effect.

Tollner et al. (1984) reported that near surface temperatures (in the top 5 cm of soil) of conventionally tilled soybean plots were often 5-6°C warmer early in the season, however, towards the end of the season (mid-August), no difference was detected.

Soil temperatures generally are cooler early in the growing season with no-tillage corn. Differences of 2 to 3 °C at a 10 cm depth are common during May and June (Griffith et al., 1977). At 8 weeks after planting corn, the difference between no-till and conventional till soil temperature is about 4°C, with soil temperatures in chisel and ridge-till soils about 2°C warmer than no-till soils.

At eight weeks, no difference in soil temperature between fall and spring plowed plots was evident.

24. Soil bulk density is increased by no-till, and minimum-till, decreased by chemical till-in, and is not affected by contouring, contour strips, spring plowing, or winter cover.

Soil bulk density is primarily determined by soil type, however, agricultural activities modify this parameter at different depths and at different times of the year.

The bulk density of the plow layer over the year reflects the action of tillage and weathering. The greatest difference between tillages is evident immediately after planting, but before planting Blevins et al. (1977 and 1983a,b) reported no difference between no-till and conventional till corn surface layer bulk density measurements.

Bulk densities of the "plow layer" under no-till revert to those found under undisturbed sod after several years.

Tollner et al. (1984), summarizing six years of data collected at unspecified times of year, found soil bulk density in the 0-15 cm layer of no-till soybeans to be 0.14 g/cm^3 higher than conventionally tilled soils, but the traffic pan (caused by twice annual plowing) in the conventionally tilled plot at 30-35 cm was 0.10 g/cm^3 more dense than the corresponding depth in the no-till plot.

Lindstrom and Onstad (1984) compared the soil bulk density after planting corn under three tillages: (1) conventional, (2) chisel, and (3) ridge-till. The ridge-till plot was considered to be a "modified" no-till treatment.

They found that immediately after planting the top 15 cm of soil under no-till was about 0.30 and 0.17 g/cm^3 more dense than the surface layer under the conventional and chisel tillages, respectively. They found no tillage-related difference in the bulk densities of deeper layers (though their definition of these layers, 15-30 and 30-75 cm, may have obscured any distinction).

Griffith et al. (1977) reported bulk densities for the top 20 cm of five different soils 2 to 3 weeks after planting corn. No-till bulk densities exceeded conventional till bulk densities by

0.11 to 0.36 g/cm^3 (average of 0.19 g/cm3 more). The finer the soil texture, the less the difference in bulk density.

After planting, the bulk densities of till-plant soils were 0.12 to 0.51 g/cm^3 less than those of no-till (averaging 0.30 g/cm^3 less).

The bulk densities of chisel till surface soils were closer to those of conventionally tilled soils than to no-till or till-plant soils.

A study by Edwards (1982) pointed out the differences in bulk density between wheel-track and uncompressed portions of a field over time. No-till corn plow layer (5-15 cm) bulk densities remained constant throughout the year. Furthermore, compaction by farm machinery did not affect this parameter.

On conventionally tilled plots, the pre-plant bulk density was about 0.30 g/cm^3 lower than the no-till soil, but after plowing this difference increased to 0.45 g/cm^3 in the uncompressed rows. This difference gradually decreased to less than pre-plant levels by December.

The plow layer bulk density increased by 0.04 g/cm^3 in the conventional till wheel-track areas. (An estimated 25% of a field is compressed by farm implement traffic under conventional tillage.)

CHAPTER 2. WATER QUALITY IMPACTS OF ANIMAL WASTE MANAGEMENT

INTRODUCTION

This chapter is intended as an aid in adapting existing numerical models of nonpoint source pollution to capture the features of animal waste management. Such modeling requires not only an understanding of the chemistry and hydrology of manure transport and transformation, but also relevant conceptualization of farm operations.

The chemical and physical properties of manure; the transformations manurial nutrients undergo after excretion; the effects of different types of manure storage on manure composition and on surface and ground waters; the contribution of pasture, rangeland, manured cropland, and feedlots to water pollution; and the changes in soil properties engendered by manure application are examples of the possible components of such a numerical model.

Integration of these components within the farm activity framework is necessary if the model is to predict the water quality effects of a given manure management practice. Isolation of the model components out of context gives a distorted impression of their importance.

This chapter attempts to present important reaction rates, experimental data, and justifiable generalizations about the data which may be used in various modeling efforts. The initial use of this work is in conceptualizing and parameterizing a model of the nonpoint pollution in the Monocacy River Basin sponsored by the Maryland Department of the Environment and the Interstate Commission on the Potomac River Basin. The Monocacy model is an application of HSPF, Hydrologic Simulation Program-Fortran

I. COMPOSITION AND CHARACTERISTICS OF MANURE

A. CHEMICAL COMPOSITION OF MANURE

Table 90 shows livestock manure production and characteristics as compiled by the Midwest Plan Service (1983). The actual nutrient content of manure is affected by the methods of storage and handling, feed type, and bedding material.

Table 90
Manure Production and Characteristics
Midwest Plan Service, 1983

Animal Weight,kg	Manure kg/day	Water %	TS[1] kg/day	VS[2] kg/day	BOD_5 kg/day	N kg/day	P kg/day
DAIRY CATTLE							
68	5	87.3	0.7	0.6	0.12	0.03	0.005
113	9	"	1.2	1.0	0.20	0.05	0.009
227	19	"	2.4	2.0	0.39	0.09	0.016
454	37	"	4.7	3.9	0.77	0.19	0.033
635	52	"	6.6	5.4	1.08	0.26	0.046
BEEF CATTLE							
227	14	88.4	1.6	1.4	0.4	0.08	0.025
340	20	"	2.4	2.0	0.5	0.12	0.038
454	27	"	3.1	2.7	0.7	0.15	0.050
567	34	"	3.9	3.4	0.9	0.20	0.064
Beef Cow	29	"	3.3	2.8	0.8	0.16	0.054
SWINE							
Nursery/Growing							
16	1.0	90.8	0.09	0.05	0.03	0.007	0.0024
29	1.9	"	0.18	0.14	0.06	0.013	0.0044
Finishing							
68	4.4	"	0.41	0.33	0.14	0.031	0.010
91	5.9	"	0.54	0.44	0.18	0.041	0.014
Gestate							
125	4.0	"	0.37	0.30	0.12	0.028	0.010
Sow							
170	15	"	1.36	1.09	0.45	0.104	0.034
Boar							
159	5	"	0.45	0.38	0.16	0.035	0.012
SHEEP							
45	1.8	75	0.45	0.39	0.04	0.020	0.0030
POULTRY							
Layers							
2	0.10	74.8	0.024	0.017	0.006	0.0001	0.0005
Broilers							
1	0.06	"	0.016	0.011	0.0001	0.0001	0.00024
HORSE							
454	20	79.5	4.3	3.4	0.12	0.012	0.021

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[1] Total Solids
[2] Volatile Solids

Muck and Richards (1983) reported major nitrogen species in dairy cow feces and urine (Table 91). Urine comprised about 21% of the manure on a weight basis. Guest et al. (1973) estimated that percentage as 29%.

Table 91
Nitrogen Compounds in Dairy Manure
Muck and Richards, 1983

	NH_3-N	Organic-N	Urea-N	TKN
	milligrams of N per gram manure wet basis			
Urine	0.38 ± 0.24	10.22 ± 2.90	7.30 ± 2.55	10.60 ± 3.05
Feces	0.13 ± 0.08	4.46 ± 0.70	0	4.58 ± 0.70

The nutrient content of manure is widley variable. These tables are offered as ballpark figures.

B. ERODIBILITY OF MANURE

Estimates of a manure erodibility factor, K, analogous to the K factor in the Universal Soil Loss Equation, are given by Khaleel et al., 1979b (See Table 92). Note that to convert from English K from the metric units: 1 K(English) = 0.1317 $kg \cdot h/N \cdot m^2$

Table 92
Manure Erodibility Factors for Various Types of Surface-Applied Manure
Khaleel et al., 1979b

Type of Manure	K factor $kg \cdot h/N \cdot m^2$	Remarks
Unpaved beef feedlot	0.0210	Measured
Liquid swine manure	0.0026	Measured
Poultry manure	0.0026	Calculated from USLE
Dairy manure	0.0066	Calculated from USLE

C. PARTICLE SIZE OF MANURE

The d_{50} (the size of particle than which 50% of the sample is finer) is assumed to be representative of the sample for transport analyses. Typical d_{50} values and particle densities are given in

Table 93. For comparison sake some typical values for soil particles are included. Chang and Rible's figures are for manure as deposited on a surface. Refer to the original for particle size analysis of composted and fresh collected manure.

Table 93
Median Particle Size and Density for Various Manure Types
Khaleel et al., 1979b and Chang and Rible, 1975

	Particle size, d_{50} Khaleel mm	Particle size, d_{50} Chang & Rible mm	Particle Density Khaleel g/cm^3
Manure Type			
Poultry	0.035	0.5	1.80
Beef	0.090	> 1	1.44
Dairy	0.100	> 1	1.44
Swine	0.070		1.44
Soil Fraction			
Fine primary	0.005		2.65
Coarse primary	0.040		2.65
Coarse aggregates	0.270		2.00

D. MANURE EXCRETION RATES

1. SPATIAL

The extent of manure cover will depend on the number of cow-hours spent on a pasture and on the dimensions of the pasture. Some rules of thumb from Peterson et al. (1956) for calculating manure cover are

1. On the average each animal defecates every 2 hours

2. Each defecation covers 0.093 m^2

2. TEMPORAL

Daily manure excretion rates are listed in Table 94 (Pennsylvania Department of Environmental Resources, 1986b). This can be compared to the rates in Table 90.

can be compared to the rates in Table 90.

Table 94
Manure Production and Characteristics (Manure as Voided)
Pennsylvania Department of Environmental Resources, 1986b

Animal	Weight lb	Manure Production lb/day per 1000 lb	% Dry Matter	N lb/ton	P as P_2O_5 lb/ton
Cattle					
Dairy	150-1500	82	13	10	4
Beef	400-1400	60	12	11	7
Veal	100-350	63	16	8	2.4
Swine					
Pigs	35-200	65	9	14	11
Gestating sow	275	32	9	14	11
Sow and 8 pigs	375	88	9	14	11
Boar	350	31	9	14	11
Sheep	100	40	25	23	8
Horse	1000	45	20	12	5
Poultry[1]					
Liquid		300	5	10	7
Fresh wet		61	25	30	20
Moist		32	50	40	40
Crumbly		22	70	60	55
Dry		18	85	100	70

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[1] Storage losses for all poultry manure categories already deducted

Another way to express manure production rate is presented by the Soil Conservation Service (Table 95). These figures give the amount of manure produced by a group of similar animals whose total weight is 1000 pounds).

Table 95
Manure Production and Characteristics
Soil Conservation Service, 1978

Animal	Manure Production	N	P
	lb/day per 1000 lb live weight		
Cattle			
Dairy	85	0.37	0.069
Beef	62	0.43	0.090
Swine			
Feeder	69	0.45	0.17
Breeder	50	-	-
Poultry	53	0.86	0.40
Ducks		1.42	0.62
Sheep	36	0.40	0.075
Horses	50	0.30	0.12
Catfish		1.6	0.25
People	31	0.20	0.024

II. EFFECTS OF COLLECTION, STORAGE, AND APPLICATION ON MANURE COMPOSITION

More than 50% of the nitrogen in fresh manure may be present as ammonia or be converted to ammonia in a very short time following excretion (Vanderholm, 1975). Much of this ammonia volatilizes into the air. Organic nitrogen compounds have also been observed to volatilize from feedlot surfaces (Elliot et al., 1971). Soluble forms of nitrogen and phosphorus may be leached by water passing through manure storage areas. When modeling manure management practices, it is important to take in account the nutrient losses occurring before land disposal as well as after it.

Manure decomposition begins immediately after excretion. Manure collection and storage systems vary tremendously in their ability to conserve N. In a review, Vanderholm (1975) indicated that N losses can range from 8 to 90%, depending on the type of collection, storage, and land application system. Of all manure management systems, N loss was highest from anaerobic lagoons and oxidation ditches. Nitrogen losses from a liquid deep pit system varied from 30 to 65%. Vanderholm (1975) summarized 32 studies of nutrient loss from manure (Table 96).

Table 96
Estimated Nutrient Losses During Manure Storage and Handling
Vanderholm, 1975

System	Percent Loss N	Percent Loss P
Deep pit storage, liquid spreading	30-65	
Anaerobic lagoon, irrigation or liquid spreading	60-80	30-50
Oxidation ditch, anaerobic lagoon, irrigation or liquid spreading	70-90	
Bedded confinement, solid spreading	30-40	
Open lot, solid spreading runoff collected and irrigated	50-60	

Gross N loss can be roughly partitioned into its contributing components: collection losses, storage losses, and application losses.

A. COLLECTION LOSSES

Muck and Richards (1983) showed that 1-day volatile N losses, especially from urinary N, are temperature dependent. They collected samples from barn floors (scraped daily) and measured ambient temperature as well as manure N species. Little TKN-N was lost below 0°C, but around 50% of the TKN-N was lost at 25°C. Their data could be summarized by a linear equation, though the authors offered a more sophisticated model (Muck and Steenhuis, 1982):

$$\text{TKN \% Loss upon collection} = 2.2 * (\text{Temperature } ^\circ\text{C}) \qquad [9]$$

$$\text{for} \quad 0 < {}^\circ\text{C} < 30$$

and

$$\text{TKN \% Loss upon collection} = 0 \qquad \text{for } {}^\circ\text{C} < 0 \qquad [10]$$

B. STORAGE AND APPLICATION LOSSES

Collection losses are distinct from the losses due to storage (i.e., the difference between manure N in receiving pits and in storage tanks), which tend to be on the order of 5 to 15 percent of the as excreted manure N (Welty et al., 1984).

The type and duration of storage before spreading influence nitrogen loss through ammonia volatilization. The Pennsylvania Department of Environmental Resources (PA DER, 1986a) estimates the nitrogen loss from swine manure stored in pits, anaerobic lagoons, and surface aerated lagoons to be 20%, 50% and 90% respectively.

Most reports of the nutrient value of manure treat storage and application losses together. In another publication (PA DER, 1986b) the nitrogen lost under 11 storage situations and 3 storage plus spreading systems is estimated (Table 97). These losses presumably include physical losses from processes such as wash off, as well as chemical processes such as volatilization of ammonia.

Table 97
Nitrogen Loss from Manure Storage and Handling Systems
Pennsylvania Department of Environmental Resources, 1986b

Storage and Handling System	Percent N Lost
NOT INCLUDING LOSS AS A RESULT OF FIELD APPLICATION	
1. Bottom loading, covered, watertight liquid and solid storage	0-20
2. Manure liquids and solids hauled daily	15-25
3. Manure liquids and solids held in a covered, watertight structure	20-30
4. Manure liquids and solids held in an uncovered, watertight structure	30-40
5. Manure liquids and solids held in a storage pond. Contents agitated before spreading	30-40
6. Manure and bedding held in roofed storages	30-40
7. Manure without bedding held in unroofed storages; leachate is lost; solids are spread	45-55
8. Manure stored on open feedlot surface; only the solids are spread	50-60
9. Poultry layer manure stored in roofed shallow pit cleaned every 3 to 6 months	30-40
10. Broiler manure on sawdust or shavings in warm, humid climate; house cleaned every 4 months	45-55
11. Poultry layer manure in cool humid climate; stored in roofed, fan ventilated pits cleaned yearly	50-60
INCLUDING LOSS AS A RESULT OF FIELD APPLICATION	
1. Manure treated by anaerobic lagoon; applied on surface by irrigating or liquid spreading	55-65
2. Manure treated by anaerobic lagoon or stored in waste storage pond (if diluted more than 50%); applied on surface by irrigating or liquid spreading	70-80
3. Manure treated by aerobic lagoon or oxidation ditch followed by anaerobic storage of effluent; applied on surface by irrigating or liquid spreading	80-90

Some of the lower estimates seem to be in conflict with the results of Muck and Richards (1983), who found TKN losses of up to 60% occurring in a single day <u>before</u> the manure was even gathered for storage.

The PA DER (1986b) also describes the nitrogen loss effect of leaving stored manure on the surface of the land for various amounts of time (Table 98). It is not clear from the text whether these estimates include runoff losses or just volatilization losses. The percentages reported in this table are relative to the nitrogen in stored manure applied to land, not to the nitrogen in fresh manure.

Table 98
Percentage of Total Manure Nitrogen Remaining Available to Crops After Storage and Handling as Affected by Application Method
Pennsylvania Department of Environmental Resources, 1986b

Time of Application	Time of Incorporation	Manure N Available, %	
		Poultry	Other
This year	Immediate	75	50
	After 2 days	45	35
	After 4 days	30	30
	After ≥ 7 days	15	20
Previous fall		15	20
1 year ago		3	6
2 years ago		1	2

Conceptually, manure N is lost at three points during the collection and disposal process. Relative to fresh manure, collection N losses can be from 20 to 60%, storage losses from 5 to 15%, and application losses from 10 to 60%. Further N losses, once manure is applied to fields, are treated in Section IV of this report.

Phosphorus, unlike nitrogen, is not lost to volatilization, but is lost mainly in liquid discharge from manure accumulations and by washoff.

III. ENVIRONMENTAL CONSEQUENCES OF MANURE STORAGE

A. TOTAL CONTAINMENT TYPE STRUCTURES

The major purpose of manure storage is to accumulate and protect manure until it can be applied to land. Aside from the nitrogen volatilization losses that occur with storing and handling manure, manure storage does not result in the removal of nutrients from the ecosystem. Instead, manure storage allows a dairy farmer to time the field application of manure to his own convenience.

If the nutrient content of manure is to be best utilized, the manure should be applied to a nutrient demanding crop (such as corn) at a time (or times) when the crop is able to take up the nutrients. Usually that means before planting or just after planting. Once the crop is established, further fertilizer applications are generally not applied (though split fertilizer application is practiced by a minority of conservation-minded farmers).

A storage structure is sized according to the number of days of storage it affords a herd. In order to save all manure generated over the winter for spring application on corn, the minimum storage size is 180 days. A 180 day facility is emptied twice a year, once in the spring before planting and once in the fall after harvest. Unless a cover crop is planted, the fall applied manure is for the most part wasted.

Storages of other sizes serve different purposes. Short term storage (< 90 day storage) helps a farmer avoid spreading manure during severe snow or rainfall events, thus avoiding large runoff nutrient losses. Short term storage implies that not all the manure will be disposed of optimally for nutrient uptake, i.e. on corn fields. Alternate disposal sites include pasture, fallow land, and other available fields.

Examples of how manure storage capacity influences the nutrients leaving the fields to which manure is applied are presently non-existent. Several before-and-after-BMP evaluations of storage options are now in progress, but none are finished. Instead, we offer a modeling study (Young et al., 1985; Crowder and Young, 1985).

This very informative analysis compared nutrient management alternatives for Lancaster County, Pennsylvania dairy farms. The study area is the most intensely farmed portion of the Chesapeake Bay watershed. Because of large dairy, poultry, and swine populations, annual manure applications to cropland average 40 tons/ac.

Based on actual field conditions, estimates of field level losses of soil and nutrients under several alternative nutrient management scenarios were made with CREAMS (Chemicals, Runoff, and Erosion from Agricultural Management Systems model).

The authors emphasize that CREAMS modeling results do not represent actual nutrient losses. This is because the CREAMS parameters were not calibrated to monitoring data. For this reason modeling results from this model should be used to compare only the relative losses from different management practices.

Crowder and Young present results for a variety of crop sequences, slopes, manure application rates and practices, and

BMPs. Conventional management practices included conventional spring tillage[6] for corn for grain or silage and alfalfa hay. Several BMPs were modeled: reduced tillage[7], no-till planting, 120-foot wide strips, contouring, winter cover, grassed waterways, and pipe outlet terraces.

Manure storage capacities of 6 and 12 months were contrasted to daily hauling of manure. Daily spread manure was modeled with and without incorporation. Liquid manure injection was modeled for stored manure. Manure applications rates of 40, 30, 20, 15, and 10 tons/acre were modeled for corn, rates of 20, 10, and 0 tons for grass pasture and establishment year alfalfa, and 10, 5, and 0 tons for established alfalfa meadows. For crops whose nutrient needs could not be met by manure and starter fertilizer applications (for example, corn with animal manure applications of less than 20 tons), additional chemical fertilizer was plowed down at the time of tillage or topdressed under no-till.

The assumptions in the model are important in the interpretation of the results. Perhaps the most important assumption was that all farmers managed their farms to minimize nutrient movement. This was construed by the authors to mean that there were no barnyard losses. No spreading occurred on frozen or snow covered ground or within 2 days in advance of major precipitation. As a result, nutrient losses in surface runoff probably understate true losses for a typical farm.

The good management assumption also extended to residue management. After harvest of hay and corn for grain, it was assumed that adequate residue was left for runoff protection, so winter cover was not planted. Commercial fertilizer, plowed under for conventional and reduced tillage and topdressed for no-till, was added until <u>minimum</u> fertilization levels were satisfied for all crops. Part of the commercial fertilizer requirements were met by starter fertilizer applications, banded 2 inches below seed depth under all practices.

Daily manure spreading systems involved field spreading every 1 to several days. Manure was assumed to be stacked either in a barnyard or in an open shed (not in fields, as is typically encountered) during runoff events and periods when snow cover existed.

The modeling results presented did not account for precipitation dependent barnyard and storage losses of nutrients

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[6] one pass with each of the following: moldboard plow, tandem disk harrow, spike-tooth harrow

[7] consisting of one pass with a chisel plow followed by one pass with a tandem disk harrow

to surface and subsurface waters. The authors acknowledge that significant runoff losses from unimproved barnyards and stacks of manure can occur, and that these circumstances commonly exist on farms with daily spreading systems.

As good nutrient managers, the same high level of ability to prevent barnyard runoff was attributed to farmers conducting daily spreading as to those with storage systems. While this assumption is not realistic, the authors rationalized that it did not affect the comparison of <u>field</u> losses between daily spreading and storage systems. For the more general goal of evaluating BMPs in terms of their water quality impact, barnyard runoff pollution should be factored back in.

Manure applications from daily spreading systems occurred immediately before corn planting and after harvesting. Additional applications were made in late fall and late winter as necessary. It was assumed that a specific point in any field was manured no more than four times a year. The total application rate was divided by the number of applications per year to arrive at the incremental application rate (e.g. the annual rate of 40 tons per acre was delivered in four 10 ton/acre trips). These assumptions may also tend to underestimate nutrient loss from this management system.

Manure destined for alfalfa was applied after the last cut in established meadows and before planting of newly seeded alfalfa. Pasture was manured during the spring and summer growing season when other land was not available for manure disposal.

For 6 month storage systems, manure was modeled as applied prior to planting and crop growth, and after fall harvest for both corn and alfalfa. Pasture was also manured in both spring and fall. For 12 month storage systems, manure was applied so that nutrients would be available during the crop's growing season, namely, in spring for all crops except established alfalfa which received manure in the fall after the last cutting. The assumed nutrient requirements are listed in Table 99.

The authors assumed that the manure applied was a mixture of cattle, swine, poultry, horse and mule manures containing 11 pounds of N per ton (as voided) and 5.5 pounds of P_2O_5 per ton. Each manure handling option was associated with an a priori nutrient loss which was deducted from the manure nutrient content before land application. The losses combined barnyard runoff, storage, transfer, and application losses. These are found in Table 99 part B.

Table 99
Modeling Assumptions for CREAMS Runs
Crowder and Young, 1985

A. Nutrient Requirements

Crop	Yield	N required	P_2O_5 required
corn grain	120 bushels	120	30
corn silage	20 tons	140	32
alfalfa hay			
establishment year	2.9 tons	20	20
established meadow	4.4 tons	0	20
grass pasture	2.5 tons	50	50

B. Non-Field Nutrient Losses

	%N loss	%P loss
Daily spread	50	25
6 month storage		
spring plowdown	30	25
spring topdress	40	25
fall topdress	50	25
spring injection	23	25
fall injection	44	25
12 month storage		
spring plowdown	30	25
spring topdress	40	25
spring injection	23	25

All manure nutrients applied to the field were considered to be available to the crop.

Crops were modeled as growing on a Duffield silt loam. This series consists of deep well-drained highly productive limestone soils with high moisture holding capacity. Pasture was modeled on Lindside silt loam at 1.5% slope because this series was commonly found adjacent to stream channels. These soils are deep, moderately well drained to somewhat poorly drained, and fertile. They develop from alluvium.

The results of these simulations are summarized in 40 pages of tables in the appendix of Crowder and Young. An excerpt of annual nutrient loads for a continuous corn silage crop followed by a winter cover of rye planted in late September grown on 5% slope at 20 and 40 tons/acre manure application rates is found in Table 100.

Table 100
CREAMS Estimates of Annual Field Level Nutrient Losses for Corn Silage on 5% Slope at Two Manure Loading Rates
Young et al., 1985

	N Loss Percolate		N Loss Surface		N Loss Total		P Loss Total	
Manure Application Rate (t/ac)	20	40	20	40	20	40	20	40
				lbs/ac•yr				
Conventional tillage								
Daily spread	35	54	52	89	86	143	25	49
6 mo. storage; injection	38	74	51	87	89	161	24	40
12 mo. storage; injection	16	62	51	91	67	153	24	40
Reduced tillage								
Daily spread	35	55	34	59	69	114	16	27
6 mo. storage; injection	39	76	33	57	72	133	15	25
12 mo. storage; injection	18	65	33	60	51	125	15	25
Reduced tillage; contoured terraces; sod waterway; winter cover								
Daily spread	38	59	12	22	50	81	4	8
6 mo. storage; injection	40	77	12	21	52	98	4	7
12 mo. storage; injection	18	64	13	25	31	88	4	7

Storage and proper application of manure were effective practices for reducing nutrient losses at the proper nutrient loading rate but not at high nutrient application rates. At a 20 tons/acre•yr loading rate, field level losses are similar for daily spreading and 6 month storage (exclusive of barnyard runoff losses) and daily spreading. With 40 tons/ac•yr of manure, N losses are higher with 6 month storage than with daily spreading, because of increased percolate losses, even with conventional tillage. (With fertilizer injection or plowdown, runoff losses are not expected to increase much.)

The important conclusions to draw from Table 100 are that (1) six month[8] (or shorter) storage of dairy manure does not necessarily result a reduction in the amount of the N and P

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[8] 12-month storage doesn't exist in the Monocacy basin. The largest capacity encountered is 6 months.

discharged to the environment from manured fields, (2) manure storage without structural BMPs may not produce the desired amount of nutrient loss reduction, and (3) nutrient management can reduce nutrient loss significantly (35 to 60% in this case).

Manure storage used in concert with a variety of conservation practices (including injection) can result in nearly 40% N savings over daily spreading (Table 101) at the 20 tons/ac rate, but without the expensive structural BMPs, the edge of field nitrogen loss is only reduced by 20%.[9] It appears (if the BMPs were modeled correctly) that for situations when fields are over fertilized (40 tons/ac), manure storage without other structural BMPs is of little overall value. If the installation of a manure storage facility implied reduced barnyard runoff losses (which was not an assumption of this study), the actual N savings could be significantly greater.

Table 101
Edge of Field Nitrogen Loss Reduction From Manure Storage Plus Other Conservation Measures at Two Manure Application Rates
after Young et al., 1985

Conservation Practice	Percent N Change Relative to Daily Spread Manure for Conventional Tillage Silage Corn with Winter Rye		
20 tons/ac	Percolate	Surface	Total
6 month storage; manure injection; reduced tillage	+4	-21	-19
6 month storage; reduced tillage 120 ft contoured terraces with a sod waterway; and winter cover	+6	-45	-38
40 tons/ac			
6 month storage; manure injection; reduced tillage	+15	-22	-7
6 month storage; reduced tillage 120 ft contoured terraces with a sod waterway; and winter cover	+16	-48	-31

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[9] This finding is relevant to current Chesapeake Bay Program efforts in assigning nutrient credits to BMPs.

Table 102
CREAMS Estimates of Nutrient Losses for Corn Silage Following Corn Silage on a 5 Percent Slope
Crowder and Young, 1985

Management Practice /manure application	Manure t/ac	Storage Period	Runoff-N	Sediment-N	Percolate-N	Surface-N	Total-N	Runoff-P	Sediment-P	Total-P
						pounds	per acre	lost per	year	
Conventional/topdress	40	daily	13.58	75.63	53.96	89.21	143.17	2.60	39.33	41.93
Terraces/topdress	40	daily	12.54	27.62	57.08	40.16	97.24	2.36	14.36	16.72
Reduced Till/topdress	40	daily	12.34	46.89	54.99	59.22	114.22	2.35	24.38	26.73
All BMPs/topdress	40	daily	9.47	12.71	59.20	22.18	81.38	1.78	6.61	8.39
No-till/topdress	40	daily	14.93	30.48	49.03	45.41	94.44	3.63	15.85	19.48
Conventional/plowdown	40	6 month	9.35	75.63	66.28	84.98	151.26	0.59	39.33	39.92
Terraces/plowdown	40	6 month	8.67	27.62	70.66	36.29	106.94	0.54	14.36	14.91
Reduced Till/plowdown	40	6 month	8.54	46.89	67.84	55.43	123.28	0.57	24.38	24.96
All BMPs/plowdown	40	6 month	6.27	12.71	72.99	18.98	91.97	0.30	6.61	6.91
No-till/topdress	40	6 month	10.63	30.48	53.37	41.11	94.48	1.58	15.85	17.43
Conventional/plowdown	40	12 month	9.23	75.63	49.57	84.86	134.43	0.27	39.33	39.60
Terraces/plowdown	40	12 month	8.59	27.62	53.06	36.20	89.26	0.25	14.36	14.61
Reduced Till/plowdown	40	12 month	8.18	46.89	51.72	55.07	106.80	0.27	24.38	24.65
All BMPs/plowdown	40	12 month	7.85	12.71	55.74	20.56	76.30	0.23	6.61	6.84
No-till/topdress	40	12 month	12.47	30.48	30.30	42.95	73.26	1.48	15.85	17.33
Conventional/injection	40	12 month	15.29	75.63	62.19	90.95	153.11	0.32	39.33	39.65
Terraces/injection	40	12 month	14.18	27.62	66.47	41.80	108.27	0.30	14.36	14.66
Reduced Till/injection	40	12 month	13.43	46.89	64.96	60.32	125.28	0.30	24.38	24.68
All BMPs/injection	40	12 month	12.38	12.71	63.70	25.09	88.79	0.29	6.61	6.90
No-till/injection	40	12 month	16.85	30.48	67.45	47.33	114.78	1.14	15.85	16.99
Conventional/topdress	20	daily	6.31	45.38	34.65	51.69	86.34	0.78	24.20	24.98
Terraces/topdress	20	daily	5.99	16.57	36.17	22.56	58.73	0.75	8.84	9.58
Reduced Till/topdress	20	daily	5.88	28.13	34.87	34.01	68.88	0.76	15.00	15.77
All BMPs/topdress	20	daily	3.98	7.63	37.83	11.60	49.44	0.31	4.07	4.38
No-till/topdress	20	daily	7.71	18.29	27.63	26.00	53.63	1.65	9.75	11.40
Conventional/plowdown	20	6 month	5.29	45.38	36.70	50.67	87.36	0.39	24.20	24.59
Terraces/plowdown	20	6 month	4.91	16.57	38.94	21.48	60.42	0.36	8.84	9.20
Reduced Till/plowdown	20	6 month	4.86	28.13	37.32	33.00	70.31	0.38	15.00	15.38
All BMPs/plowdown	20	6 month	3.59	7.63	40.28	11.22	51.50	0.23	4.07	4.30
No-till/topdress	20	6 month	6.63	18.29	30.04	24.92	54.96	1.25	9.75	11.00
Conventional/plowdown	20	12 month	4.80	45.38	26.10	50.18	76.28	0.23	24.20	24.43
Terraces/plowdown	20	12 month	4.47	16.57	27.73	21.04	48.77	0.21	8.84	9.05
Reduced Till/plowdown	20	12 month	4.29	28.13	26.74	32.43	59.17	0.23	15.00	15.23
All BMPs/plowdown	20	12 month	4.00	7.63	29.12	11.63	40.75	0.20	4.07	4.26
No-till/topdress	20	12 month	6.15	18.29	13.69	24.44	38.13	1.19	9.75	10.95
Conventional/injection	20	12 month	5.89	45.38	16.30	51.27	67.57	0.25	24.20	24.45
Terraces/injection	20	12 month	5.56	16.57	18.02	22.13	40.15	0.24	8.84	9.08
Reduced Till/injection	20	12 month	5.21	28.13	17.65	33.34	50.99	0.24	15.00	15.24
All BMPs/injection	20	12 month	5.13	7.63	18.36	12.76	31.12	0.23	4.07	4.30
No-till/injection	20	12 month	6.35	18.29	15.82	24.64	40.46	1.03	9.75	10.78

Manure injection is not universally practiced. Runoff nutrient losses are significantly greater when manure is topdressed. Table 102 gives some of the nutrient loads generated by CREAMS (5% slope, continuous corn silage/winter rye, 20 and 40 tons/ac manure) for the full suite of BMPs and application methods.

Soil losses were greatest for corn silage following corn silage on a 9% slope with conventional tillage practices (28 tons/ac) and least for pasture and established alfalfa meadow on a 2% slope (<0.1 ton/ac).

The soil loss reductions acheived by various BMPs for a field growing corn silage following corn silage (with rye winter cover) on a 5% slope, relative to soil loss from conventional practices are listed in Table 103.

Table 103
Predicted Soil Loss Reductions Resulting From BMP Installation
Crowder and Young, 1985

Practice	% Soil Loss Reduction
stripcropping	22
contouring and stripcropping	39
terraces (contoured)	72
sod waterways	64
winter cover crops	14
reduced tillage	44
reduced tillage/strips/contours/winter cover/terraces/grassed waterway	89
no-till/winter cover	68
no-till/strips/contours/winter cover/terraces/grassed waterway	93

B. SOLID/LIQUID SEPARATION

A common form of manure storage in Maryland dairy farms is the collection of manure solids in walled structures which allow the liquids to drain out. Such structures are called picket fence dams or solid/liquid separators, among other names. Solids are retained for later land disposal and liquids are discharged onto a grassed waterway, a holding structure, a stream, or sometimes a wetland. Storage areas of 5 to 10 m^2 per cow are typical for small (<30 animals) herds (Converse et al., 1975).

Converse et al. (1975) measured the volume and chemical characteristics of the seepage from three different solid/liquid separator type manure storage facilities. Seepage was collected in holding tanks for sampling (at an unspecified frequency). Two of the storage areas were uncovered, and one was roofed. Average values for all three are reported in Table 104.

Seepage liquid in this study includes collectable urine and precipitation. During the winter, the cows were kept in a stanchion barn except for 30-45 minute daily exercise periods. Barn gutters were scraped daily. During the summer, the cows were kept in yards except when milked, and the gutters were cleaned about 3 times per week.

Comparable data from Loehr (1974) reporting the work of Cramer et al. (1971) are also found in Table 104.

Converse et al. note that heavy rains in summer resulted in washout, that is, manure flowing out of unenclosed stacks.

Magdoff et al. (1977) monitored the seepage and runoff coming off an uncovered picket fence type solid/liquid separator manure storage facility. Its dimensions were 18 x 26 x 1 meters, except on one 18 m long wall which was an earthen dike 0.33 m high. The bottom was water tight plastic, and runoff from adjacent areas and parlor waste water was not discharged into the facility. Occasionally, however, snow from adjacent areas did blow into the pit. Manure from a 62 cow stanchion barn containing sawdust bedding was stacked in the facility from January through April. This manure was spread in May and manure storage began again in mid-May through October. Liquids leaving the picket dam were monitored even when manure was not being stored.

Table 104

Characteristics of Seepage from Stacked Dairy Manure after Converse et al., 1975 and Loehr, 1974

	Winter	Summer	Annual
Volume of seepage, liters per day per 454 kg live body weight			
Converse et al.	8.0	5.1	6.5
Loehr	11.4	4.5	8.0
Total solids, percent			
Converse et al.	2.2	2.0	2.1
Loehr	2.8	2.3	2.5
Volatile solids, % of total solids			
Converse et al.	59.1	54.3	56.7
Loehr	55	53	54
BOD, ppm			
Converse et al.	9398	9136	9267
Loehr	13800	10300	12400
COD, ppm			
Converse et al.	23000	22000	22500
Loehr	31500	25900	28700
Total nitrogen, ppm as N			
Converse et al.	2339	1623	1981
Loehr	2350	1800	2075
Ammonia, ppm as N			
Converse et al.	1559	1119	1339
Loehr	1600	1330	1465
Total phosphorus, ppm as P			
Converse et al.	126	106	116
Loehr	280	190	235
Potassium, ppm as K			
Converse et al.	3804	3664	3734
Loehr	4700	3400	4050
pH Converse et al.	7.8	7.7	
Ratio of collected seepage volume to precipitation falling on storage unit (seepage/precipitation)[1]			
Converse et al.	1.1	0.4	0.8

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[1] The winter ratio does not take into account drifting snow. Study site was in Wisconsin.

The ranges in pollutant concentrations were substantial (Table 105), with the maximum values near those reported for unpaved beef cattle feedlot runoff and seepage from dairy manure stacks. The authors calculated that, using the mean values for COD and flow rate and assuming no turbulence, runoff from the pile could deplete the dissolved oxygen in a stream with a flow rate of about 17 l/s (0.6 cfs).[10]

Table 105
Runoff Characteristics From a Solid/Liquid Separator Manure Storage Facility
Magdoff et al., 1977

	Minimum	Maximum	Mean	Annual Loss kg/cow·yr
Flow rate (l/day)	0	6975	775	4447.27
Solids, ppm[1]	1400	57000	1800	82.02
Total-N, ppm	78	3953	1354	6.02
NH_4-N, ppm	49	2795	991	4.41
P, ppm	7	255	92	0.41
K, ppm	83	5400	1870	8.32
Ca, ppm	2	1330	384	1.71
Mg, ppm	6	590	128	0.57
Cl, ppm	85	3842	1335	5.94
COD, ppm	1129	50713	15543	69.12

---

[1] wet basis for all concentrations

Annual losses in the runoff amounted to 6.02 kg N, 0.41 kg P, and 8.32 kg K per cow. The amount of manure stored at the site was estimated to be 12.3 metric tons wet weight per cow. Based on this estimate, 2.9 percent of the solids were lost annually, 9.8 percent of the total N, 2.9 percent of the P, and 19.7 percent of the K. Phosphorus appears to be closely associated with the solids. About 73 percent of the nitrogen lost in runoff was ammonium nitrogen.

The authors presented a regression on monthly total N loss against monthly runoff volume with an R of 0.81:

Total N Loss, kg/month = -3.6 + 0.0014(1000 liters/mo. runoff)

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[10] (15,543 mg COD/l) * (0.00896 l/s) = 139.3 mg COD/s. If 139.3 mg COD were diluted in 17 l of water, it would exert an oxygen demand equal to the solubility of oxygen in water (8 mg/l).

The data (read from a graph) are listed in Table 106. The conversion for runoff volume is 1 cm precipitation at the site is equivalent to 4680 liters.

Table 106
The Relationship Between Nitrogen Loss and Runoff Volume For the Discharge From a Solid/Liquid Separator Manure Storage Facility
Magdoff et al., 1977

Total N Loss kg/month	Runoff Volume cm/month	Runoff Volume 1000 l/month
3.29	1.65	7.72
4.12	1.89	8.85
12.18	1.92	8.99
25.35	2.05	9.59
3.29	2.12	9.92
22.55	2.39	11.19
16.46	2.59	12.12
12.23	2.93	13.71
7.08	3.65	17.08
24.36	3.34	15.63
72.43	8.33	38.98
53.00	9.34	43.71
44.94	9.37	43.85
45.10	10.68	49.98
76.71	10.86	50.82
88.89	11.49	53.77

Magdoff et al. conclude that the amounts of nutrients in runoff from this kind of manure storage facility were high enough to cause deterioration of water quality in small streams and ponds. Substantial nutrient value was lost during manure storage. They state that runoff from such a facility should be confined in a lagoon and irrigated on cropland, or the manure stack should be covered to reduce the volume of effluent.

C. GROUND WATER POLLUTION FROM MANURE STORAGE FACILITIES

Field studies of lagoons shortly after start-up have demonstrated an initial flush behavior (Reese and Loudon, 1983). A seal develops in 2 or 3 months but elevated nutrient concentrations in ground water near the lagoons persist for about a year.

Different soil types have been studied as structural materials for manure storage. Though not all studies are in agreement, generally nitrogen accumulation below storages located in medium or fine textured soils are reported to be minimal (Ritter et al., 1981; Miller et al., 1976), but excessive accumulations of nitrogen below storages located on coarse textured soils have been noted (Phillips and Culley, 1985; Sewell, 1978; Miller et al., 1976).

Phillips and Culley (1985) compared the ground water chemistry under manure pits installed in three soil types (Table 107). After three years the seal phenomenon was least evident on the sandy soil. Also the pollutant concentrations increased over time.

Table 107
Nitrate Nitrogen Concentrations in Ground Water
Below Small Earthen Manure Storage Pits
Phillips and Culley, 1985

Soil	Depth	Nitrate-nitrogen, ppm			Total Phosphorus,ppb		
	m	1982	1983	1984	1982	1983	1984
Clay	1.75	0.44	0.09	0.35	86	212	184
	2.5	0.52	0.08	0.90	60	142	144
	3.5	0.38	0.12	0.68	200	346	199
	mean	0.45	0.09	0.64	115	233	176
Loam	1.75	0.84	0.11	4.52	34	73	32
	2.5	0.14	0.03	4.57	26	49	38
	3.5	0.05	0.02	0.33	26	28	34
	mean	0.34	0.05	3.14	29	50	35
Sand	1.75	7.26	22.2	10.2	43	32	281
	2.5	2.12	25.4	44.7	17	29	108
	3.5	8.18	10.9	22.6	22	13	329
	mean	5.85	19.5	25.8	27	25	239

Other factors may influence the severity of the ground water pollution. Older lagoons (> 6 years use) may leak more than newer ones (<3 years) (Miller et al., 1976). Local conditions such as tree roots or burrowing animals may circumvent self-sealing of the soil (Ritter et al., 1981). Abandoned lagoons often show high nitrate instead of ammonium pollution (ibid.).

Sometimes a lagoon might not leak, but the method of transferring manure to the lagoon might contaminate the ground water. Ritter and Chirnside (1984) reported ground water ammonia concentrations of 90 mg/l near a swine waste lagoon which was attributed not to the lagoon but to a messy pump station sump.

Ciravolo et al. (1979) reported ground water pollution from lagoon overflow rather than seepage. They noted that many lagoons were likely to overflow in the direction of ground water flow unless liquid removal preceded heavy rainfalls.

The severity of the ground water pollution is highly dependent on the local geohydrology and the time frame of the inquiry. Ground water nitrate concentrations vary with distance from source, depth of well, as well as with magnitude of source. Ritter and Chirnside (1984) reported average nitrate concentrations of wells located within 152 m of poultry houses (in a Delaware study area) of 15 mg/l, while the average concentration in wells more than 305 m from poultry houses was 4 mg/l. Forest ground water nitrate levels were about 1.5 mg/l. Wells near low density residential development or non-irrigated cropland had levels around 8 mg/l.

IV. APPLICATION OF MANURE TO CROPLAND AND DISPOSAL SITES

A. POLLUTION FROM PASTURE AND RANGELAND

1. CHANGES IN INFILTRATION

Westerman and Overcash (1980) measured infiltration rates on three pasture and three dairy feed lot sites (Table 108). The soil type was not mentioned, but it was indicated that it was the same for all sites. The pasture sites were irrigated with dairy manure lagoon effluent. The open lots were stocked at a rate of 100 m^2/cow (a relatively low stocking rate).

Table 108
Infiltration Rates Measured in Feedlot and Pasture Sites
Westerman and Overcash, 1980

Test Minutes	Infiltration rates cm/hr Dairy Feedlot Sites F1	F2	F3	Dairy Pasture Sites P1	P2	P3
0-15	5.08	2.95	1.32	19.1	13.7	5.38
15-30	1.42	0.71	0.71	15.2	9.1	3.86
30-60	0.36	0.56	0.51	12.2	8.0	2.84
60-90	0.10	0.41	0.30	11.2	7.3	1.68
90-120	0.05	0.25	0.20	10.2	7.5	1.37
120-150	-	-	-	8.6	7.2	-

Taraba et al. (1983) applied liquid and semi-solid dairy manure to test plots on an established fescue pasture on Murray silt loam (Kentucky). They showed that infiltration rates were initially reduced by manure application, but that this effect disappeared within four to five days if no rainfall was applied.

When simulated rainfall of 9 to 10 cm/hr was applied immediately after the manure was spread, the authors claim that infiltration rates were reduced by 6 to 15% for semi-solid manure and 23 to 31% for liquid manure.

The effect was not consistently observed with every trial. Nor were the estimates of infiltration rates of unmanured pasture without variance. Since the paper gave insufficient detail to allow for critical evaluation of the methods, the data are not reproduced in this report.

Unlike the previous studies, Converse et al. (1976) noted that runoff amounts were greater from unmanured alfalfa plots than from manured plots. They explained the increased infiltration of manured plots by the larger number of worms present in the plow layer of the manured plot (4 times more) and by the greater amount of plant material (both grass and mulch) on manured plots (twice the amount of grass and 8 times the amount of mulch).

Even though runoff was reduced by manure application, the overall nutrient export from manured plots was no different than that from the unmanured plots, because of the higher concentrations of nutrients in runoff from manured plots.

2. N AND P IN RUNOFF FROM PASTURE AND RANGELAND

Table 109 summarizes data from several studies on N and P losses in runoff from pastures and rangelands containing grazing animals, as compiled by Khaleel et al. (1979b).

Khaleel et al. conclude that total N and P yields from pastures are not related to the number of animals involved, but rather to hydrological and management factors, namely grazing management system, erosion and sediment transport control, and runoff control. This conclusion was also reached by Loehr (1974).

A field study of runoff from small homogeneous land use watersheds in northern Virginia was conducted in 1976 for the Metropolitan Washington Council of Governments (1979). One of these watersheds was a 6.7 ac pasture with a 4% slope and 290 ft overland flow length. This was part of a 30 acre pasture supporting 30 head of Charolais cattle. There was a pond on the site. No cattle were grazed in the third quarter of the 1 year study (winter) and limited grazing was observed in the fourth quarter (spring). Average concentrations are reported in Table 110.

Table 109
Quality of Runoff Water From Pastures and Rangelands
Khallel et al., 1979b

Location	Description of the site	Total N*		Total P		BOD		COD		Reference
		$mg\ l^{-1}$	$kg\ ha^{-1}\ yr^{-1}$	$mg\ l^{-1}$	$kg\ ha^{-1}\ yr^{-1}$	$mg\ l^{-1}$	$kg\ ha^{-1}\ yr^{-1}$	$mg\ l^{-1}$	$kg\ ha^{-1}\ yr^{-1}$	
Texas	Continuously grazed	0.94	—	0.04	—					Smeins (1976)
	Rotationally grazed	0.64	—	0.07	—					
Oklahoma	Continuously grazed	—	9.73	—	4.60					Olness *et al.* (1975)
	Rotationally grazed	—	2.09	—	1.27					
Tennessee	Heavily grazed pasture I	4.5†	—	7.1‡	—					Sewell & Alphin (1975)
	Heavily grazed pasture II	5.1	—	1.2	—					
S. Dakota	Pasture (with animals)	—	1.52	—	0.25					Dornbush *et al.* (1974)
	Grassland	—	0.97	—	0.10					
	Cultivated land	—	1.28	—	0.30					
N. Carolina	Site F—control	1.2	3.8	0.2‡	0.6‡	2.0	6.0	—		Robbins *et al.* (1971)
	Site E—200 sows on 1.2 ha of drylots plus wastes spread on 2 ha	1.1	2.8	1.2	3.0	4.7	11.0	—		
	Site K—200 hogs on 2.5 ha of drylots	1.4	2.8	1.9	3.8	6.4	13.0	—		
	Site H—dairy cows on 22 ha of pasture and wastewater from milking area followed directly into stream	7.1	14.1	4.6	9.2	17.0	34.0	139	276	
	Site J—dairy cows on 5 ha of pasture and wastewater from milking area followed directly into stream	19	145	18	143	165	1260	504	3918	
	Site P—54 mt of poultry wastes plus shavings spread on 6 ha, yearly	1.9	3.7	1.2	2.0	5.2	9.4	63	117	
	Site D—150 sows on 4 ha of drylot plus discharge from 100 confined hogs	13	78	6.9	40.9	73	429	—	—	
	Site Z—35 dairy cows on 6 ha pasture	1.2	2.1	1.1	2.0	9.8	17	149	264	
	Site X—20 mt of poultry wastes spread on 1.6 ha, once	60	159	—	—	265	679	600	1395	
S. Carolina	Site A—50 to 600 m above major farm drainage area§	2.5	—	2.5	—	12.4	—	17.7	—	Janzen *et al.* (1974)
	Site B—adjacent to major farm drainage area	3.6	—	3.8	—	50.5	—	68.3	—	
	Site C—50 to 600 m downstream of major farm drainage area	3.0	—	2.8	—	19.4	—	27.7	—	
Florida	Average of two pasture sites	—	5.3	—	0.3	—	—	—	—	Wanielista *et al.* (1975)

* Total N = Organic N + NH_4-N + NO_3-N.
† Includes only NO_3-N.
‡ Includes only Ortho-P.
§ Average of three disposal systems.

Table 110
Runoff Quality from Dairy Pasture
Metropolitan Council of Governments, 1978 and 1979

	mean ± s.d.	n
Total N, mg/l	5.1 ± 1.7	30
Dissolved N, % of TN	51.7 ± 24.4	30
Organic N, % of TN	88.0 ± 7.3	30
Ammonia N, % of TN	3.4 ± 2.4	30
TKN, % of TN	91.4 ± 5.6	30
NO_3-N +NO_2-N, % of TN	8.6 ± 5.6	30
Total P, mg/l	0.82 ± 0.40	30
Dissolved P, % of TP	39.5 ± 23.0	30
Ortho-P, % of TP	17.6 ± 16.3	30
Organic-P, % of TP	82.4 ± 16.3	30

As part of a literature review of loading factors for agricultural nonpoint pollution, Beaulac and Reckhow (1982) summarized several studies of nutrient export from grazed and pastured watersheds (Table 111).

3. PASTURE AND GRASSLAND AS MANURE DISPOSAL SITES

Dairies are often located in regions where land accessibility for waste application is restricted during large portions of the year by cropping patterns and climatic factors. Sometimes, for want of more suitable manure disposal sites, manure is spread on pasture, often at rates greatly exceeding the nutrient requirement of the vegetation.

Westerman and Overcash (1980) measured runoff nutrients from pasture plots irrigated with dairy manure lagoon effluent (see Table 106 above for infiltration rates) at a rate of about 1 cm of effluent every other week for during a 30 month study period. The lagoon effluent had average concentrations of 360 mg/l TKN, 144 mg/l NH_4-N, 68 mg/l TP, and 48 mg/l o-PO_4-P.

The authors developed linear relationships describing chemical transport as a function of the event flow volume X in cm (Table 112). Most of the data used in these regressions was collected during December, January or March, so the equations are more representative of winter/spring conditions than of summer/fall conditions.

Table 111
Total Nutrient Export from Grazed and Pastured Watersheds
Beaulac and Reckhow, 1982

Land Use	Fertilizer kg/ha yr N	Fertilizer kg/ha yr P	Soil Type	Precipitation cm/yr (range)	Runoff cm/yr	Nitrogen Export kg/ha yr (range)	Phosphorus Export kg/ha yr (range)
Mod. grazing[1]	37	16		106 (104-120)	21 (12-25)	3.46 (2.41-3.83)	0.14 (0.12-0.16)
Heavy grazing[2]	149	64		106 (104-120)	26 (20-32)	10.99 (8.31-18.05)	0.16 (0.11-0.70)
Pasture			sandy clay loam	58	4	1.52	0.25
Winter Grazed[3]	56	0	silt loam	108	13	30.85	3.6
Summer Grazed	56	0	silt loam	108	3	21.85	0.85[4]
Rotat'n Graze[5]	168	39	silt loam	74 (73-78)	4 (1-4)	2.32 (0.47-4.28)	0.25 (0.08-0.51)
Pasture	0	0		164	62		1.35
Grazing[6]			sandy loam	115		13.0	3.8
Grazing[7]	0	0	silt loam	88 (51-105)	15 (2-28)	6.13 (1.33-9.23)	1.46 (0.27-3.86)
Rotat'n Graze[8]	0	0	silt loam	88 (52-109)	6 (<1-18)	1.48 (0.15-2.3)	0.25 (0.02-1.44)
Cont's Grazing[9]	83	72	silt loam	76[10]	15	9.20	4.90
Rotat'n Graze	87	76	silt loam	785	4	4.72	3.09
Cont's Grazing[11]	0	0	silt loam	76[5]	10	5.19	0.76
Rotat'n Graze	0	0	silt loam	78[5]	4	1.73	0.20

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[1] moderate dairy grazing on bluegrass cover, four year median; sediment phase not sufficiently examined
[2] bluegrass cover
[3] winter grazed and summer rotational grazed; orchardgrass and bluegrass cover
[4] major contribution from underground spring
[5] rotational grazing; three year median
[6] continuous grazing with some supplementary winter feeding; some hay production
[7] continuous grazing, little bluestem cover, active gullies; four year median
[8] continuous grazing, good cover, four year median
[9] continuous grazing, little bluestem cover
[10] nine year mean
[11] continuous grazing, little bluestem cover, active gullies

Table 112
Runoff Nutrients from Manure-Irrigated Pasture: Winter-Spring
Westerman and Overcash, 1980

Independent Variable kg/ha·event	Number of Events	Constant a	Slope b	R^2
TKN	10	0.011	1.30	0.93
NO_3-N	9	0.004	0.80	0.69
TP	10	0.022	0.64	0.98

B. POLLUTION FROM MANURED FIELDS

Khaleel et al. (1980) also reviewed studies of runoff pollution from manured plots growing various crops (Table 113), and developed equations for predicting the annual load and concentration of nutrients in runoff from manured cropland (Table 114). They found strong linear relationships (i.e., $Y = a + bX$) between nutrient loading rate to the field and nutrient delivery in runoff.

Nutrient runoff data obtained under extreme weather conditions were not included in the regression analysis, so the relationships describe nutrient transport under ordinary climatic conditions.

Time of application, method of application, soil and cropping management practices were not studied. Nearly all data presented are for small plot sized areas, not watershed studies, so the dependent variable estimates may be higher than what would come off similar large watersheds. All predictions are edge-of-field.

In the equations, the unit of measurement of load (the dependent variable) is kg/ha·yr, but the year is divided into two parts, summer-fall and winter-spring. This poses a problem of interpretation: does Y represent 6 or 12 months' load? From the text we infer (weakly) that the equations represent 12 month loads for manure applications occurring under a single seasonal regime.

The effects of ground cover, field characteristics, and elapsed time before manure incorporation are not part of these models. The higher nutrient loads predicted in the winter are attributed to snowmelt. (For a 1000 kg/ha·yr application of N and P, the runoff nitrogen loads are 13 times higher and the phosphorus loads are 3 times higher than for the summer-fall predictions.)

Table 113
Transport of Total N and P in Runoff Water From Plot Areas Receiving Animal Wastes
Khaleel et al., 1979b

Location	Type of manure	Time of application	Total N applied	Total N* runoff	Total P applied	Total P runoff	BOD runoff	COD runoff	Remarks	Reference
			$kg\ ha^{-1}\ yr^{-1}$							
Wisconsin	Fresh dairy	Winter	120	12.7	41.5	2.9	—	—	8 plots @ 3.0 × 12.2 m, 10–12% slope, silt loam soil, corn, 3 yr study	Minshall *et al.* (1970)
	Fermented	Spring	115	4.0	42.4	0.8	—	—		
	Liquid	Spring	95	3.6	35.8	1.0	—	—		
Wisconsin	Dairy	Fall	128	13.1	22.5	3.5	—	—	10 plots @ 3 × 13.2 m, 10–12% slopes, silt loam soil, alfalfa, 3 yr study	Converse *et al.* (1976)
		Winter	121	18.7	24.8	2.4	—	—		
		Spring	114	9.1	25.9	1.6	—	—		
N. Carolina	Swine lagoon effluent		1344	23.4	605	11.6	—	202	9 plots @ 2.8 × 4.6 m, 1–3% slope, sandy loam, coastal bermuda grass, 2 yr study	Overcash (1976, Unpublished data)
			672	9.4	312	2.8	—	156		
			336	8.0	151	3.0	—	124		
Alabama	Liquid dairy		5661	13.8	1943	9.0†	39.1	208	12 plots @ 1.5 × 3.0 m, 3.3% slope, sandy loam soil, grassland, 1 yr study	McCaskey *et al.* (1971)
			3774	8.5	1296	5.2	22.9	125		
			1782	11.0	612	5.0	25.3	120		
	Semi-liquid dairy		2416	8.2	1034	5.0	19.1	84		
			1611	6.5	689	2.2	13.7	60		
			805	8.2	345	2.6	17.7	56		
	Dry dairy		7769	18.3	—	7.6	25.4	107		
			5179	17.7	—	4.6	22.9	165		
			2590	7.5	—	1.4	12.5	48		
Ottawa	Dairy liquid	Winter	224	16.0	58	9.0	—	—	14 plots @ 75.6 × 11.6 m, sandy loam soil, 0.8% slope 1 yr study	Phillips *et al.* (1975)
			560	54.8	149	8.5	—	—		
			897	97.7	235	12.0	—	—		
Minnesota	Dairy	Fall 1972	285	37.6	55.3	7.4			Alfalfa 8 plots @ 4 × 23 m, 9% slope, 3 yr study	Young & Mutchler (1976)
		Fall 1973	265	15.6	48.0	6.7			Alfalfa	
		Spring 1972	206	22.1	30.8	3.7			Alfalfa	
		Spring 1973	371	0.4	62.4	0.1			Alfalfa	
		Fall 1972	285	4.9	55.3	0.6			Corn, manure plowed under	
		Fall 1972	285	5.2	55.3	1.6			Corn, manure on frozen ground	
		Spring 1972	206	5.4	30.8	0.6			Corn, manure on snow	
		Spring 1974	339	1.0	14.8	0.1			Corn, manure on snow	
	Liquid beef	Fall and Spring 1974	558	31.8	24.8	6.2			Alfalfa	
New York	Dairy	Winter	170	0.9‡	42.7	0.3			24 plots @ 61.0 × 53.5 m, continuous corn, 3 yr study	Klausner *et al.* (1976)
			478	18.4	117.3	3.6				
			924	3.1	213.0	1.5				
Alabama	Dairy		800	3.2‡	—	—	15.8	—	2 plots @ 0.04-ha, <2% slope, manure incorporated, 3 yr study	Lund *et al.* (1975)

* Total N = Organic N + NH_4-N + NO_3-N.
† Includes only PO_4.
‡ Includes only inorganic N (NH_4-N + NO_3-N).

Table 114
Linear Relationships Between Nutrient Loading Rates (kg/ha•yr) and Runoff Nutrient Load and Concentrations from Manured Plots
Khaleel et al., 1980

Time of Application	Dependent Variable Y[1]	a constant	b slope	R^2	n number
Winter, Spring	N concentration, mg/l	2.45	0.081	0.70[2]	16
Winter, Spring	N yield, kg/ha•yr	-6.39	0.105	0.86[2]	14
Winter, Spring	P concentration, mg/l	3.27	0.054	0.52[3]	17
Winter, Spring	P yield, kg/ha•yr	1.69	0.064	0.64[2]	14
Summer, Fall	N concentration, mg/l	-22.84	6.6[4]	0.70[2]	20
Summer, Fall	N yield, kg/ha•yr	5.60	0.0016	0.74[2]	20
Summer, Fall	P concentration, mg/l	2.29	0.0071	0.81[2]	22
Summer, Fall	P yield, kg/ha•yr	1.28	0.0028	0.88[2]	21

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[1] Y can be in any of the four units listed, but X is either N or P yield, in kg/ha•yr

[2] Significant at 99.9% level

[3] Significant at 99.0% level

[4] Independent variable for this equation is ln X, where X is N load in kg/ha•yr. In all other regressions X (N or P load in kg/ha•yr) is not log transformed.

Other studies have also demonstrated that substantial nutrient losses can occur with field runoff when manure is applied on frozen or snow covered soil during the winter months (Magdoff et al., 1977; Klausner et al., 1976; and Hensler et al., 1970, Converse et al., 1976). Much of the nutrient loss from manure application on frozen or snow covered ground is associated with applying manure during a thaw (Klausner et al., 1976) or when a thaw and rainfall occur soon after manure application (Hensler et al., 1970).

Hensler et al. (1970) reported that as much as 20% of the nitrogen and 12% of the phosphorus applied to tilled frozen soil was lost, mostly during a single runoff event which occurred a few hours after the manure was applied. These studies point out the water quality benefits to be realized from manure storage during the winter.

Long (1979) found that the runoff quality (in terms of nitrate and ammonium concentrations and yields) from millet and winter rye plots was similar for three fertilizer treatments: (1) incorporation of dairy manure at 807 kg/ha (2) surface-application of the same amount of dairy manure, and (3) surface broadcasting of

450 kg/ha N and 160 kg/ha P chemical fertilizer. Fertilization occurred at the beginning of the growing season and after each of the four clippings.

Mueller et al. (1984b) compared P losses from 3 tillage methods, both with and without surface applied manure, using a rainfall simulator on test corn plots. Manured plots received a surface application of 8 metric tons per hectare (dry weight) of dairy manure prior to tillage. At planting 250 kg/ha of 10-26-26 fertilizer was banded, and in June 125 kg N per ha as ammonium nitrate was side-dressed. (Manure application coupled with regular chemical fertilization is a common practice.) Runoff tests were conducted in late May, mid-July, and early September of the first year, and in early June and late August of the second year.

The manure treatment reduced the total phosphorus losses on the chisel plowed plots (from 92 to 10 g/m^3), but had no effect on the conventionally tilled or no-till plots. Dissolved phosphorus did not show the same response. The manure treatment increased dissolved phosphorus losses from the no-till plot (from 5 to 80 g/m^3), but had no effect on the dissolved phosphorus losses from the other two tillage treatments.

In an often cited article, Beaulac and Reckhow (1982) reviewed the literature on the influence of land use on nutrient export to surface runoff. Table 115 is excerpted from this paper. They concluded that the type of fertilizer (commercial or manure) used is not as important to nutrient flux as the time of application. Also incorporation of fertilizer after application reduces export. Excessive or insufficient fertilization can increase nutrient flux.

C. GROUND WATER POLLUTION FROM LAND APPLICATION OF MANURE

Gerhart (1986) found that on cropland fertilized by manure, recharge can cause an increase or decrease in dissolved nitrate concentration in ground water depending on the amount of manure spread on the site prior to the storm. The study took place in the Conestoga River headwaters in Lancaster County, Pennsylvania, on a 21.7 acre corn and alfalfa field site located on a dairy farm.

The soils at the site were deep, well drained silt loams ranging from less than one inch to more than 1.5 feet deep. The soil is underlain by dolomite of the Cambrian Zooks Corner Formation. Test well depths ranged from 85 to 130 feet. All wells were cased to bedrock (25 to 100 feet).

Direct recharge from a storm in a month in which 0.8 tons/ac of manure (10 kg/ha nitrogen) had been spread, resulted in a rapid decrease in ground water dissolved nitrate concentration of about 2.5 mg/l as nitrogen. Direct recharge from a storm in a month

Table 115
Land Use and Annual Nutrient Flux Relationships:
Total Nutrient Export (Particulate plus Dissolved) in Runoff from Row Crops
Beaulac and Reckhow, 1982

Land Use	Fertilizer kg/ha yr N	Fertilizer kg/ha yr P	Soil	Precipitation cm/yr (range)	Runoff cm/yr (range)	Nitrogen Export kg/ha yr (range)	Phosphorus Export kg/ha yr (range)
Corn[1]	0	0	silt loam	77 (66-78)	11 (8-22)	3.96 (3.61-5.53)	1.22 (1.22-1.49)
Corn[1] fresh manure applied in winter	109	39	silt loam	77 (66-78)	12 (6-19)	7.97 (3.05-26.88)	2.00 (1.03-5.77)
Corn[1] fermented manure applied in spring	102	44	silt loam	77 (66-78)	12 (6-15)	3.38 (3.35-5.32)	0.75 (0.68-0.96)
Corn[1] fermented manure applied in spring	78	33	silt loam	77 (66-78)	12 (6-16)	2.88 (2.81-5.07)	0.95 (0.76-1.18)
Corn[2]	0	0	silt loam		12 (9-14)	4.33 (4.08-4.58)	1.30 (1.00-1.60)
Corn[2] fresh manure applied in winter	108	39	silt loam		9 (7-12)	15.25 (4.44-26.06)	3.40 (1.13-5.66)
Corn[2] fermented manure applied in spring	108	34	silt loam		9 (7-10)	4.22 (3.68-4.76)	0.81 (0.73-0.90)
Corn[2] liquid manure applied in spring	108	39	silt loam		9 (8-11)	3.88 (3.70-4.07)	0.94 (0.91-0.97)
Corn[3]	112	29	loam	63	9	79.6	18.6
Corn[4]	29	81	loam	66	10	44.2	14.0
Corn[4] surface spread manure	268	124	loam	66	4	27.9	8.6
Corn[4] plowdown manure	268	123	loam	66	4	33.0	9.8

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[1] 3 year median, Minshall et al., 1970
[2] 2 year mean, Hensler et al., 1970
[3] 10 year mean, Young and Holt, 1977
[4] 3 year mean, Young and Holt, 1977

after which 17.8 tons/ac of manure (225 kg/ha nitrogen) had been spread resulted in a rapid increase in dissolved nitrate nitrogen of about 3 mg/l.

Changes in concentration occurred both rapidly (2-3 days, lasting for about a week) as well as gradually (the effects are perceived several weeks or more after the storm event), because the recharge entered the ground water through two different routes: direct recharge through near-surface bedrock fractures and sinkholes, and slow recharge through small channels and pores in the unsaturated zone. The study characterized "pre-BMP" conditions. Loads were not estimated.

From 1980 to 1984 the 21.7 acre site was farmed as two corn fields separated by a 4 acre alfalfa field. Manure from about 65 dairy cows and 35 heifers was spread on the corn fields, mainly during the nongrowing seasons. The manure applications from September, 1983 through August, 1984 and ground water response are listed in Table 116.

Table 116
Ground Water Nitrate Levels in Response to Manure Application
Gerhart, 1986

date	tons of manure applied	ground water nitrate level, mg/l maximum	minimum
September, 1983	101		
October, 1983	90		
November, 1983	8		
December, 1983	110		
January, 1984	116		
February, 1984	90		
March, 1984	19	15	11
April, 1984	384	13	12
May, 1984	135	14	13
June, 1984	55	16	13
July-August, 1984	0	17	13

There were 9.3 pounds of nitrogen in each ton of manure. Nitrate concentrations in ground water (35 to 37 feet deep) varied from 11 to 17 mg/l in response to recharge. Ground water levels responded to major storms within several hours due to recharge through direct pathways to the water table such as fractures and sinkholes.

The changes in nitrate levels were observed within two to three days after a storm began. The storm's effects dissipated in about one week. Gradual recharge through small channels and pore spaces in the unsaturated zone reached the water table more slowly and affected dissolved nitrate concentrations for several weeks or more after the storm ended.

The amount of fresh manure on the land surface at the time of the storm determined whether nitrate concentrations increased or decreased, and determined the magnitude of the increase or decrease.

Hubbard et al. (1987) studied shallow ground water pollution from fields irrigated with dairy manure slurry at two rates, 530 and 1083 kg/ha•yr nitrogen (reporting only 1984 figures of the three year study). A third area received 464 kg/ha•yr inorganic N fertilizer. The areas were planted to coastal bermudagrass during the summers and ryegrass during the winters.

The study site was located on plinthic soils of the Georgia coastal plain. These soils contain horizons of low permeability ranging in depth from 75 to 200 cm below the soil surface. These horizons tend to restrict downward percolation of water.

Surface runoff nitrogen losses were 67.5, 13.0, and 2.0 kg/ha•yr for the high rate, low rate, and chemical fertilizer treatments.

Ground water nitrate was sampled at 3 depths, 0.9 to 1.2 m, 2.4 m, and 3.6 meters. Mean monthly nitrate N concentrations at the 1 m depth ranged from 10 to 70 mg/l, averaging about 43 mg/l. No significant differences in shallow ground water nitrate concentrations were found with both waste water application rates.

The range of monthly mean nitrate-N concentrations from the 2.4 m piezometers ranged from 10 to 50 mg/l, averaging 36 and 28 mg/l for the low and high application rates, respectively.

Monthly nitrate-N concentrations from the 3.6 m depth ranged from 5 to 35 mg/l, and averaged 16 mg/l, demonstrating the vertical movement of nitrate through the flow restricting layers into the water table.

C. RUNOFF POLLUTION FROM FEEDLOTS AND LOAFING LOTS

1. HYDROLOGY

Overcash and Phillips (1978) estimated that the annual runoff from dairy lots was 30 to 40% of precipitation for dirt lots and 80 to 95% for concrete lots, with no runoff occurring until 0.87 cm of rainfall had fallen.

The HSPF parameter, UZSN, upper zone storage, which is the sum of depression storage and pore storage in the upper zone, may be estimated as about 1 cm for feedlots based on this observation.

2. CHEMISTRY

Beaulac and Reckhow (1980) estimated that from 2 to 10 percent of manure generated in feedlots is lost to surface runoff. Their literature review showed that the total nutrient export from feedlot and unprotected manure stacks is two to three orders of magnitude greater than nutrient export in runoff draining other agricultural activities. Nutrient export variability is also much higher.

Westerman and Overcash (1980) measured chemical constituents of dairy open lot runoff. Dairy open lots are used primarily for a holding or loafing and exercise area. The open lot is usually not vegetated or has a concrete surface. Lots were stocked at about 100 m^2/cow. (Normal stocking rates in feedlots range from 10 to 50 m^2 per animal (Clark et al., 1975).) Cows spent 8 to 12 hours in the lot depending on the season.

The authors found significant relationships between mass of contaminant transported per runoff event and total runoff:

$$\text{LOAD (kg/ha)} = \text{Intercept} + \text{Slope} * \text{Runoff per Event (cm)}$$

They found that in general 12% of the applied P appeared in runoff. They also observed that high intensity storms were responsible for most pollutant transport even though these storms were responsible for only 17% of the total precipitation.

Table 117
Linear Regression Coefficients for Chemical Load on Dairy Open Lot Runoff
Westerman and Overcash, 1980

Parameter kg/ha	Number of Events	Intercept	Slope	R^2
COD	32	11.68	102.2	0.77
TOC	31	2.10	37.3	0.88
TKN	32	0.92	5.23	0.83
TP	31	0.18	2.85	0.94
Cl	31	2.92	9.37	0.66

Westerman and Overcash (1980) also reviewed other studies of runoff pollution from open lots (Table 118).

Table 118
Runoff Pollutant Concentrations for Beef and Dairy Open Lots
Westerman and Overcash, 1980

(all concentrations as mg/l)

Type	Total Solids	COD	TKN	TP	NO_3-N	Reference
Dairy	2800-8400	1021-4846	33-411	-	-	Gershon et al., 1975
Dairy	6510	4910	270	430 (soluble)		Thormodsgard, 1972
Dairy	-	585-1820	39-142	22-47	0.01-34.3	Westerman and Overcash, 1980
Beef	7500-17500	2100-17800	90-1750	50-300	-	Clark et al., 1975 and Wise, 1972

Khaleel et al. (1979b) summarize several studies of nutrient concentration from feedlot runoff (Table 119). Snowmelt runoff had higher concentrations than rain water runoff.

The range of concentrations of runoff pollutants from feedlots is wide. Snowmelt runoff resulting from winter thawing conditions produced higher concentrations of pollutants than those produced by rainfall runoff (by a factor of 2 to 3).

In a literature review, Loehr (1974) reported runoff losses from feedlots ranging from 100-1600 kg/ha•yr for total N and 10-620 kg/ha•yr for total P. These ranges are considerably higher than those reported for manured fields (Table 113) which were 1-98 kg/ha•yr for total N and 1-12 kg/ha•yr for total P.

Kreis et al. (1972) reported chemical analyses of rainfall runoff from a beef feedlot (Table 129). The runoff was held in detention ponds with several days detention time which were connected to 3.6 km of grassed waterway which led to a reservoir.

Detention in holding ponds reduced the total amount of solids and organic pollutants by about one half. The total amounts of nutrients such as ammonia, nitrate, and phosphate were not significantly reduced. There was a reduction in the total amount of nitrogen, from nitrate reduction (Table 121). Interestingly, though total amounts were unchanged, concentrations of dissolved

Table 119
Concentrations of Pollutants in Feedlot Runoff
Khaleel et al., 1979b

Type of Runoff	Total N, mg/l		Total P, mg/l		BOD	COD, mg/l		Remarks
	Range	Average	Range	Average	Average	Range	Average	
Rainfall	85-1580	583	9-482	83		860-16100	6855	Average of results from two sites: 0.82 ha with 1% slope and 11.1 ha on 1.3% slope. Stocking rate: 28 m²/animal. 2yrs. Manges et al., 1975
Snowmelt	590-2340	1240	65-459	190		7300-35760	14915	
Rainfall	11-8593	854	2-1425	151		1300-8200	3100	0.01 ha on 3%., 6%, and 9% slopes.
Snowmelt	190-6528	2105	5-917	292		14100-77100	41000	Stocking rate: 19 m²/animal. 2 yr study. Gilbertson et al., 1975
Rain & Snow		277		184	942		4093	6 sites, 2 to 15% slopes. Stocking rate:52 m²/animal. Snowmelt was 30% annual runoff. 2 yrs. Madden and Dornbush, 1971
Rainfall	31-493	228	21-223[1]	69	2201	1439-16320	7210	13 ha on 13% slope. Stocking rate:9 m²/animal. 2 yrs. Kreis et al., 1972
Rainfall	4-125	50	5-305	85		500-14000	4000	7.5 ha on 3% slope. Stocking rate:35 m²/animal. 1 yr. Wise & Reddell, 1973
Rainfall	600-2400	1083	100-500	205		10000-20000	15700	4 ha on 1.5% slope. Stocking rate:12 m²/animal. 1 yr. Clark et al., 1974
Rain & Snow		1153		93			17800	0.4 ha on 6% slope. Stocking rate:19 m²/animal, 3 yrs. Clark et al., 1975

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[1] PO_4 only

solids decreased by 90% and organic pollutant concentrations by 70% (Table 122). The decrease in concentration was attributed to dilution by clean runoff water from another tributary of the pond.

Once the solids were reduced in the holding ponds, pumping the effluent from the ponds through the 2-mile long grassed waterway had no significant effect on the waste water quality.

Table 120
Chemical Constituents in Direct Runoff From Beef Feedlots
Kreis et al., 1972

Parameter (mg/l)	No. of Samples	Mean	Minimum	Maximum
Total Solids	8	11429	3110	28882
Total Suspended Solids	8	5912	745	17202
Volatile Suspended Solids	7	3426	475	9286
Total Dissolved Solids	8	5526	882	22372
Chloride	7	450	97	648
Total Phosphate Phosphorus	16	69	21	223
Nitrate Nitrogen	15	0.64	<0.05	2.3
Ammonia Nitrogen	15	108	4	173
Total Organic Nitrogen	15	228	31	493
Carbonaceous Oxygen Demand	15	7210	1439	16320
5-Day Biochemical Oxygen Demand	4	2201	1075	3450
Total Organic Carbon	15	2010	150	4400
Calcium	6	698	194	1619
Magnesium	6	69	28	89
Sodium	6	408	130	655
Potassium	6	761	226	1352

Table 121
Effect of Detention Ponds on Feedpen Runoff
Kreis et al., 1972

Parameter	Feedpen Runoff metric tons	Pond Effluent metric tons
Total Solids	390	202
Total Suspended Solids	202	51.4
Total Dissolved Solids	189	147
Total Phosphate Phosphorus	2.36	2.63
Nitrate Nitrogen	0.0165	0.165
Ammonia Nitrogen	3.69	3.99
Total Organic Nitrogen-N	7.81	4.36
Carbonaceous Oxygen Demand	246	138
5-Day BOD	75.3	40.8
Total Organic Carbon	69.0	49.9

Table 122
Changes in Parameter Concentration of Feedlot Runoff at Different Treatment Stages
Kreis et al., 1972

Parameter mg/l	Feedpen Runoff	Pond Effluent	Ditch Effluent	Reservoir Effluent
Total Phosphate Phosphorus				
mean	69	37	38	26
maximum	223	45	65	40
minimum	21	29	21	5
no. of samples	16	6	15	11
Total Organic Nitrogen				
mean	228	62	64	39
maximum	493	136	142	80
minimum	31	36	38	11
no. of samples	15	14	15	11
Ammonium Nitrogen				
mean	108	63	50	35
maximum	173	112	70	58
minimum	4	45	30	10
no. of samples	15	14	15	10
Nitrate Nitrogen				
mean	0.64	0.21	0.20	0.22
maximum	2.3	0.50	0.90	0.36
minimum	<0.05	0.09	<0.05	<0.05
no. of samples	15	14	14	10

B. GROUND WATER POLLUTION FROM FEEDLOTS

Norstadt and Duke (1985) investigated the changes in salt and nitrogen movements beneath feedlots that resulted from changing from a manure plus sawdust manure pack to a manure pack without bedding material.

They installed lysimeters below the feedlot and took soil cores for chemical analysis after two and a half years under the sawdust and manure pack, then again two and a half years after the pack had been changed to pure manure. The lysimeters were installed at two depths, 0.5 and 1.3 m. The lysimeter pits were filled with four different soil combinations to test the effect of soil texture on nutrient movement. The soil fill sequences were: all clay loam, all sand, clay loam over sand, and sand over clay loam.

Table 123
Soil Chemistry Characteristics Beneath a Beef Feedlot
Norstadt and Duke, 1985

	water content	pH	Conductivity	Ammonium-N		Nitrite-N		Nitrate-N		Total Mineral-N		Phosphate-P	
				soil sol'n.	dry soil	soil sol'n.	dry soil	soil sol'n.	dry soil	soil sol'n.	dry soil	soil sol'n.	dry soil
	g/kg		dS/m	mg/kg	mg/kg	mg/kg	mg/kg	mg/kg	mg/kg	mg/kg	mg/kg	mg/kg	mg/kg
Lysimeter #1, Clay Over Clay Fill, no bedding material													
Upper fill	146	8.0	6.7	390	54	22	2.8	310	42	722	99	7	1.3
Lower fill	186	8.2	1.2	10	1	1	0.1	60	11	71	12	0	0
Lysimeter #1, Clay Over Clay Fill, with sawdust bedding													
Upper fill	157	8.0	4.3	340	49	48	6.8	360	52	748	108	3	0.3
Lower fill	205	8.3	0.7	10	2	0	0	50	10	60	11	2	0.1
Lysimeter #2, Sand Over Sand Fill, no bedding material													
Upper fill	23	8.7	3.9	1770	30	220	5.0	290	7	2280	42	33	0.5
Lower fill	26	8.6	1.9	240	5	24	0.6	570	14	834	20	4	0.1
Lysimeter #2, Sand Over Sand Fill, with sawdust bedding													
Upper Fill	43	8.4	6.1	2650	139	175	4.4	910	24	3735	167	361	25.1
Lower Fill	34	7.6	1.8	370	12	39	1.3	1090	33	1499	46	26	0.8
Lysimeter #3, Clay Over Sand Fill, no bedding material													
Upper Fill	149	7.7	7.4	420	62	12	1.9	440	67	872	131	4	0.6
Lower Fill	14	8.8	1.1	50	1	1	0	220	3	271	4	9	0.1
Lysimeter #3, Clay Over Sand Fill, with sawdust bedding													
Upper Fill	160	7.7	4.8	380	66	11	1.9	290	46	681	114	3	0.2
Lower Fill	19	8.2	0.8	70	1	2	0	130	2	202	3	16	0.2
Lysimeter #4, Sand Over Clay Fill, no bedding material													
Upper Fill	26	8.7	3.9	1030	17	513	11.0	200	4	1743	32	63	1.3
Lower Fill	191	8.1	2.0	30	6	7	1.3	60	4	97	11	0	0
Lysimeter #4, Sand Over Clay Fill, with sawdust bedding													
Upper Fill	84	8.2	5.2	1370	57	229	7.2	470	19	2069	83	280	19.5
Lower Fill	231	7.7	1.2	50	11	1	0.1	20	4	71	15	1	0.2

Animals were stocked at a rate of one animal, weighing 270 to 450 kg, per 15.2 m^2, or about 6.6 times the rate proposed as minimal for an effective manure pack. Table 123 presents the soil chemistry data. These data are included to to give an idea of initial storages of nutrients below feedlots.

VII. MANURE AMENDED SOIL CHEMISTRY

A. VERTICAL NUTRIENT DISTRIBUTION

Stewart et al., 1967 studied the nitrate concentrations in soil cores taken from several land uses in Colorado. Cores from 47 corral, 28 irrigated fields, and 21 dryland cropped fields were collected from May and July. Average nitrate concentrations are presented in Table 124.

The amount of nitrate N under corrals was variable. The totals ranged from nearly none to more than 5600 kg/ha in a 20 foot profile.

Table 124
Soil Nitrate Concentrations Below Corrals and Fields[1]
Stewart et al., 1967

Depth m	Dryland Fields ppm NO_3-N	Irrigated Fields ppm NO_3-N	Corrals ppm NO_3-N
0.15	10.0	65.0	
0.46	11.0	16.4	70.0
0.76	2.7	7.2	27.9
1.07	2.3	6.4	30.5
1.37	1.4	4.5	23.1
1.68	1.5	3.8	22.3
1.98	4.4	4.4	22.1
2.29	4.6	2.3	22.8
2.59	5.5	2.1	14.1
2.90	3.8	2.1	14.9
3.20	1.8	2.1	17.1
3.51	2.3	2.7	13.3
3.81	2.1	2.7	11.3
4.11	1.9	2.3	6.2
4.42	2.4	3.6	5.4
4.72	1.8	2.4	5.4
5.03	1.4	2.8	6.2
5.35	1.4	1.9	5.3
5.64	1.5	3.1	3.5
5.94	0.6	2.6	3.8

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[1] Read from a graph

Reddy et al., 1980 studied the build up of phosphorus in soils from repeated manure applications. The rate of manure application is important. Increasing application rate increased soluble P, acid extractable P, P desorbed, and equilibrium phosphorus concentrations, and decreased the phosphorus adsorption capacity of the soil. At the highest loading rate tested the maximum depth of phosphorus migration also increased.

They had test sites on two soils: (1) a coastal plain soil in the Norfolk series, a deep, well drained, loamy sand, a member of the fine loamy siliceous, thermic Typic Paleudult family; and (2) a piedmont soil of the Cecil series, a sandy loam member of the clayey, kaolinitic, thermic family of Typic Hapludults. The bulk density of the Norfolk soil was about 1.65 g/cm^3 and its porosity was 0.38. The bulk density of the Cecil soil was about 1.53 g/cm^3 and the total porosity was 0.42. Fescue grass was grown on the Cecil plots and a coastal bermudagrass grown on the Norfolk plots.

The change in soluble P with depth and phosphorus loading rate is presented in Table 125 (taken from graphs). Soluble P is that dissolved in 0.01 M $CaCl_2$. Manure was applied at three rates for 5 years on the Norfolk plots and for 3 years on the Cecil plots.

Other results from this paper are relevant to selecting parameter values for the PHOS subroutine of HSPF dealing with absorption and desorption of phosphorus in soil. Table 126 lists soil characteristics by depth including the amount of phosphorus desorbed by treatment with 0.01 M $CaCl_2$ for one hour, the amount of phosphorus extracted from soil during 5 minutes in 0.05N HCL plus 0.025N H_2SO_4, and the pH in the 0.01M $CaCl_2$ soil solution.

Tables 127 and 128 present phosphorus adsorption measurements and calculations for equilibrium conditions. The experimental method for obtaining these parameters was as follows: 5 g of soil mixed with 100 ml of phosphate solution (in 0.01M $CaCl_2$) of known initial concentration of 0, 5, 10, 20, and 50 μg/ml and toluene, were equilibrated under continuous shaking for 18 hrs. After incubation, the supernatant liquid was filtered and analyzed for P remaining in solution. The equilibrium phosphorus concentration is the concentration of phosphorus buffer such P is neither gained or lost by the solid. Phosphorus adsorption parameters were determined using the Langmuir isotherm equation:

$$C/S = C/S_{max} + 1/KS_{max}$$

where C is the final solution concentration and C/S is the quantity of P adsorbed or desorbed per unit mass of adsorbent. S_{max} the adsorption maximum, was obtained from the regression of C/S on C. The adsorption maximum was also empirically determined (for comparison) by incubating soil at 1,000 μg P/g.

Table 125
Soluble Phosphorus and Extractable Phosphorus with Depth as Influenced by Loading Rate of Swine Lagoon Effluent
Reddy et al., 1980

Loading Rate kg/ha·yr	depth, cm 0-15	15-30	30-45	45-60	60-75	75-90
Norfolk Soil Series (Coastal Plain)						
	Soluble Phosphorus, µg/g of soil					
0	0.24	0.00	0.00	0.00	0.00	
81	2.45	0.22	0.16	0.16	0.08	
161	3.47	1.92	0.24	0.08	0.08	
322	11.84	6.24	2.56	0.21	0.21	
	Extractable Phosphorus, µg/g of soil					
0	30.0	5.2	2.0	0	0	0
81	91.0	16.6	10.0	4.0	2.8	0.6
161	131.4	70.4	8.8	4.0	3.2	1.8
322	151.0	120.0	45.0	25.6	21.0	4.0
Cecil Soil Series (Piedmont)						
	Soluble Phosphorus, µg/g of soil					
0	0.14	0				
31[1]	0.16	0				
161	0.56	0.08				
322	3.79	0.22				
	Extractable Phosphorus, µg/g of soil					
0	26.0	10.0	2.0			
31[1]	10.5	9.0	2.2			
161	45.4	5.0	2.3			
322	89.0	17.4	5.2			

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[1] inorganic P fertilizer used on this plot, not swine lagoon effluent

Table 126
Selected Soil Characteristics Relating to Phosphorus Adsorption For Two North Carolina Soils
Reddy et al., 1985

Soil Depth cm	% Clay	% Silt	% Sand	Desorbed P[1]	Extractable P[2]	pH of $CaCl_2$
				µg P per g of soil		
			Norfolk soil			
0-15	3.2	11.8	85.0	0.20	43.5	4.2
15-30	4.3	9.7	86.0	0.10	5.5	4.1
30-45	3.1	9.4	87.5	0.08	1.5	4.0
45-60	3.7	10.0	86.3	0.08	0.5	4.0
60-75	7.9	8.7	83.4	0.01	0.2	3.9
75-90	28.9	6.0	65.8	0	0.2	3.9
90-105	21.9	6.5	71.0	0	0.1	3.7
			Cecil soil			
0-15	16.1	16.5	67.4	0.15	18.5	4.9
15-30	27.6	20.4	52.0	0.05	11.0	4.7
30-45	48.2	16.9	34.9	0.05	1.5	4.6
45-60	58.2	15.5	25.8	0.02	0.2	4.5
60-75	56.7	14.5	28.8	0.01	0.1	3.9
75-90	42.5	15.7	41.8	0	0.1	3.9
90-105	33.1	18.8	48.1	0	0.1	3.8

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[1] by 1 hr in 0.01M $CaCl_2$
[2] Acid extracted for 5 minutes in 0.05N HCl plus 0.025N H_2SO_4, 1:10 soil to solution ratio

Adsorption capacity of the soil increased with increasing depth due to the higher clay content of the subsoil. The adsorption capacity of the Norfolk soil was low to a depth of 75 cm, which explains the deeper P movement in this soil than in the Cecil soil.

The conclusions of this study were that application of animal wastes increased soluble P, acid extractable P,and equilibrium P concentration values, and decreased the P sorption capacity of the soil. These parameters were directly related to the loading rates of animal wastes. The decrease in P sorption capacity may result in more soluble P moving into deeper soil layers and ground water, depending on the soil.

Table 127
Phosphorus Adsorption by a Norfolk Soil Receiving Swine Lagoon Effluent for a Period of 5 Years
Reddy et al., 1985

Depth cm	Adsorption Max Calculated µg/g	Adsorption Max Experimental µg/g	Equilibrium P Concentration µg/ml	pH in 0.01M $CaCl_2$
Norfolk Soil				
Loading Rate: 322 kg P/ha per year				
0-15	--	18	22.0	3.8
15-30	67	49	5.30	3.9
30-45	118	114	0.46	4.1
45-60	138	129	0.29	4.6
60-75	169	162	0.09	4.0
75-90	872	656	0.05	3.6
90-105	877	829	0.02	3.4
Loading Rate: 161 kg P/ha per year				
0-15	71	55	4.10	4.0
15-30	84	75	0.74	4.1
30-45	128	124	0.06	4.2
45-60	123	120	0.02	4.2
60-75	154	151	0.01	4.5
75-90	895	586	--	3.9
90-105	1176	897	0.02	3.5
Loading Rate: 81 kg P/ha per year				
0-15	90	58	1.65	4.0
15-30	125	117	0.12	3.9
30-45	81	81	0.01	4.2
45-60	103	102	0.46	4.0
60-75	224	221	0.01	4.2
75-90	652	567	--	3.8
90-105	1058	883	0.03	3.3

Table 128
Phosphorus Adsorption by a Cecil Soil Receiving Swine Lagoon Effluent for a Period of 3 Years
Reddy et al., 1985

Depth cm	Adsorption Max Calculated μg/g	Adsorption Max Experimental μg/g	Equilibrium P Concentration μg/ml	pH in 0.01M $CaCl_2$
		Loading Rate: 322 kg P/ha per year		
0-15	71	76	0.88	5.4
15-30	482	451	0.01	5.7
30-45	841	825	0.01	5.0
45-60	--	--	--	4.3
60-75	1081	938	--	3.8
75-90	--	--	--	3.6
90-105	940	876	--	3.4
		Loading Rate: 161 kg P/ha per year		
0-15	209	190	0.16	5.4
15-30	896	742	0.01	5.3
30-45	951	865	--	5.0
45-60	--	--	--	4.4
60-75	1192	952	--	3.9
75-90	--	--	--	3.7
90-105	893	875	--	3.3
	Loading Rate: 31 kg P/ha per year (inorganic fertilizer P)			
0-15	234	220	0.03	4.7
15-30	887	762	0.01	4.7
30-45	480	475	--	4.6
45-60	--	--	--	4.8
60-75	950	895	--	3.9
75-90	--	--	--	3.6
90-105	743	814	--	3.2

B. MINERALIZATION OF N IN MANURE AMENDED SOIL

Reddy et al. (1979) formulated a model of organic nitrogen mineralization in manure amended soils. Their experimental measurements of the first order rate coefficients for this process are listed in Table 129. "k" here is the aggregate rate parameter which describes the accumulation of nitrate as the sum of nitrate from organic N mineralization from soil and from waste.

Table 129
First Order Rate Constants for Nitrate Accumulation in Soils Treated with Animal Waste
Reddy et al., 1979

Type of Waste	Temperature °C	k Measured 1/day	k at 30°C 1/day	Half Life[1] at 30°C days
Beef feedlot waste[2]	28	0.0046	0.0053	131
Beef waste (fresh)[3]	22	0.0048	0.0082	85
Liquid dairy waste[4]	20	0.0275	0.0541	13
	20	0.0047	0.0092	75
Poultry waste[5]	15.6	0.0126	0.0334	21
Poultry waste (fresh)[6]				
Waste surface applied.	22	0.0132	0.0227	31
Waste incorporated.	22	0.0142	0.0244	28
Pig slurry[7]	30	0.0495	0.0495	14
Swine waste (fresh)[8]				
Waste surface applied.	22	0.0081	0.0139	50
Waste incorporated.	22	0.0085	0.0146	47
Sewage sludge[9]	--	0.0043	--	--
Sewage sludge[10]	22	0.0257	0.0442	16
Clover straw residue[11]	30	0.0109	0.0109	64
Clover + rice straw[12]	30	0.0059	0.0059	117
Rice + clover straw[13]	30	0.0026	0.0026	267
Soil only[14]	30	0.0016	0.0016	433
Soil[15]	35	0.0077	0.0055	126

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[1] Time after which 50% of the initially available organic N was converted to nitrate.

[2] Pullman silty clay loam. Laboratory study. Waste was incorporated. 90-day incubation period. Mathers and Stewart (1970)

[3] Norfolk loamy sand. A laboratory study. Surface applied. 120-day incubation. Walter et al. (1970)

[4] Plainfield sand. Dairy waste containing about 38% of total N as ammonium. First rate constant represents nitrification, and second rate constant represents mineralization. Walter et al.(1974)

[5] Ammonification in the manure only (no soil). Hashimoto (1974)

[6] Norfolk loamy sand. A laboratory study. 120-day incubation period. Reddy et al. (1977)

[7] Acid loam soil. Laboratory study. Pig slurry containing mostly ammonium N. Cooper (1975)

[8] Norfolk loamy sand. A laboratory study. 120-day incubation period. Reddy et al. (1977)

[9] Soil samples obtained from field plots treated with sewage sludge and measured mineralization under laboratory conditions. Stark and Clapp (1977)

[10] Freehold sandy loam. Laboratory study. Waste incorporated. 54 day incubation period. Yuch Ping Hsieh (1976)

[11] Crowley silt loam, laboratory study, 120-day incubation period. Residue incorporated. C/N ratio of the residue 15. Tusneem (1970)

[12] C/N ratio of the residue 21. Ibid.

[13] C/N ratio of the residue 35. Ibid.

[14] Crowley silt loam. Ibid.

[15] Average value of 39 soils collected across the United States. Stanford and Smith (1972).

The N transformation model is based on the following assumptions: (a) nitrate N accumulation in the soil-waste system follows first order kinetics, (b) ammonification rates are slower than nitrification rates resulting in no appreciable accumulation of ammonium, (c) the nitrogen present in the waste in excess of microbial requirements[11] is potentially mineralizable, and (d) the soil and waste residual N pool is mineralizable.

Poultry and swine wastes decompose more rapidly (with a shorter half life) followed by wastes containing more fibrous material, such as beef wastes.

The aggregate mineralization rate constants for animal wastes were found to be in the range of 0.01 to 0.03 per day with the rates for waste containing high fractions of ammonia being in the range of 0.03 to 0.05 per day (corrected to 30°C). Beef feedlot waste, having been stabilized, had a lower degradation rate, 0.005 per day. Mineralization for soil organic N and residues are in the range of 0.002 to 0.006 per day (30°C) or about 5 to 10 times lower than applied wastes. Under field conditions these half life values may be longer, because of fluctuations in temperature and moisture conditions.

The rate of organic N mineralization in the soil-waste system is influenced by several factors. These, according to Reddy et al. (1979) are: (a) soil and environmental factors, which include temperature, pH, moisture content, available nutrients in the soil, soil type (texture and structure), and (b) additional factors such as chemical composition of the applied waste, its carbon-nitrogen ratio, lignin content, and oxygen demand, ammonium toxicity, loading rate, time and method of application, and particle size. The model developed by Reddy et al. accounts for the better understood and most important variables: temperature, soil moisture, pH, and method of application.

The rate of mineralization (organic N to nitrate) increases with temperature, reaching a maximum between 40 and 60°C, and essentially ceases when the soil is frozen. The conversion of ammonium N to nitrate N stops at 45 °C.

The equation for adjusting k for temperature is:

$$k_T = k_S \theta \exp(T_T - T_S) \qquad [11]$$

where k_T is the temperature corrected rate constant, k_S is the constant at some standard temperature, θ is the temperature correction coefficient, and T_T and T_S are the desired corrected

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[11] approximated by the point when the C:N ratio of the waste reaches 10, the average C:N of microbial protoplasm and humus

temperature and the standard temperature, respectively.

Stanford et al. (1973) concluded that the rate of mineralization process doubled with a temperature increase of 10°C, which yields a temperature correction coefficient, θ, of 1.07. Walter et al. (1974) have shown a similar relationship for soil treated with dairy waste, with θ of 1.13 for the equation [11] when soil temperature is between 0 and 35°C. Other temperature correction coefficients, θ, of interest are listed in Table 130.

Table 130
Temperature Correction Coefficients for Nitrate Accumulation in Soil
Reddy et al., 1979

Description of the Experiment	Temperature Range, °C	Θ	Reference
1. Mineralization of soil organic N	5-15	1.08	Stanford et al. (1974)
	15-25	1.07	
	25-35	1.07	
2. Soil treated with dairy waste	9-20	1.13	Walter et al. (1974)
3. Nitrification of ammonium N	6-13	1.05	Endelman et al. (1974)
	13-20	1.04	
4. Mineralization of organic N	7-24	1.07	Broadbent (1966)
	24-32	1.04	
5. Nitrification of ammonium N	10-25	1.06	Gerretsen (1942)
6. Nitrification of ammonium N	7-27	1.05	Frederick (1956)
7. Nitrification of ammonium N	10-25	1.08	Sabey et al. (1969)
	average θ	1.07 ± 0.03	

The rate constant, k, is also influenced by soil moisture and pH and the method of waste application. Reddy et al. (1979) give multiplicative correction factors which are applied thus:

$$k = k_T * F_m * F_{pH} * F_A \cdot \theta^{(T_T - T_S)} \qquad [12]$$

where k is the corrected first order rate constant, k_T is as defined in equation [11], F_m, F_{pH} and F_A are the soil moisture rate correction factor, the pH correction factor, and the factor for application method. Values for these factors are found in Tables 131 through 133.

Mineralization may occur at a slower rate for moisture tensions of 10 to 15 bars and may cease completely at moisture tensions greater than 15 bars. Organic N is mineralized at moderate rates at excessively high moisture conditions. At such conditions oxygen depletion may occur at the waste incorporation site. N mineralization will slow and nitrification of ammonium will cease.

Table 131
Soil Moisture Correction Factor Values for N Mineralization Rate
Reddy et al., 1979

Mineralization of Organic N (Miller and Johnson, 1965)

$F_M = 1.51 + 0.453 \ln MT$ $\quad 0.07 \leq MT \leq 0.33$

$= 1.00$ $\quad 0.33 \leq MT \leq 0.85$

$= 0.97 - 0.118 \ln MT$ $\quad 0.85 \leq MT \leq 10.0$

Nitrification of Ammonium N (Sabey, 1969)

$F_M = 0.599 - 0.173 \ln MT$ $\quad 0.10 \leq MT \leq 15.0$

where FM is the relative rate, [0,1], of nitrate accumulation and MT is the moisture tension, bars.

Assuming all other factors are favorable, the production of inorganic N will be greater in neutral than in acid soils. Waste applications temporarily increase the pH of soil. The optimum pH range may be between 6.5 and 8.5. Conversion of organic N to ammonium is not markedly influenced by pH, but the nitrification process (conversion of ammonium to nitrate) is relatively sensitive to pH.

Table 132
Acidity Correction Factor Values for N Mineralization Rate
Reddy et al., 1979

Mineralization of Organic N (Reddy et al., 1979a)

$F_{pH} = 0.19\ pH - 0.173$ $\quad 4.0 \leq pH \leq 6.2$

$= 1.00$ $\quad 6.2 \leq pH \leq 9.0$

$= 3.25 - 0.25\ pH$ $\quad 9.0 \leq pH \leq 10.0$

Nitrification of Ammonium N (Reddy et al., 1979a)

$F_{pH} = 0.307\ pH - 1.269$ $\quad 4.5 \leq pH \leq 7.0$

$= 1.00$ $\quad 7.0 \leq pH \leq 7.4$

$= 5.367 - 0.599\ pH$ $\quad 7.4 \leq pH \leq 9.0$

An increased rate of mineralization was observed when wastes are incorporated, as compared to surface application.

Table 133
Application Mode Correction Factor Values for N Mineralization Rate
Reddy et al., 1979

Type of waste or residue	Method of Application	F_A	Reference
1. Beef Waste	Incorporated	1.00	Reddy et al. (1977)
	Surface Applied	0.70	
2. Poultry Waste	Incorporated	1.00	
	Surface Applied	0.93	
3. Swine Waste	Incorporated	1.00	
	Surface Applied	1.05	
4. Corn Stalks	Incorporated	1.00	Parker (1962)
	Surface Applied	0.51	
5. Liquid Sludge	Incorporated	1.00	King (1973)
	Surface Applied	0.53	

Reddy et al. (1979a) also provide a method for estimating the amount of potentially mineralizable N from waste applied to soil for the first order nitrate accumulation equation. There are two sources of this N, the waste and the soil. The N from the waste is modeled by

$$NX = NW - 0.043 * NW * C{:}N_W \qquad [13]$$

where NX is the amount of mineralizable N (positive or negative) in the waste, NW is the nitrogen content of the waste, and $C{:}N_W$ is the carbon nitrogen ratio of the waste. Selected $C{:}N_W$ values are found in Table 134.

The other source of mineralizable N is the soil itself, which can be estimated from the following relationship (Stanford and Smith, 1972)

$$NS = -591 + 112 \ln(TN) \qquad [14]$$

where NS is the potentially mineralizable N present in the soil ($\mu g/g$ of soil), and TN is the total organic nitrogen in the soil from organic N present before waste application, $(TON)_S$, and residual organic N from previous waste applications, WON = NW - NX.

$$TN = (TON)_S + WON/(1.5 \cdot BD)) \qquad [15]$$

where BD is the bulk density of the soil in g/cm^3 and the factor $1.5 \cdot BD$ converts kg/ha to $\mu g/g$ of soil, assuming 15 cm soil depth.

Examples of estimates of NX and NS obtained by applying equations [13] and [14] to data in the literature are found in Table 135.

Table 134
Nitrogen and Carbon Content of Waste, Dry Matter Basis
Reddy et al., 1979a

Type of Waste	%N	%C	C:N	Remarks
Cattle feedlot waste	2.6	36	14.1	Average of 5 reports.[3]
Beef feedlot waste	1.9	29	15.2	Average of 8 samples.[4]
Dairy waste	3.9	41	10.5	Average of 17 reports.[3]
Poultry waste	6.2	31	5.0	Average of 17 reports.[3]
Swine waste	7.8[1]	39[2]	5.0	Average of 17 reports.[5]
Sheep waste	4.0	30	7.6	Average of 2 reports.[3]
Anaerobic sewage sludge	5.0	27.6	5.5	Average of 35 reports.[6]
Aerobic sewage sludge	4.9	31.7	6.5	Average of 10 reports.[6]
Rice straw residue	0.43	40	93.0	Tusneem (1970)
Corn residue	1.15	39.6	34.4	Banwart and Bremmer (1976)
Alfalfa	4.04	40.8	10.1	Ibid.
Orchard grass	2.92	38.5	12.3	Ibid.

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[1] range 3.5 - 11.3
[2] range 30 - 43
[3] Crane (1978)
[4] Ruehr (1976)
[5] Overcash (1976)
[6] Sommers (1977)

Table 135
Nitrogen Potentially Available for Nitrate Accumulation in Soils Treated with Wastes
Reddy et. al, 1979a

Type of Waste	Nitrogen Content			C:N Ratio	Potentially Mineralizable N		Reference
	Total N	Organic N	Ammonium N		From Waste, NX	From Soil, NS	
	% dry wt.	% Total N	% Total N		% N applied	% Total Soil org-N	
Beef feedlot waste	3.25	100.0	0.0	9.8	57.5	16.8	Mathers and Stewart, 1970
Fresh beef waste	2.45	94.2	5.8	19.9	7.9	19.5	Reddy et al., 1977
Liquid dairy waste	--	62.0	38.0	8.3	63.9	--	Walter et al., 1974
Poultry waste	--	85.9	14.1	5.4	76.5	--	Hashimoto, 1974
Poultry waste	6.03	50.8	49.2	3.7	84.0	17.9	Reddy et al., 1977
Swine waste	3.97	85.1	14.9	11.2	63.8	20.4	Reddy et al., 1977
Sewage sludge	4.90	95.9	4.1	--	29.0[1]	5.5	Yuch Ping Hsieh, 1976
Sewage sludge	3.08	91.9	8.1	--	38.0[1]	5.5	Yuch Ping Hsieh, 1976
Clover Straw	3.07	100.0	0.0	15.0	34.8	18.7	Tusneem, 1970

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[1] Determined from the experiments

The incorporation of wastes with C:Ns greater than 23, such as the plant residues plowed down at the end of harvest, requires another modification to the modeling approach. Before nitrate can accumulate, the C:N must be reduced to (approximately) 23 by carbon mineralization. This is achieved by lagging the beginning of nitrate appearance by t_D days, the amount of time it takes for the C:N ratio to reach 23. This is found by

$$t_D = \{\ln C{:}N_0 - \ln 23\}/\{ A * \theta * \exp(T_T - T_S) * F_M * F_{pH} * F_A \} \qquad [16]$$

Values of A for barley straw and corn stalk residue have been found to be 0.104 and 0.0082 day^{-1}, respectively. The equation describing the lagged first order accumulation of nitrate is

$$(NO_3)_{t - t_D} = NS \{1-\exp(-k*(t - t_D))\} \qquad [17]$$

where NS is estimated from equation [14]. k is the corrected rate constant for the soil organic N mineralization, about 0.0077 per day at 35°C. For waste with C:N < 23 the equation is simply

$$(NO_3)_t = (NX + NS) \{1-\exp(-k*t)\} \qquad [18]$$

where the rate constant is from equation [12] and t is time in days.

This useful model tells us that about 50% of the N available for mineralization is converted in 3 to 6 weeks for poultry and swine wastes. In comparison, beef wastes required approximately 18 weeks under similar conditions.

Sims (1986) studied the effects of temperature and moisture on net N mineralization of poultry manure incorporated in Evesboro loamy sand (mesic, coated Typic Quartzpsamments) from Delaware, in a laboratory setting. Mineralization, i.e., change in inorganic N (ammonium and nitrate nitrogen) from manure, and pH were measured at 30, 90, and 150 days.

Most of the net mineralization observed at 25°C and 40°C occurred within the first 90 days. Mineralization was reduced at 0°C, but between 90 and 150 days considerable accumulations of inorganic N were detected at this temperature.

Two moisture levels were contrasted, (1) field capacity maintained for 150 days and (2) moisture stress, which started at field capacity but was not maintained by water additions. Thirty to 60% of the organic N added to the soil was mineralized in the unstressed moisture experiment.

Differences in N mineralization between manure samples from different sources were noted (sample 2 behaved differently than samples 1 and 3). Accumulations of ammonium under cold or dry conditions resulted in elevated soil pH levels. Considerable soil acidification occurred under warm, moist conditions due to rapid denitrification of ammonium. Net mineralization over time is reported in Tables 136 and 137. The negative numbers in Table 136 represent N losses relative to an unamended soil sample.

Table 136
Temperature Effects on Mineralization of Organic Nitrogen in an Evesboro Loamy Sand Amended with Poultry Manure and Incubated at Field Capacity Moisture
Sims, 1986

Temperature °C	Elapsed Time, days	Manure Sample 1	2	3
		percent organic N mineralized		
0	30	-7	-12	10
	90	8	-6	28
	150	37	7	30
25	30	16	-6	16
	90	38	6	36
	150	40	25	37
40	30	15	-7	-2
	90	47	32	29
	150	51	17	64
LSD(0.05)		16	10	18

Table 137
The Influence of Soil Moisture Stress on Ammonium and Nitrate Levels in an Evesboro Loamy Sand Amended with Poultry Manure Incubated at 25°C
Sims, 1986

Moisture	Time	Sample 1		Sample 2		Sample 3	
		NH_4^+-N	NO_3^--N	NH_4^+-N	NO_3^--N	NH_4^+-N	NO_3^--N
	days	mgkg	mg/kg	mg/kg	mg/kg	mg/kg	mg/kg
Unstressed	30	2	190	1	152	3	177
	90	1	236	4	182	4	223
	150	5	238	2	216	2	231
Stressed	30	29	177	12	180	95	62
	90	24	175	15	183	97	60
	150	35	181	13	173	92	45
LSD(0.05)		16	26	23	57	40	28

Sims concluded that even at cold temperatures (0°C) mineralization of N can be expected. Fall applications of poultry manure could thus result in considerable N loss via leaching early in the following year due to the combined effect of spring rainfall and minimal crop uptake of N. For spring applications (25°C) mineralization will proceed rapidly and could be essentially complete by early summer (90 days).

C. AMMONIA VOLATILIZATION FROM LAND AREAS RECEIVING ANIMAL WASTE

Reddy et al. (1979b) provide a model for ammonia volatilization. This process can cause substantial losses of N from animal wastes having high initial NH_4 concentrations.

Initial NH_4 in animal manures is mostly derived from the hydrolysis of urea and uric acids in the urine and feces and results in a temporarily high pH, thus creating conditions favorable for NH_3 volatilization.

Volatile losses of NH_3 occur when manures are exposed to air movement in animal production facilities and after manure spreading in fields. In pastures and rangelands, most of the ammonia-N in the animal wastes is rapidly lost after defecation by the animals.

Several equations have been proposed to describe ammonia volatilization under various conditions such as lagoon storage, NH_3 removal as a pretreatment process, aquatic systems, and in animal manures (Bouldin et al., 1974; Srinath and Loehr, 1974; Koelliker and Miner, 1973; Steenhuis et al., 1976). All these researchers basically followed the fundamental principles of NH_3 chemistry, i.e., the rate of NH_3 loss is proportional to the difference in activity of NH_3 in the atmosphere and in the manure.

The equations presented by Koelliker and Miner (1973) are useful for describing the desorption of NH_4-N in anaerobic animal manure lagoons, by applying mass transfer relationships to NH_3 loss. Their data also provide an evaluation of the mass transfer coefficient.

Reddy et al. (1979b) focus on ammonia loss resulting from land application of manure. The following discussion is taken from that paper.

Ammonia N in animal manures is mainly derived from (a) hydrolysis of urea present in urine and feces, and (b) ammonification of organic N. The former process occurs rapidly and accounts for almost all of the initial NH_4-N present in the manures. The initial NH_4-N concentration of the manures is dependent on the type of waste, waste handling and collection practices, and the pretreatment of wastes.

Reddy et al. assume that the nitrification (conversion of ammonium to nitrate) process is insignificant when volatilization is occurring. They represent the total ammonical N (ammonium plus aqueous ammonia) in the soil waste system at time t, $(TAN)_t$ as

$$(TAN)_t = (TAN)_0 \exp(-Kt) \qquad [19]$$

where K is an aggregate first order rate coefficient, in day^{-1}. Here equilibrium conditions are assumed. Also it is assumed that the pH of the soil-waste system does not change appreciably with NH_3 volatilization.

Under most soil and environmental conditions NH_3 volatilization occurs in one or two stages, approximated by a first order relationship at each stage. The first stage loss is generally rapid, governed by NH_3 concentrations in the manure. The second stage losses are generally characterized by low NH_3 concentrations as manure is subjected to drying.

Reddy et al. (1979b) consider the losses of NH_3 in the second stage to be insignificant compared to the first stage losses. When losses occur in the first stage, NH_3 volatilization is generally terminated after one week because of the rapid decrease in total ammonical nitrogen (TAN) and increasing dominance of the nitrification process.

Reddy et al. (1979b) summarize lab and field measurements of NH_3 volatilization rates (Table 138). The K values as estimated from field studies are lower compared to the values obtained from laboratory experiments.

Under field conditions, the K values for first stage losses range from 0.179 to 1.059 day^{-1} (with a "half life" of $t_{1/2}$ = 0.7 to 3.9 days). K estimates for second stage losses range from 0.019 to 0.261 day^{-1} ($t_{1/2}$ = 2.7 to 365 days). However, the maximum loss period under field conditions of the experiments conducted was shown to be 9 days during the first stage and 20 days during the second stage.

The variation in K with initial ammonia content indicates that either the first order approximation model is incomplete or that severe measurement error existed.

Table 139 contains a list of temperature correction coefficients for the rate of ammonia volatilization.

Table 138
Rate Constants for Ammonia Volatilization from Waste Treated Land
Reddy et al., 1979b

Type of Manure	Temp. °C	K measured day^{-1}	K at 20°C day^{-1}	Percent N Loss (max)	Number of days K is applicable	Remarks and References
Dairy manure						
applied in April	10					Lauer et al., 1976 Field study. Manure surface applied.
20 kg NH_4-N/ha		0.371	0.801	84	5	
183 kg NH_4-N/ha		0.198	0.427	63	5	
350 kg NH_4-N/ha		0.179	0.386	80	9	
applied in June	20					
45 kg NH_4-N/ha		0.377	0.377	85	5	
277 kg NH_4-N/ha		0.265	0.265	73	5	
applied in August	20					
189 kg NH_4-N/ha		0.201	0.201	63	5	
Dairy manure						
applied in June	20					Lemon, 1977
25 kg NH_4-N/ha		1.059	1.059	96	3	Field study. Loss of NH_3 determined from NH_3 flux.
Dairy manure						
applied in January	-2 to 10	0.063	0.185			Steenhuis et al., 1976 Field study. K value estimated from half life. Temperature correction coefficient, θ, for -20 to 0 °C was 1.15, and from 0 to 10 °C, θ = 1.10
Dairy manure _ low wind: 200 cm^3/min airflow	-20	0.022	1.678	8	4	Steenhuis et al., 1976 Laboratory study using dairy waste and urine but no soil.
	0	0.407	1.897	80	4	
	10	1.386	2.992	100	4	
high wind: 400 cm^3/min airflow	-20	0.061	4.653	22	4	
	0	0.866	4.037	97	4	
	10	2.131	4.599	100	4	
low waste application	-3	0.433	3.069	98	9	
	10	1.555	3.357	100	9	
high waste application	-20	0.022	1.678	18	9	
	-3	0.204	1.446	84	9	
	10	0.693	1.496	100	9	
Sewage sludge applied in May						Beauchamp et al., 1978
150 kg NH_4-N/ha	7-20	0.193	0.318	60	5	Digested sludge applied to soil surface.
applied in October						
89 kg NH_4-N/ha	4-20	0.117	0.217	56	7	
Stream water	20	1.820	1.820	100		Stratton, 1968 Laboratory study. Water pH 8.5 θ = 1.07 from 15 to 30 °C.
	30	3.690	1.709	100		
Ammonium sulfate	12	0.076	0.304	66	14	Fenn and Kissel, 1974 Laboratory study. Fertilizer applied on calcareous soil, pH 7.9 θ = 1.03 for 12 to 32 °C.
	22	0.096	0.082	74	14	
	32	0.131	0.052	84	14	

Table 139
Temperature Correction Coefficient Values for the Rate of NH_3 Volatilization
Reddy et al., 1979b

Experimental Conditions	Temperature Range °C	θ	Reference
Diary manure	-20 - 0	1.15	Steenhuis et al., 1976
	0-10	1.10	
Stream water pH 8.5	15-30	1.07	Stratton, 1968
$(NH_4)_3SO_4$ fertilizer surface applied on calcareous soil	12-32	1.03	Fenn and Kissel, 1974
NH_3 desorption in manure	0-35	1.06	Srinath and Loehr, 1974
NH_3 absorption coefficients	10-50	1.08	Haslam et al., 1924
range	-20 to 50	1.03 to 1.15	
average		1.08 ± 0.04	

The reaction rate, K, is increased approximately 1.3 to 3.5 times for each 10°C rise in temperature when the system temperature is between 0 and 30°C. When the temperatures were from -20 to 0°C the reaction increased from 10 to 18 times for each 20°C rise. Increasing the temperature increases the ionization constant and thus a larger proportion of total ammonical nitrogen is transformed to aqueous ammonia.

Ammonia losses were also inversely proportional to the adsorption capacity (cation exchange capacity, CEC) of a soil. The contact of ammonium ions with soil adsorption sites is controlled by the method of fertilizer application.

Surface application of solid and semi-liquid manure results in a large fraction of ammonium N not in contact with adsorption sites in the soil. However, when manures were incorporated into the soil system, contact with adsorption sites increased and ammonia losses decreased. Surface spreading can result in from 61 to 99 percent of total ammonical nitrogen loss (Lauer et al., 1976; Lemon, 1977).

Losses from alkaline or limed soils can be appreciable. This is because these soil complexes have a greater calcium ion saturation, displacing ammonium ions from exchange sites, and because the increased pH of these soils favor ammonia formation (and hence loss).

Based on the work of Adriano et al. (1971) a correction factor for K, F_C, relating K linearly to CEC (cation exchange capacity in meq/100 g of soil) can be expressed:

$$F_C = 1 - 0.038\ \text{CEC} \qquad \text{for manure incorporation} \qquad [20a]$$

$$FC = 1 \qquad \text{for surface application} \qquad [20b]$$

When the manure is surface applied $F_C = 1.00$ because the effect of CEC on volatilization loss was assumed to be zero. $K(CEC_1)$ measured in soil having CEC_1 can be adjusted for a different soil having CEC_2 by

$$K(CEC_2) = K(CEC_1) * F_C(CEC_2)/F_C(CEC_1) \qquad [21]$$

A correction for wind speed was obtained by Reddy et al. (1979b) from the data of Watkins et al., (1972)

$$F_{AS} = 1.44 + 0.16 \ln AS \qquad \text{for } AS < 0.06 \qquad [22a]$$

$$F_{AS} = 1 \qquad \text{for } AS \geq 0.06 \qquad [22b]$$

where AS is air speed, km/hr and F_{AS} is the air speed correction factor.

Since the derivation of K includes a pH adjustment, a separate correction factor for pH is not necessary. The fully corrected K, i.e., for a different temperature, cation exchange capacity of the soil, and air speed, and half life of total ammonical nitrogen, is obtained

$$K(T_2, AS_2, CEC_2) = K(T_1) * \theta \exp(T_2 - T_1) * F_{AS} * F_C(2)/F_C(1) \qquad [23]$$

$$t_{½} = 0.693/K(T_2, AS_2, CEC_2) \qquad [24]$$

where, as before, T is temperature. Reddy et al. provide an example of the temperature correction in Table 140. They use a reference K value for incorporated manure of $K = 0.409\ \text{day}^{-1}$ at 20°C. It was assumed that F_{AS} and F_C equaled 1.

The average K value for incorporated animal manure was adjusted for cation exchange capacity of various soils in Table 141.

The extent of ammonia volatilization losses depend on the type of land areas receiving animal manure. These are pastures and rangelands, croplands, disposal areas, feedlots, and manure storage areas (ordered by increasing volatilization potential).

Table 140
Calculated K Values for Ammonia Volatilization as Influenced by Temperature
Reddy et al., 1979b

Temperature °C	Animal Manure K, day^{-1}	Animal Manure t½, days	Stream Water K, day^{-1}	Stream Water t½, days
-10	0.021	33.00	0.181	3.83
0	0.087	7.96	0.390	1.77
10	0.189	3.67	0.843	0.82
15	0.278	2.49	1.238	0.56
20	0.409[1]	1.69	1.820[2]	0.38
25	0.601	1.15	2.674	0.26
30	0.883	0.78	3.929	0.18

---

[1] Average K value obtained from field studies of Lauer et al. (1976), Steenhuis et al. (1976), Lemon (1977), and Beauchamp et al. (1978).

[2] Average K value obtained from a laboratory study on stream water by Stratton (1968)

Table 141
Calculated K Values for NH_3 Volatilization as Influenced by Cation Exchange Capacity of the Soil
Reddy et al., 1979b

CEC meq/100 g	F_C	K day_{-1}	t½ days
0[1]	1.00	0.409[2]	1.69
5	0.81	0.331	2.09
10	0.62	0.254	2.73
15	0.43	0.176	3.94
20	0.24	0.098	7.07
25	0.05	0.020	34.65

---

[1] Represents surface applied manure

[2] Average K value obtained from field studies of Lauer et al. (1976), Steenhuis et al. (1976), Lemon (1977), and Beauchamp et al. (1978) at an average daily temperature of 20 °C.

D. ATMOSPHERIC REDEPOSITION OF NITROGEN

Portions of the volatilized nitrogen is redeposited locally while an unquantified amount is transported over long distances. Hoeft et al. (1972) found atmospheric deposition of organic nitrogen to be two times greater at sites within 0.8 km of a barnyard than at sites further than 0.8 km. Ammonium nitrogen concentrations were 4 to 5 times greater at the sites near a barnyard.

Hoeft et al. measured N concentrations at 18 barnyard sites, 2 rural sites (4.8 km from nearest barnyard), and three urban sites in Wisconsin. They found, in the rural areas, adjacent to barnyards, the ratio of ammonium-N to nitrate-N was approximately 3.5:1, and organic-N to nitrate-N was 4:1. The urban and non-feedlot rural site ammonium:organic nitrogen:nitrate ratios were about 1:1.5:1.

Ammonium and organic N concentrations followed a seasonal pattern. Values were highest during months when the ground was not frozen (April, Many and June) and lowest when the ground was frozen. Nitrate concentrations did not vary as much with season, and were higher in urban sites (suggesting an industrial origin).

Ammonium-N concentrations ranged from 0 to 13 ppm for feedlot locations and from 0 to 3 ppm for the other sites. Ammonium accounted for 42% of the total precipitation N at feedlot sites and for 24% at the other sites.

Organic-N concentrations were about twice as high at barnyard sites than at the other sites. Table 142 gives the seasonal average concentrations at the three types of site.

Table 142
Nitrogen Species in Rural and Urban Precipitation
Hoeft et al., 1972

	5/27-6/23	6/23-9/15	9/15-1/6	1/6-3/31	3/31-6/23	Average
		kg/ha per 28 days				kg/hay·yr
Rural, < 0.8 km from Barnyard						
NH_4-N	1.73	1.38	0.63	0.28	1.92	12.22
NO_3-N	0.33	0.21	0.21	0.18	0.38	3.15
Org-N	1.90	1.31	0.86	0.27	1.86	14.43
Total	3.96	2.90	1.70	0.73	4.16	30.16
Rural, removed from Barnyard						
NH_4-N	0.38	0.33	0.12	0.05	0.28	2.86
NO_3-N	0.27	0.15	0.30	0.06	0.32	2.73
Org-N	0.68	1.65	0.60	0.09	0.62	7.54
Total	1.33	2.13	1.02	0.20	1.22	13.13
Urban						
NH_4-N	0.82	0.25	0.07	0.06	0.48	3.61
NO_3-N	0.53	0.27	0.28	0.08	0.38	3.73
Org-N	0.35	0.86	0.58	0.07	0.95	6.19
Total	1.70	1.38	0.93	0.21	1.81	13.53

In addition to the return of N through precipitation, direct absorption of ammonium nitrogen from the air into water surfaces in the vicinity of feedlots has been observed (Hutchinson and Viets, 1969).

Though some of the N losses are returned locally, much of the ammonia is transported greater distances, as evidenced by the neutralizing effect exerted by agriculturally generated ammonia on gaseous sulfur oxides originating from midwestern coal burning power plants (Gorham et al., 1984; Möller and Schieferdecker, 1985). The factors controlling redeposition, dry and wet, of ammonia need clarification.

VIII. CHAPTER 2 CONCLUSIONS

When modeling manure management practices, it is important to take into account the nutrient losses occurring before land disposal as well as after it.

Conceptually, manure N is lost at four points during the collection and disposal process: collection, storage, application, and weathering. Relative to fresh manure, collection N losses can be from 20 to 60%, storage losses from 5 to 15% and application losses from 10 to 60%. Further N losses, once manure is applied to fields, are dependent on in-field conditions, and meteorology.

Phosphorus is lost mainly by washoff and in liquid discharge from manure accumulations.

COLLECTION LOSSES

More than half the nitrogen in fresh manure may be present as ammonia or be converted to ammonia in a very short time following excretion. Much of this ammonia volatilizes into the air. Some organic forms of nitrogen are also lost to the gas phase. This collection loss can amount to up to 50% of the total manurial nitrogen.

The other side of ammonia volatilization is atmospheric ammonia redeposition. A portions of the volatilized ammonia is redeposited locally while an unquantified amount is transported over long distances. This phenomenon is not well quantified.

A more elusive collection loss is the manure that escapes collection efforts and leaves the farmstead with precipitation or wash water. Though potentially significant, we did not find any studies directly addressing the issue of collection efficiency as distinct from the volatilization problem.

STORAGE LOSSES

The major purpose of manure storage is to accumulate and protect manure until it can be applied to the land. Manure storage, ideally, allows a farmer to time the field application of manure to periods when it can be used as fertilizer for some crop. Improperly sized storage facilities can force a farmer to waste manure.

Manure collection and storage systems vary tremendously in their ability to conserve N. Nitrogen losses can range from 10 to 90%.

Modeling studies have shown that manure storage does not necessarily result a reduction in the amount of the N and P discharged to the environment from manured fields. Manure storage without structural BMPs may not produce the desired amount of nutrient loss reduction.

Nutrient management, with or without manure storage, can reduce nutrient loss from fields significantly.

SOLID/LIQUID SEPARATORS

Soluble forms of nitrogen and phosphorus may be leached by water passing through manure storage areas. Solid/liquid separators effectively turn a nonpoint source into a point source by discharging a nutrient rich effluent into the environment. The effluent from such facilities should be collected in a holding structure for later land irrigation and not discharged into a grassed waterway or other channel. Farmstead runoff and parlor waste water should be diverted away from these structures to reduce runoff losses.

EARTHEN MANURE STORAGE PITS

Coarse textured soils are unsuitable for earthen manure storage pits. Fine and medium textured soils tend to restrict nutrient leaching in the short term. A first flush effect is often observed in ground water under such pits in the first year. Elevated nitrate levels diminish after one year, presumably due to soil sealing.

However, ground water pollution beneath manure pits installed in fine textured soils has been observed. This pollution has been attributed to the actions of burrowing animals, penetrating tree roots, lagoon overflow, and pumping and manure transfer problems.

The severity of ground water nitrate pollution is dependent on the local geohydrology, the distance from the source, the depth of the sampled well, and the magnitude of source.

LAND APPLICATION LOSSES

Manure can be applied to land in several ways. It can be surface spread, injected as a slurry, plowed down, or spray irrigated. The method of application influences the mobility of the nutrients.

The timing of manure application is also important. More ground and surface water pollution is generated by fall manure application than from spring application. The type of fertilizer (commercial or manure) used is not as important to nutrient flux as the time of application.

After the land application of manure, ammonia volatile losses are inversely proportional to the adsorption capacity (cation exchange capacity) of a soil. The contact of ammonium ions with soil adsorption sites is controlled by the method of fertilizer application.

LOSSES TO AIR

Surface application of solid and semi-liquid manure results in a large fraction of ammonium N not in contact with adsorption sites in the soil. However, when manures are incorporated into the soil system, contact with adsorption sites increase and ammonia losses decrease. Surface spreading can result in from 61 to 99 percent of total ammoniacal nitrogen loss.

Losses from alkaline or limed soils can be appreciable. This is because these soil complexes have a greater calcium ion saturation, displacing ammonium ions from exchange sites, and because the increased pH of these soils favor ammonia formation (and hence loss).

LOSSES TO SURFACE WATER

Nutrient runoff losses from non-row crops (millet, rye) are similar for surface applied manure and commercial fertilizer applied at the same rates.

Phosphorus loss from corn fields is related to the type of tillage used. Surface applied manure can increase the dissolved phosphorus in runoff from no-till fields and the total phosphorus from chisel plowed fields. Surface applied manure had no effect on phosphorus loss from conventionally tilled fields.

LOSSES TO GROUND WATER

One of the most important factors affecting the rate at which organic nitrogen is mineralized to nitrate is the method of manure application. This in turn determines the amount of dissolved inorganic plant nutrients available for crop uptake or for leaching or runoff loss.

Nitrogen leaching losses from applied manure are greatest from fall applied manure because of the combination of spring rainfall and minimal crop uptake of N.

Even at cold temperatures mineralization of N can be expected. Fall applications of poultry manure could thus result in considerable N loss via leaching early in the following year due to the combined effect of spring rainfall and minimal crop uptake of N. For spring applications mineralization will proceed rapidly and could be essentially complete by early summer (90 days).

WEATHERING LOSSES

PASTURE

Grazed pasture and pasture spread with waste manure do not contribute the same pollutant loads.

Total N and P yields from grazing pastures are not related to the number of animals involved, but rather to hydrological and management factors, namely grazing management system, erosion and sediment transport control, and runoff control.

CROPPED FIELDS

LOSSES TO SURFACE WATER

For manured fields, there seem to be linear relationships between manure application annual rate and annual runoff nutrient yields, if the data are segregated by date of application.

Substantial nutrient losses can occur with field runoff when manure is applied on frozen or snow covered soil during the winter months. Much of the nutrient loss from manure application on frozen or snow covered ground is associated with applying manure during a thaw or when a thaw and rainfall occur soon after manure application. This points out the water quality benefits to be realized from manure storage during the winter.

LOSSES TO GROUND WATER

The surface application of manure can reduce infiltration rates initially, but the effect is not detectable after 4 to 5 days. On the long term, manure amended soils may have increased infiltration rates due to the activity of earthworms.

On cropland fertilized by manure, recharge can cause an increase or decrease in dissolved nitrate concentration in ground water depending on the amount of manure spread on the site prior to the storm.

Changes in ground water nitrate concentration can occur both rapidly (2-3 days, lasting for about a week) as well as gradually (the effects are perceived several weeks or more after the storm event), because recharge enters the ground water through two different routes: direct recharge through near-surface bedrock fractures and sinkholes, and slow recharge through small channels and pores in the unsaturated zone.

The amount of fresh manure on the land surface at the time of a storm determines whether nitrate concentrations increase or decrease, and determine the magnitude of the increase or decrease.

The application of animal wastes increases soluble P, acid extractable P,and equilibrium P concentration values, and decreases the P sorption capacity of the soil. These changes are directly related to the loading rates of animal wastes. The decrease in P sorption capacity may result in more soluble P moving into deeper soil layers and ground water, depending on the soil.

FEEDLOTS

The range of concentrations of runoff pollutants from feedlots is wide. Snowmelt runoff resulting from winter thawing conditions produced higher concentrations of pollutants than those produced by rainfall runoff (by a factor of 2 to 3).

From 2 to 10 percent of the manure generated in feedlots is lost to surface runoff. The total nutrient export from feedlots is two to three orders of magnitude greater than nutrient export in runoff draining other agricultural activities.

CHAPTER 3. THE EFFICACY OF VEGETATED FILTER STRIPS

INTRODUCTION

Vegetated Filter Strips (VFSs) are bands of planted or indigenous vegetation situated downslope of cropland or animal production facilities. They provide localized erosion protection and filter sediment, and to a lesser extent, nutrients, organics, pathogens, and pesticides from agricultural runoff before these contaminants reach receiving waters. Because of low installation and maintenance costs, and perceived effectiveness in removing pollutants, VFS use has been encouraged by conservation and regulatory agencies.

The major pollutant removal mechanisms associated with VFSs involve changes in flow hydraulics that enhance the opportunity for runoff and pollutants to infiltrate into the soil, deposition of suspended solids, filtration of suspended sediment by vegetation, adsorption on soil and plant surfaces, and absorption of soluble pollutants by plants. Surface runoff must pass slowly and uniformly through the VFS to provide sufficient contact time for removal mechanisms to function.

Because the grasses grown on VFSs offer high resistance to shallow overland flow, they decrease overland flow velocity at the boundary between the source and the filter strip as well as in the filter. This decreases sediment transport capacity of the runoff significantly. If the transport capacity is less than the incoming load of suspended solids, excess solids (and sediment bound pollutants) are deposited and trapped in the VFS.

The two major types of vegetation grown on VFSs are grasses and trees. Grass filter strips have been advocated for waste water treatment as well as for edge of field stabilization. Forested filter strips are often established to protect stream banks or at field edges as wind breaks. This paper deals only with grassed VFSs.

I. REMOVAL OF SOLID PARTICLES

Sediment removal efficiency of grassed filters has been reported to be more than 90% (Neibling and Alberts, 1979). These authors studied the particle size of sediment leaving 7% slope bluegrass filters ranging in length from 0.6 to 4.9 m. Upslope sediment source plots on Miami silt loam were moldboard plowed, disc-harrowed, and leveled to prevent runoff channelization.

Five inches of simulated rainfall were applied to each 1.8 x 6.1 bare source plot over a 2 day period, first as a 1 hour dry run, followed 24 hours later by two 30 minute wet runs separated by a 30 minute interval. Rainfall intensity was 2.5 in/hr (6.3 cm/hr).

Sediment entering and leaving the strips in runoff was grab sampled five times during each experiment for the calculation of sediment discharge curves.

Discharge rates (in g/m/sec) for the clay size fraction were reduced by 37, 78, 82 and 83% for the 0.6, 1.2, 2.4, and 4.9 m filters, respectively. Table 143 gives the size breakdown of the sediment entering and leaving the 4.9 m strip. Note the enrichment in particles in the finer classes.

Table 143
Particle Size Distribution of Sediment Entering and Leaving a VFS
Neibling and Alberts, 1979

Sediment Size Class mm	Percent of Total Entering	Percent of Total Leaving[1]
> 2	2.2	0.5
2 - 1	5.5	0.5
1 - 0.5	19.1	0.5
0.5 - 0.21	18.4	1.1
0.21 - 0.05	8.0	2.6
0.05 - 0.035	7.0	1.0
0.035 - 0.02	10.1	5.5
0.02 - 0.01	13.2	17.6
0.01 - 0.002	13.0	31.5
< 0.002	4.9	39.2

---

[1] Data read from histograms. Approximate graph reading error = ±0.7% (i.e., the equivalent in percent of the smallest unit on the ruler used to read the graph.)

Hayes and Hairston (1983) evaluated the efficiency of vegetative filters (Kentucky 31 tall fescue, 26 x 2.6 m, and 2.4% slope) over 16 months to determine the effective duration of sediment control. Figure 2 and Table 144 present the change in sediment trapping efficiency of a grass filter over time. The authors warn that there was considerable measurement error. The results from the second instrumented filter are not reported here because of the large amount of experimental noise. The sediment source was a repeatedly disked upland area.

When a filter strip is new, the majority of sediment deposition occurs just upslope and in the first meter of the filter. This condition persists until upper portions of the filter are buried in sediment. Subsequent sediment flow into the filter results in the advance of a wedge shaped sediment deposit down through the filter.

Hayes and Hairston (1983) did not observe sediment wedge movement because of the small particle size of the sediment entering the filters. (Filtration of solid particles by vegetation during overland flow can be expected to remove larger soil particles, aggregates, and manure particles more than fine silts and clays.) They did observe increased deposition in the first 15 cm of the filter.

Table 144
Change in Sediment Trapping Efficiency Over Time
Hayes and Hairston, 1983

Storm Date	Influent concentration mg/l	Effluent concentration mg/l	Fraction trapped	Inflow rate[1] cm$_3$/sec·cm
8-19-81	1965	151	0.834	0.11
8-20-81	1207	131	0.810	0.43
9-4-81	1793	243	0.833	0.31
9-13-81	2292	283	0.806	0.27
9-15-81	896	221	0.692	0.23
2-2-82	1787	428	0.736	0.17
2-8-82	1870	1202	0.343	2.92
2-15-82	2419	1801	0.249	0.71
2-26-82	900	643	0.284	0.25
3-6-82	1463	1006	0.303	1.55
4-4-82	2988	1295	0.554	6.24
4-20-82	957	350	0.445	2.55
5-30-82	2545	2227	0.226	15.80
5-31-82	3395	2530	0.258	4.53
6-2-82	2444	1156	0.520	14.98
6-4-82	3452	1867	0.563	3.64
6-12-82	6234	2214	0.636	6.36
7-16-82	7363	1070	0.759	4.52

---

[1] Inflow rate averaged over each storm

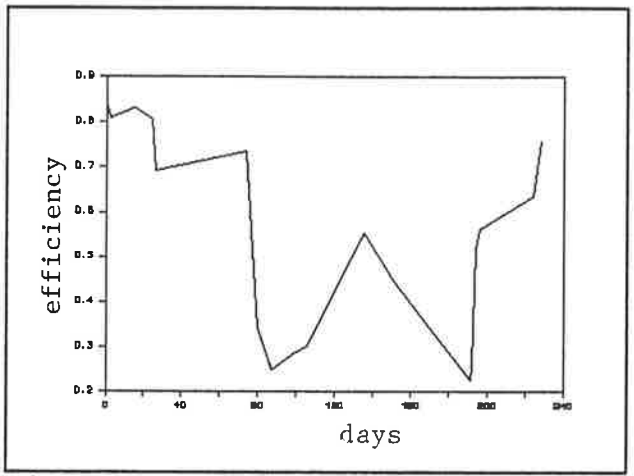

Figure 2. Change in Sediment Removal Efficiency Over Time
Hayes and Hairston, 1983

The depth of runoff flowing through the filter was an important determinant of filter performance. High trapping efficiencies were reported as long as the vegetal medium was not submerged, but decreased dramatically at higher runoff rates that inundated the medium.

Though it was one of the stated goals of the experiment, the authors did not make any conclusions on the lifespan of vegetated filters.

They commented that research is needed on the conditions under which sediment deposited in grassed filters is eroded from grassed filters.

Other authors (Magette et al., 1987 and Dillaha et al., 1988, see sections below) also report diminished solids filtering ability of grassed strips after sequential runoff events. Solids filtering efficiency generally drops from near 90% to roughly 50 or 70% within a single season.

II. NUTRIENT REMOVAL

A. TREATING FEEDLOT RUNOFF

Dickey and Vanderholm (1980) and Vanderholm et al. (1979) studied channelized and overland flow grassed systems for treating feedlot runoff. They observed over 80% reductions in concentrations and 95% reductions in weights of nutrients, solids and oxygen demanding material in runoff from a 1750 foot grassed filter.

They also developed filter design criteria based on influent load and contact time. For instance, the minimum contact time (time to traverse the grassed waterway) recommended for treating runoff from different sized feedlots are listed in Table 145.

Table 145
Recommended Minimum Contact Times for a 1750 ft VFS
Dickey and Vanderholm, 1980

feedlot area, m^2	minimum time, hr
929	2
1394	3
1858	4
2323	5

The minimum flow lengths to achieve 95% removal in 2 hr contact time for various slopes for Manning's n = 0.3 and design flow depth = 0.5 inch are listed in Table 146.

Table 146
Recommended Minimum Flow Lengths for VFSs of Various Slopes
Dickey and Vanderholm, 1980

slope,%	flow length, m
0.5	91.4
0.75	113
1	131
2	185
3	227
4	262

In these studies the subject of interflow quality was not addressed.

Bingham et al. (1980) and Overcash et al. (1981) applied chicken manure to grassed areas upslope of a series of grassed (fescue) terraces and measured runoff quality at numerous downslope distances. Poultry manure was applied monthly and runoff from natural storm events was collected and analyzed. The schedule of spreading and dates of storm events are found in Table 147. The terrace soils were Cecil clay loam over clay with a 6 to 8% slope.

The reader is referred to Bingham et al. for graphs of the individual pollutant concentrations at each storm at eight buffer lengths (expressed by the authors as buffer length to disposal area length ratio). Table 148 provides the average (over all storms) concentration of pollutants at the various ratios. The authors only reported results from large events because no nutrient reduction

Table 147
Pollutant Application Schedule and Rainfall Dates for Grassed Terrace Study
Bingham et al., 1980

Date	TKN	T-P	COD	Cl	TOC	Rainfall	days elapsed[1]	previous storms[2]
	kg/ha					cm		
8-23-77	158	31	1833	26	194			
9-29-77	78	23	923	11	155			
10-12-77	35	8	384	5	47			
10-26-77						5.5	15	1
11-2-77	53	12	480	5	112			
11-6-77						4.4	5	1
11-21-77	31	6	259	4	47			
12-15-77	70	19	991	9	113			
1-5-78	28	9	376	4	36			
1-8-78						2.6	4	1
1-14-78						4.3	9	2
1-17-78						2.1	13	3
1-19-78						4.4	15	4
1-25-78						2.8	21	5
1-27-78	69	19	847	7	117			
4-21-78	80	24	1028	14	140			
4-26-78						14.0	5	0
5-5-78	60	22	816	14	145			
5-8-78						4.3	3	0
5-17-78	60	24	741	12	167			
Total	722	197	8678	111	1273			

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[1] Days after waste applied

[2] Number of previous storm events

was observed for the small storms. (This was attributed to contamination of the sampler. Whatever the cause, the sampling bias should be remembered when considering these data).

It should be noted that the experimental design is not typical of grassed waterways in that runoff was collected at intermediate points for chemical analysis, then released by a slotted distribution gutter onto the next test section of buffer, eliminating channelized flow and reducing the runoff velocities, two factors which would increase the pollutant removal of the system.

The authors concluded that in order to achieve background levels of contamination, terraced fescue buffers constructed according to the experimental design should be at least as long as the length of the area generating the polluted runoff. They qualified their conclusions as being valid only for waste areas less than 13 meters in length.

The variation in pollutant concentrations at a given location in the filter terrace system from one storm event to another was affected by the number of days since the last waste application, previous rainfall, and climatic and seasonal factors.

Table 148
Average Concentration of Runoff Pollutants at Various Buffer Terrace Length/Waste Area Length Ratios
Bingham et al., 1980

Ratio[1]	COD	TKN	T-P	NO_3-N	Cl	TOC	VS/TS[2]
	....	....	mg/l	....	....	....	
none[3]	96.9	6.9	5.1	9.5	4.0	29.0	0.35
0.2	87.8	5.9	4.7	9.1	4.7	28.1	0.33
0.3	74.8	5.6	2.4	7.9	4.3	26.5	0.40
0.5[4]	96.2	4.9	2.1	0.2	5.0	32.5	0.29
0.75[4]	64.5	3.8	1.3	0.1	3.0	26.0	0.22
1.0	64.5	3.0	1.1	1.0	2.8	19.0	0.24
1.4	62.0	3.4	1.4	1.1	3.6	19.9	0.27
1.6	56.1	2.6	0.8	0.8	3.2	18.6	0.21
2.0	65.0	2.8	0.8	0.2	2.4	19.2	0.22
2.6	65.9	3.0	1.0	0.2	3.0	19.9	0.21
Control[5]	67.8	3.5	0.7	1.7	3.4	18.4	0.19

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[1] Buffer area length/waste area length
[2] COD = chemical oxygen demand, TOC = total organic carbon, VS/TS = volatile solids/total suspended solids
[3] Pure waste area runoff, no buffer
[4] For two events; other averages are for nine events
[5] A buffer but no waste disposal area above it

These studies would have been more meaningful had the authors reported the volume of runoff along with the pollutant concentrations so that areal loading rates could be compared. The authors single out three major factors affecting buffer zone effectiveness: (1) influent pollutant concentration, (2) dilution, and (3) infiltration. Without more hydrologic information it is difficult to estimate the contributions of dilution and infiltration.

The mechanical interruption and redistribution of runoff in this study makes its efficiency reports not generalizable to VFSs lacking these features.

The reader is referred to Overcash et al. (1981) for a one-dimensional mathematical model of pollutant reduction for buffer strips of various buffer area length to waste area length ratios.

Edwards et al. (1983) monitored storm runoff from a paved feedlot at several points in a runoff purification system. Runoff was sampled for 3 years as it left a paved feedlot, after it passed through a shallow concrete settling basin, and below two consecutive 30.5 m, 4.5 m wide, 2% slope fescue filter strips in Ohio. The 243 m^2 feedlot was stocked with 56 steer calves.

Resulting changes in effluent quality are summarized in Table 149. The basin was considered more effective for large storm events and the filter strips more effective when the basin was slowly drained after solids settling.

Table 149
Feedlot Runoff Pollutant Reduction by a Settling Basin and Two Consecutive 30 Meter Vegetated Filter Strips
Edwards et al., 1983

	Runoff	TS	Total P	Total N
		percent reduction		
leaving basin	17	54	41	35
after first strip	-2	23	29	31
after second strip	-5	10	14	16
TOTAL		87	84	82

Dillaha et al. (1988) provide a critical examination of the effectiveness of VFSs for treating feedlot runoff. Unlike previous studies, they document the detrimental effects of channelized flow within a filter strip and the progressive reduction of sediment trapping ability as successive storms clog and bury a filter.

The pollutant source areas for this study were simulated feedlots, compacted 55 by 18.3 m bare plots to which fresh dairy manure had been added. The manure was applied 24 to 40 hours before each of two sets of simulated runoff events, at rates to approximate the waste accumulations in a feedlot after 7 (Test 1) and 14 (Test 2) days.

After each manure application, simulated rainfall was applied three times at an intensity of 5 cm/hr: first for 60 minutes, followed 24 hours later by a 30 minute application, which was followed by a 30 minute dry period before the final 30 minute application, for a total of 10 cm rainfall per test. The 60 minute application corresponded to the typical 2 to 5 year storm of the Blacksburg, Virginia test site.

The plots were constructed on an eroded Groseclose silt loam (clayey mixed, mesic Typic Hapludult) soil.

The most important feature of this study was the treatment of slope, which was given two components: main slope of the filter strip and cross slope factor. In previously cited papers the filter strips have been oriented strictly up and down slope (with a cross slope component of zero) to encourage shallow uniform flow. In practice, vegetated filter strips tend to have more fully developed drainage patterns that result in concentrated flow.

The characteristics of the nine plots are tabulated below (Table 150). Plots 1 through 6 were on very steep slopes, but as can be seen from the results (Table 151) of the runoff tests, the filter strips with the cross slope, though less steep than the other plots, performed considerably worse than the steeper plots. When reading Table 151, recall that the amount of manure applied to the source area before the simulated rainfall was twice as large in test 2 as in test 1.

Table 150
Experimental Plot Characteristics for VFS Evaluation Study
Dillaha et al., 1988

Plot number	1	2	3	4	5	6	7	8	9
Slope, %	11	11	11	16	16	16	5	5	5
Cross Slope, %	<1	<1	<1	<1	<1	<1	4	4	4
Filter Length, m	9.1	4.6	0	9.1	4.6	0	9.1	4.6	0

The cross slope plots experienced runoff flowing to one side of the plots which moved as if channelized. Flow was generally through a 0.5 to 1 m wide strip (down slope with respect to cross slope) of each filter. Little flow entered other portions of the filter.

Table 151
Percent Reduction of Sediment, Nutrient, and Water Yields by Experimental VFSs Relative to Yields from the Plots without VFSs
Dillaha et al., 1988

Slope %	Test	Filter Length m	TSS %	NH_4 %	NO_3 %	TKN %	Total N %	Total P %	PO_4 %	Total TKN %	Total Filtered P %	Runoff %
11	1	9.1	97	78	9	86	84	88	53	67	56	38
	2		90	59	1	66	62	64	11	67	16	10
	overall		95	69	4	80	70	80	30	67	35	25
11	1	4.6	87	59	-36	80	77	81	9	53	34	10
	2		87	9	-36	31	27	27	-43	50	33	-24
	overall		87	34	-36	64	61	63	-20	51	33	-6
16	1	9.1	91	75	18	90	87	88	47	76	37	24
	2		81	-203	13	53	54	12	-133			-20
	overall		88	-35	17	72	71	57	-51			1
16	1	4.6	82	57	24	80	78	79	34	60	-49	27
	2		65	-139	-53	58	56	15	-228			5
	overall		76	-21	3	69	67	52	-108			16
5x4	1	9.1	60	9	-36	16	14	18	-21			0
	2		55	-11	-1650	3	1	19	44			-1
	overall		58	-11	-158	9	7	19	31			-1
5x4	1	4.6	36	-11	-18	30	28	33	-40			5
	2		20	2	-867	-21	-22	-17	6			-10
	overall		31	1	-82	1	0	2	-3			8

Sediment reduction results for the concentrated flow plots were 58% (for the 9 m strip) and 31% (4.6 m strip), much less than sediment reduction in the uniform flow plots at the steeper slopes. Moreover, even though the sediment loading to the 16% slope uniform flow plot was 3 times higher than the load entering the concentrated flow plot, the concentrated flow plots had sediment losses of 5400 kg/ha and 3200 kg/ha for the short and long filters, respectively, compared to 5600 and 2900 kg/ha for the short and long VFSs on the uniform flow strips.

Filters with concentrated flow were 40 to 60% less effective with respect to sediment removal, 70 to 95% less effective for P, and 61 to 70% less effective for N removal than were uniform flow plots.

This points out the shortcoming of many VFS studies which refer to length of the VFS as the only significant variable. Effective width (which controls depth of flow) is as significant as filter length or slope.

Total P runoff loss from the plots during the first 3 simulations (Test 1) followed the same general trends as sediment loss except that percent reductions in P were generally smaller. The relationship did not hold for Test 2.

Total N was reduced by an average of 67 and 74% by short and long VFSs. Total Kjeldahl nitrogen[12] accounted for approximately 97% of the total N entering the VFSs, and about 90% of the TKN was sediment bound. Including nitrate, about 87% of the total N entering VFSs was particulate.

After passage through the 4.6 and 9.1 m filters, sediment bound TKN accounted for 77 and 80% of the total N leaving the filters, indicating, as with P, that the filters were not as effective in removing soluble species as they were in removing sediment bound species.

Filter effectiveness in removing N deceased as sediment and nutrients built up in the filters. The best removals usually occurred in the first three runs (Test 1).

The authors give three possible explanations for the large amount of nutrients leaving the grassed filters.

(1) They reason that if deposition and filtration of suspended sediments are the predominant mechanisms controlling VFS performance, then VFSs will be more effective in removing larger particles such as soil aggregates, sand, and larger manure particles. The filter effluent will then be enriched with smaller, more easily transported particles, such as

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[12] ammonium N plus organic N

primary clay, silt, and small manure particles. Because these small particles may have a much higher capacity for P sorption than the original soil mass, passage of significant amounts of these particles through the filter may explain considerable P transport in spite of a large decrease in gross sediment transport.

(2) The common assumption that P is predominantly sediment bound is more appropriate for inorganic P than for manure P which mobilizes upon degradation.

(3) Since soil moisture in Test 2 was higher than in Test 1, the deteriorating performance may also have been contributed to by increased runoff caused by decreased infiltration.

The contrast of concentrated flow plots with uniform depth flow plots is telling. The uniform flow plots were located on steep, eroded slopes, a worst case situation. The concentrated flow plots were on much gentler slopes. Still the uniform depth plots were capable of reducing pollution in runoff, though in general, the VFS lengths used in this study were not effective in removing soluble N and P.

The 16% slope plots failed to remove the dissolved species: NH_4, NO_3 and PO_4, on either the 4.6 or the 9.1 m plot, though particulate species were retained (again contributing to the dissolved species' release).

Unlike the 16% slope plots, the 11% slope plots did show some sensitivity to the length of the filter strip.

The difference between Test 1 and 2 showed up in the 11% plots, but not in the other plots. This means that even a functioning filter strip can be overloaded to the point that filtering efficiencies drop. In this case an overload consisted of two weeks of feedlot manure accumulation, hardly an uncommon loading.

The 9.1 m filters on uniform flow plots reduced sediment loss by an average of 91% while the 4.6 m strips reduced sediment loss by 81%. Doubling the filter length only added 10% reduction in sediment yield.

Most sediment removal is localized in the first few meters of the VFSs and in the ponded area just upslope of the VFSs (where decreased runoff velocity results in deposition of the heavier particles and aggregates.)

The authors observed that as the experiment progressed, the ponded area upslope of the filter gradually filled with sediment. After the ponded area was filled, typically sediment would fill a

0.5 m long strip of the filter until much of the vegetation was buried. The runoff would then flow over the horizontal, filled area and deposit sediment below it, with the sediment front gradually moving down the strip.

The 4.6 m filter on 16% slope was filled by sediment by the end of the experiment. Sediment removal in this plot decreased from 90% in the first simulation to 77,66,74, 41, and 53% in the second to sixth simulations.

Some interesting conclusions emerge. A 4.6 m filter strip with concentrated flow essentially passes through all constituents except TSS, which is reduced slightly. The dramatic increase in NO_3 is probably due to NO_3 released from trapped TSS. The 9.1 m strip gave some TSS and Total-P retention, but no Total-N retention.

Dillaha et al. (1985) conclude:

> "1. Vegetative filter strips are ineffective for sediment and nutrient removal under concentrated flow conditions.
>
> 2. Most on-farm filter strips are ineffective because the majority of upland runoff tends to concentrate in natural drainageways before it reaches the filter strip and it then crosses the VFS as channel flow.
>
> 3. The effectiveness of a VFS decreases with time as sediment accumulates within it unless the vegetation can grow as fast as it is being buried.
>
> 4. The phosphorus in runoff from feedlots is not removed as effectively by VFSs as is sediment. Presumably, much of the phosphorus is soluble or associated with fine sediment and manure particles which are not removed effectively by VFSs.
>
> 5. The VFSs were not effective in removing soluble phosphorus from the simulated feedlot runoff. Ortho-P yields from the filters were often higher than the input to the filters."

For our purposes, shallow ground water nutrient measurements would have been a very interesting addition to this study.

B. TREATING CROPLAND RUNOFF

Magette et al. (1987a and b) examined the effectiveness of two lengths of filter strips in reducing nutrient and sediment losses from tilled, fertilized plots.

The experimental plots were located in the Maryland coastal plain on Woodstown sandy loam (Typic Hapludult, mesic, fine siliceous loam). There were nine plots in all, on three different slopes, 3%, 4%, and 5%. On each slope category was (1) a source area with no buffer, (2) a source area with a 4.6 m long fescue buffer, and (3) a source area with a 9.2 m buffer. Each source area consisted of a rota-tilled (parallel to the slope) and fertilized plot, 22 m long, representing conditions typical of the seedbed crop stage. (All widths were 12 meters.)

Simulated rainfall was applied 12 times and fertilizer applied twice according to this schedule:

Before Run 1: liquid nitrogen was surface applied at 112 kg N per ha
Run 1: 4.8 cm simulated rain in 1 hr
Run 2: 24 hours after Run 1, 2.4 cm simulated rain in 0.5 hr
Run 3: 1 hour after Run 2, 2.4 cm simulated rain in 0.5 hr
Runs 4 through 6: 1 week after Run 3, the same rainfall as Runs 1 through 3
Before Run 7: 287 kg N/ha broiler litter was applied (57 kg/ha available N)
Runs 7 through 12: 1 month after Run 6, the same rainfall as Runs 1 through 6

The authors designed their runs to represent a "worst case", growing season situation, two days of heavy rain right after manure application, a dry week, then two more days of heavy rain. The experiment was run first with urea ammonium nitrate fertilizer, then repeated a month later with broiler manure. Neither temperature nor naturally occurring rainfall during the experiment was recorded.

Table 152
Surface Runoff Pollutant Removal by Vegetated Filter Strips of Two Lengths Relative to Losses from Plots with No Filter Strips
Magette et al., 1987

Slope	Filter Length	TSS Removal	Total N Removal	Total P Removal
3%	9.2m	89%	46%	58%
4%[1]	9.2m	80%	51%	-25%
5%	9.2m	56%	22%	21%
mean for 9m strips		75%	35%	20%
3%	4.6m	66%	36%	59%
4%[1]	4.6m	65%	-77%	-101%
5%	4.6m	25%	-13%	34%
mean for 5m strips		42%	-15%	6%

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[1] Some runs for the 4% slope plots were lost, including the control from Runs 7 and 8.

The surface runoff nutrient removal efficiencies relative to the losses from the control plots of the appropriate slope categories (the sources with no buffer) are presented in Table 152.

Negative removal rates were explained by the authors as the result of resuspension of substances caught in the filters. These entries clearly indicate the failure of the shorter filter strip to reduce nutrient export even though, on the average, half the solids were retained.

The surface and subsurface nitrogen losses summed over Runs 1 through 6 (to avoid the problem of lost samples in later runs) are found in Table 153. Similar data for phosphorus were not published.

The negative values for leached N from the unfiltered control 3% plot suggest that the filter strips contributed N to leachate at times, instead of removing it.

It should be remembered that these results are only for the experimental runs and not annual totals. Also, though the dates of the runs were not given, they all occurred within a 2 month period, during the growing season. Filter performance would be expected to vary seasonally because of changes in infiltration, evaporation, and other meteorologically related factors, such as freezing and thawing.

The test results show that the 9 meter filter removed more pollutants than did the 5 meter filter.

Subsurface losses of nitrogen far outweighed surface losses and did not seem to be related to filter length. Surface losses accounted for only from 5 to 25% of the total nitrogen losses.

This points out one of the major ironies of VFS design guidance[13], which, based on the faulty assumption that N and P are for the most part trapped in the soil by biological, physical and chemical processes, often advocates maximum infiltration. When shallow ground water discharges to surface water or acts as recharge to deeper ground water, infiltration into VFSs can be considered a nutrient removal mechanism only to the extent that the

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[13] Some examples of the recommendations concerning the construction of VFSs to ensure adequate infiltration include that filters should not be installed on slopes steeper than 4% and that they be sized for a 1 yr-2 h storm with a minimum 2 h detention time and minimum length of 91 m (Vanderholm et al., 1979). White and Brugger (1981) recommended using 1-8% slopes or parallel contoured infiltration terraces for slopes greater than 8%.

nutrients are absorbed by plants or become trapped in or bound to the soil. The nutrients that are not immobilized are simply delivered to the receiving water by a subsurface route rather than by a surface one.

The authors conclude:

> "1. The performance of vegetated filter strips in reducing nutrient losses from agricultural lands is highly variable.
>
> 2. Vegetated filter strips are more effective in removing suspended solids from runoff than in removing nutrients.
>
> 3. Removals of runoff transported sediment (and perhaps associated chemicals) at the interface between vegetated filter strips and upslope areas may constitute a large percentage of the total amount of sediment prevented from leaving the upslope areas.
>
> 4. Vegetated filter strips appear to be less effective as time goes on in reducing nutrient and suspended solids losses in runoff.
>
> 5. The performance of vegetated filter strips generally diminishes as the ratio of vegetated to unvegetated area decreases.
>
> 6. The effectiveness of vegetated filter strips is highly dependent on the condition of the filter itself.
>
> 7. Subsurface leaching losses can be an important component of inorganic nitrogen movement from agricultural areas. When these losses are considered together with surface losses, the relationship between VFS length and nitrogen removal becomes lost.
>
> 8. Since the ability of VFSs to remove nutrients and suspended solids in this closely controlled experiment was so highly variable, the performance of VFSs in actual use is probably much less than expected.
>
> 9. Vegetated filter strips should not be relied upon as the sole, or even primary means of preventing nutrient movement from agricultural management systems.
>
> 10. Vegetated filter strips are ineffective for removing sediment and nutrients under concentrated flow conditions.
>
> 11. Vegetated filter strips did not remove soluble phosphorus effectively."

Magette et al. recommended that vegetated filter strips not be considered nutrient management techniques by themselves.

The only problem I have with this well-conceived, careful study is that the method for measuring and calculating subsurface losses was not described. From the diagrams of the test site one can infer that subsurface losses were estimated from samples of tile drainage water.

Table 153
Surface and Subsurface Nitrogen Losses From VFSs
Combined Over All Runs
Magette et al., 1987

Plot no.	Slope	Filter Length	Runoff mm	Infilt. mm	Surface N Lost kg/ha	Leached Nitrogen kg/ha	%	Surface &Leached N, kg/ha
4	3%	9.2	73	98	7.6	66.3	90	73.2
5	3%	4.6	126	53	9.4	196.6	95	206.0
6	3%	0	178	3	17.7	-24.4	--	-6.6
1	4%	9.2	72	53	4.0	11.8	75	15.7
2	4%	4.6	116	66	21.6	97.4	82	119.0
3	4%	0	104	73	11.0	88.2	89	99.2
7	5%	9.2	133	66	8.5	96.6	92	105.1
8	5%	4.6	163	32	16.2	60.5	79	76.7
9	5%	0	138	59	18.9	100.0	84	118.9

Though buffer strips are commonly grassed or forested, they need not be. Young et al. (1978) reported the ability of 80 foot long cropped areas to remove pollutants from feedlot runoff. Sediment was reduced 92%, total nitrogen 64%, total phosphorus 59%, and runoff 80%.

III. CHAPTER 3 CONCLUSIONS

Several factors influencing grassed VFS efficiency have been identified, including filter length, depth of flow, slope, and influent characteristics. The literature is too sparse to characterize all the different combinations of factors, however, some important generalizations can be made.

Much of the early work on VFSs was only concerned with surface runoff leaving a filter. The researchers reported high sediment and nutrient reductions. Recent work has shown that 95% of the total nitrogen loss from a VFS (located in the Maryland coastal plain) left the root zone and exited the watershed in subsurface flow. If this observation can be extrapolated to other sites, this means that the earlier studies were touting large reductions in the 5% of the nitrogen that left with the surface runoff, generally giving VFSs too much credit for their ability to remove soluble nutrients.

A management implication of this observation is that VFS use for nutrient removal in areas in close proximity to sensitive surface waters or in areas where shallow ground water nitrogen contamination is a concern may be detrimental to ambient water quality.

VFSs are much better suited to sediment removal from runoff. Total suspended sediment is reduced by 90 to 95 percent in a well functioning VFS. The clay fraction is reduced by roughly 80%.

The lifespan of a VFS in some part is determined by the nature of the sediment load in the influent water. Sediment wedge development and migration was observed when the source area soil was a silt loam, but wedge formation did not occur when the sediments were primarily fine (clay size) particles. A buried VFS is essentially ineffective and can even act as a source of nutrients and solids.

Solids filtering efficiency generally drops from near 90% to roughly 50 or 70% within a single season with wedge formation. Without wedge formation an expected efficiency drop could be from an initial high of 80% to between 20 and 50%, with much variation.

When the flow submerges the vegetation in the filter strip, no filtration is achieved. Such channelization occurs when the runoff source area produces too much flow for the filter or when secondary drainage patterns develop in properly sized filter strips. Cross slope components encourage the concentration of flow.

Vegetative filter strips are ineffective for sediment and nutrient removal under concentrated flow conditions. Most on-farm filter strips are ineffective because the majority of upland runoff tends to concentrate in natural drainageways before it reaches the filter strip and it then crosses the VFS as channel flow.

Uniform flow VFSs with slopes greater than 5% were not effective in removing soluble phosphorus from simulated feedlot runoff. Ortho-P yields from the filters were often higher than the input to the filters. However, total P removal ranged from 10 to 90%, depending on influent loading (10 percent removals occurred with double the manure load).

Though variable, an overall 50% total phosphorus removal from feedlot runoff by uniform flow filters might be expected in a single warm weather season. If the flow is concentrated into channels, expected total phosphorus removal ranges from 0 to 20%.

No estimate of subsurface P losses from VFSs is currently available.

Vegetated filter strips should not be relied upon as the sole, or even primary means of preventing nutrient movement from agricultural management systems.

POSTSCRIPT

The first chapter of this paper gives an encyclopedic treatment of the effects of the major tillage practices on soil properties, hydrologic parameters, and local nutrient and sediment movements. The reader is encouraged to make use of the subject and soils indices when investigating these subjects for specific applications.

The information contained in Chapter 1 may also be of value when modeling BMPs unrelated to tillage. For example, one could find a Manning's n value for a vegetated filter strip in this section.

The times when the various parameters have different values according to tillage are highlighted to prevent exaggerating the differences between tillages. For instance, surface cover varies with tillage from harvest through seedbed stage. For the rest of the year (most of the growing season) this parameter is the same for both tillages.

Some important generalizations emerge from the review. Subsurface nitrogen losses account for upwards of 20% of the annual total fertilizer nitrogen application. This is a substantial amount of nitrogen eventually entering surface water as baseflow.

There have been some confusing claims that no-till produces more runoff than conventional tillage. This can be true for brief periods of time, specifically, during the seedbed stage, when the plowed surface may temporarily generate less runoff than the less disturbed no-till surface. Such differences in runoff volume are not always dramatic or even evident, and are dependent on a number of other factors in addition to tillage (see sections on antecedent moisture, soil bulk density, crusting, macropores).

During the rest of the year, runoff from no-till fields is significantly less than from conventionally tilled fields, primarily due to increased infiltration in no-till soils. Several factors contribute to the increased infiltration, including a rougher, better protected surface, the development of macropore networks over several seasons, and less extensive areas of compacted traffic pan. Be sure to consider the soil type when generalizing on this subject. For example, poorly drained soils may not demonstrate a tillage effect for runoff volume.

No-till surface soils are cooler and wetter in the spring than are conventionally tilled soils. Sometimes this causes delays in planting. On the other hand, no-till soils are considered to be somewhat more drought resistant than conventionally tilled soils. This is due in large part to the reduction in evaporation associated with conservation tillage.

Fertilizer is usually surface broadcast on no-till fields (though various subsurface application options exist). A heavy rain after surface fertilization may result in significant nutrient loss from no-till fields, in amounts exceeding losses from conventionally tilled fields.

Another consequence of the tendency to surface broadcast no-till fertilizer is increased acidity in the top layer of the soil profile. This increased acidity can reduce the cation exchange capacity of the soil and increase the soil's exchangeable aluminum and manganese content. If the soil acidity dips to pH 5.5 or below, the triazine herbicides commonly used in no-till will be deactivated and lime application may be required.

A long term no-till field will have a roughly 20% larger organic nitrogen pool in the top 15 cm, as compared to a conventionally tilled field.

Manure as a source of plant nutrients differs from chemical fertilizer in several important ways. Manure has liquid and solid components. It must be collected, and collection is not 100% efficient. Nutrient losses can occur at a variety of points from excretion to disposal. Because of volatile losses and natural variation in its chemical composition, the nutrient value of manure is difficult to evaluate without testing. A significant portion of the nitrogen and phosphorus is in organic forms which release plant available nutrients at rates influenced by factors other than solubility, such as moisture, temperature, partial pressure, wind speed, redox potential, or microbial biomass.

Conceptually, manure nitrogen and phosphorus are lost at four points during the collection/disposal process: collection, storage, application, and weathering. Relative to fresh manure, collection nitrogen losses can be from 20 to 60%, storage losses from 5 to 15%, and application losses from 10 to 60%. Weathering losses are determined by in-field conditions and meteorology.

Manure management consists of all the activities that transform manure accumulations from a disposal, aesthetic, and health problem to a valuable, soil enhancing resource.

The amount of excreta that can be collected depends on the specific farm operations. If the animals are confined in a barn year long, large fractions of the excreted nutrients potentially can be collected. Wastes excreted while the animals are unconfined (while they are in pasture or on open lots) are generally not collected, and are subject to transport with runoff and subsurface flows. A farmstead model should not neglect the uncollected feces.

Manure storage facilities are sized according to how long the farmer wishes to wait between manure spreading trips. Optimally, the year's manure would be stored until just before spring planting

(365 day storage). Storages of this size are rare because of their expense. More commonly 180 or 90 day storages are encountered. The manure spreading not done on a field immediately before crop planting is wasted in terms of nutrient management, because nutrients not taken up by a plant or immobilized in the soil eventually are washed away in surface runoff or are lost to ground water. Manure spread on snow or frozen ground is all lost with the first thaw.

One type of manure storage facility, the solid/liquid separator, retains solids but discharges liquids. This type of storage cannot be modeled the same way as the total containment structures because it has a discharge. Solid/liquid separators can be retrofit to comply with the concept of a best management practice if the liquid effluent can be contained for later irrigation of a suitable field. Grassed waterways are not appropriate nutrient reduction devices for these effluents.

REFERENCES

Aarstad J. S. and D. E. Miller. 1978. Corn Residue Management to Reduce Erosion in Irrigation Furrows. J. of Soil and Water Cons. pp 289-291.

AAVIM. 1983. Fundamentals of No-Till Farming. American Association for Vocational Instructional Materials, Athens, GA.

Adriano, D. C., P. F. Pratt, and S. E. Bishop. 1971. Fate of Inorganic Forms of N and Salt from Land-Disposed Manure from Dairies. Int. Natl. Symp. Livestock Wastes. ASAE, St. Joseph, MI 49085. p. 243-246.

Alberts, E. E. and R. G. Spomer. 1985. Nitrate Movement in Deep Loess Soils. ASAE Paper 85-2030. American Society of Agricultural Engineers, St. Joseph, MI.

Alberts, E. E. and R. G. Spomer. 1985. Dissolved Nitrogen and Phosphorus Runoff from Watersheds in Conservation and Conventional Tillage. Soil Sci. Soc. Am. J. 40:153-157.

Alberts, E. E.., G. E. Schuman, and R. E. Burwell. 1978. Seasonal Runoff Losses of Nitrogen and Phosphorus from Missouri Valley Loess Watersheds. J. Environ. Qual. 7(2):203-208.

Aldrich, S. R. 1984. Nitrogen Management to Minimize Adverse Effects on the Environment. IN: Hauck, R.D., (ed.) Nitrogen in Crop Production. Madison, WI. American Society of Agronomy, Inc.

Allison, F. E. 1965. Evaluation of Incoming and Outgoing Processes that Affect Soil Nitrogen. IN: Bartholomew, W. V. and F. E. Clark, (eds.) Soil Nitrogen, pp. 573-606. American Society of Agronomy, Madison, WI.

Allmaras, R. R., R. W. Burwell, W. E. Larson, R. F. Holt and W. W. Nelson. 1966. Total Porosity and Random Roughness of the Interrow Zone as Influenced by Tillage. USDA Cons. Rept. No. 7.

Amemiya, M. 1977. Conservation Tillage in the Western Corn Belt. J. of Soil and Water Cons. 32(1):29-36.

Amoozegar-Fard, A., W. H. Fuller and A. W. Warrick. 1975. Migration of Salt from Feedlot Waste as Affected by Moisture Regime and Aggregate Size. J. Env. Qual. 4:468-472.

Andraski, B. J., T. C. Daniel, B. Lowery and D. H Mueller. 1985. Runoff Results from Natural and Simulated Rainfall for 4 Tillage Systems. Transactions of the ASAE 28(4):1219- 1225.

Andraski, B. J., D. H. Mueller and T. C. Daniel. 1985. Phosphorus Losses in Runoff as Affected by Tillage. Soil Sci. Soc. Am. J. 49:1523-1527.

Angle, J. S., G. McClung, M. S. McIntosh, P. M. Thomas and D. C. Wolf. 1983. Nutrient Losses in Runoff from Conventional and No-Tilled Corn Watersheds. Scientific Article No. A3274, Maryland Agric. Exp. Stn., Dept. of Agronomy, College Park, MD.

Angle, J. S., G. McClung, M. S. McIntosh, P. M. Thomas, and D. C. Wolf. 1984. Nutrient Losses in Runoff from Conventional and No-till Watersheds. J. Environ. Qual., 13:130-136.

Angle, J. C., C. M. Gross, and M. S. McIntosh. 1989. Nitrate Concentrations in Percolate and Groundwater under Conventional and No-Till Zea Mays Watersheds. Agriculture, Ecosystems and Environment, FEB. (preprint), Elsevier Science Publishers.

Baker, J. L. and H. P. Johnson. 1983. Evaluating the Effectiveness of BMPs from Field Studies. IN: Schaller R. W. and G. W. Baily (eds.) Agricultural Management and Water Quality, Iowa State University Press, Ames.

Baker J. L., J. M. Laflen. 1982. Effects of Corn Residue and Fertilizer Management on Soluble Nutrient Runoff Losses. Transactions of the ASAE. pp. 344-348.

Baker, J. L., K. L. Campbell, H. P. Johnson and J. J. Hanway. 1975. Nitrate, Phosphorus and Sulfate in Subsurface Drainage Water. J. Env. Qual. 4(3):406-412.

Baldwin, P. L., W. W. Frye and R. L. Blevins. 1985. Effects of Tillage on Quality of Runoff Water. IN: Hargrove W. L., et al., (eds.) Proc. of the 1985 Southern Region No Till Conference. USDA-ARS, Southern Piedmont Conservation Research Center, Watkinsville, GA. p. 169-174.

Banwart, W. L. and J. M. Bremmer. 1976. Evolution of Volatile Sulfur Compounds from Soils Treated with Sulfur Containing Organic Materials. Soil Biol. Biochem. 8:839-443.

Barisas, S. G., J. L. Baker, H. P. Johnson and J. M. Laflen. 1978. Effect of Tillage Systems on Runoff Losses of Nutrients, A Rainfall Simulation Study. Transactions of the ASAE 893-897.

Barker, J. C. and B. A. Young. 1982. Vegetative Filter Treatment of Dairy Wastewater and Lot Runoff in Southern Appalachia. Biological and Agricultural Engineering Dept., North Carolina State University.

Beasley, D. E., E. J. Bomke, E. R. Miller, L. F. Huggins. 1985. Using Simulation to Asses the Impacts of Conservation Tillage on

Movement of Sediment and Phosphorus into Lake Erie. J. Soil and Water Cons. 40(2):233-237.

Beauchamp, E. G., G. E. Kidd, and G. Thurtell. 1978. Ammonia Volatilization from Sewage Sludge Applied in the Field. J. Environ. Qual. 7:141-146.

Beaulac, M.N. and K.H Reckhow, 1982. An Examination of Land Use - Nutrient Relationships. Water Resources Bulletin 16(6):1013-1022.

Bennett, O. L. 1977. Conservation Tillage in the Northeast. J. of Soil and Water Cons. 32(1):9-12.

Bergström, L. and W. Brink. 1986. Effects of Differential Applications of Fertilizer N on Leaching Losses and Distribution of Inorganic N in the Soil. Plant and Soil 93:333-345.

Best Management Practices Report. 1977. Illinois Task Force on Agricultural Nonpoint Sources of Pollution. Illinois Environmental Protection Agency. Springfield.

Bingham, S. C., P. W. Westerman and M R. Overcash. 1980. Effect of Grass Buffer Zone length in Reducing the Pollution from Land Application Areas. Transactions of the ASAE 23(2):330-336,342.

Blevins, R. L., M. S. Smith, G. W. Thomas, and W. W. Frye. 1983a. Influence of Conservation Tillage on Soil Properties. J. Soil and Water Cons. 38(3):301-305.

Blevins, R. L., G. W. Thomas, M. S. Smith, W. W. Frye, and P. L. Cornelius. 1983b. Changes in Soil Properties after 10 Years of Continuous Non-tilled and Conventionally Tilled Corn. Soil Tillage Res. 3:135-146.

Blevins, R. L., G. W. Thomas, and P. L. Cornelius. 1977. Influences of No-tillage and Nitrogen Fertilization on Certain Soil Properties After 5 Years of Continuous Corn. Agron. J. 69:383-386.

Bouldin, D. R., R. L. Johnson, C. Burda and Chun-Weikao. 1974. Losses of Inorganic Nitrogen from Aquatic Systems. J. Environ. Qual. 3:107-114.

Bourg, C. W., et al. 1982. Water Quality Enhancement through Nitrogen and Irrigation Management on Irrigated Corn. Presented at the May 19-21 American Society of Civil Engineers Specialty Conference, held at Lincoln, Nebraska.

Bouwer, H. 1978. Groundwater Hydrology. McGraw Hill Inc., New York, NY. p. 351.

Broadbent, F. E. 1966. Interchange Between Inorganic and Organic Nitrogen in Soils. Hilgardia, 37:165-180.

Brodie, H. L. 1987. Maryland's Livestock Manure Management Program. ASAE Paper 87-4047. American Society of Agricultural Engineers, St. Joseph, MI.

Brown, M. J and W. D. Kemper. 1987. Using Straw in Steep Furrows to Reduce Soil Erosion and Increase Dry Bean Yields. J. of Soil and Water Cons. 42(3):187-191.

Burr, C. A., E. C. Dickey and D. P. Shelton. 1987. Tillage System, Row Spacing and Variety Influences on Soybean Residue Cover.

Burwell, R. E., R. R. Allmaras, and L. L. Sloneker. 1966. Structural Alteration of Soil Surfaces by Tillage and Rainfall. J. Soil Water Conserv. 21:61-63.

Burwell, R. E.,L. L. Sloneker, and W. W. Nelson. 1968. Tillage Influences Water Intake. J. Soil Water Conserv. 23:185-187.

Chang, A. C. and J. M Rible. 1975. Particle-Size Distribution of Livestock Wastes. IN Managing Livestock Wastes, ASAE Proc-275.ASAE, St. Joseph, MI. p. 339-343.

CIBA-GEIGY Co. 1983. Conservation Tillage Guide. Successful Farming. pp 58-60. CIBA, Greensboro, NC.

Ciravolo, T. G., D. C. Mortens, D. L. Hallock, E. R. Collins Jr., E. T. Kornegay and H. R. Thomas. 1979. Pollution Movement to Shallow Groundwater Tables from Anaerobic Swine Waste Lagoons. J. Env. Qual. 8:126-130.

Clark, R. N., A. D. Schnieder, and B. A. Stewart. 1974. Analysis of Runoff from Southern Great Plains Feedlots. ASAE Paper 74-4017. ASAE, St. Joseph, MI.

Clark, R. N., C. B. Gilbertson, and H. R. Duke. 1975. Quantity and Quality of Beef Feedyard Runoff in the Great Plains. IN Managing Livestock Wastes. pp. 429-431. ASAE Proc-275.ASAE, St. Joseph, MI.

Cogo, N. P., W. C. Moldenhauer, and G. R. Foster. 1984. Soil Loss Reduction from Conservation Tillage Practices. Soil Sci. Soc. Am. J. 48:373-396.

Colbourn, P. 1985. Nitrogen Losses from the Field: Denitrification and leaching in intensive winter cereal production in relation to tillage method of a clay soil. Soil Use and Management 1(4):117-120.

Cooper, J. E. 1975. Nitrification in Soils Incubated with Pig Slurry. Soil Biol. Biochem. 7:119-124.

Converse, J. C., C. O. Cramer, G. H. Tenpas, and D. A. Schlough. 1975. Properties of Solids and Liquids from Stacked Manure. IN

Managing Livestock Wastes, Proceedings of the 3rd International Symposium on Livestock Wastes.

Converse, J. C., G. D. Bubenzer, and W. J. Paulson. 1976. Nutrient Losses in Surface Runoff from Winter Spread Manure. TRANS. ASAE 19(3):517-519.

Cramer, C. O., J. C. Converse, G. H. Tenpas, and D. A. Schlough. 1971. The Design of Solid Manure Storages for Dairy Herds. Winter Meeting Am. Soc. Agric. Eng. Paper 71-910.

Crane, S. R. 1978. A Laboratory Investigation of the Short-Term Chemical and Microbial Transformations in the Soil Following Surface Land Application of Poultry Manure. M.S. Thesis, North Carolina State University.

Crawford, H. H., and R. K. Linsley. 1966. Digital Simulation in Hydrology. Stanford Watershed Model IV. TR No. 39. Stanford, California, 210 pp.

Crowder, B. M. and C. E. Young. 1985. Modeling Agricultural Nonpoint Source Pollution for Economic Evaluation of the Conestoga Headwaters RCWP Project. USDA Economics Research Service, Natural Resource Economics Division, USGPO 1985-460-938:20051-ERS.

Dick, W. A. 1983. Organic Carbon, Nitrogen and Phosphorus Concentrations and pH in Soil Profiles as affected by Tillage Intensity. Soil Sci. Soc. Am. J. 47:102-107.

Dickey, E. C., P.S. David, P. J. Jasa, T. R. Peterson. 1984. Soil Erosion from Tillage Systems Used in Soybean and Corn Residues. Transactions of the ASAE 28(4):1124-1129, 1140.

Dickey, E. C. and D. H. Vanderholm. 1980. Performance and Design of Vegetated Filters for Feedlot Runoff Treatment. Livestock Waste: A Renewable Resource: 257-260.

Dillaha, T. A., R. B. Reneau, S. Mostaghimi, V. O. Stanholtz, and W. L. Magette. 1987. Evaluating Nutrient and Sediment Losses from Agricultural Lands: Vegetative Filter Strips. USEPA Region III, CBP/TRS 4/87, Philadelphia, PA.

Dillaha, T. A., J. H. Sherrard, D. Lee, S. Mostaghimi, V. O. Stanholtz. 1985. Sediment and Phosphorus Transport in Vegetative Filter Strips: Phase I, Field Studies. ASAE Paper 85-2043. American Society of Agricultural Engineers, St. Joseph, MI.

Dillaha, T. A., J. H. Sherrard, D. Lee, S. Mostaghimi, V. O. Shanholtz. 1988. Evaluation of Vegetative Filter Strips as a Best Management Practice for Feed Lots. J. Water Pollut. Control Fed., 60(7):1231-1238.

Dixon, J. E., G. R. Stephenson, A. J. Lingg, D. V. Naylor and D. D. Hinman. 1983. Comparison of Runoff Quality from Cattle Feeding on Winter Pastures. Transactions of the ASAE. pp. 1146-1149.

Donigian, A. S., J. L. Baker, and D. A. Haith. 1983. HSPF Parameter Adjustments to Evaluate the Effects of Agricultural Best Management Practices. EPA-600/3-83-066, NTIS PB83-247171.

Doran, J. W. 1980. Soil Microbial and Biochemical Changes Associated with Reduced Tillage. Soil Sci. Soc. Am. J. 44:765-771.

Doran J. W., W. W. Wilhelm and J. F. Power. 1984. Crop Residue Removal and Soil Productivity with No-Till Corn, Sorghum and Soybean. Soil Sci. Soc. Am. J. 48:640-645.

Dornbush, J. N., J. R. Andersen, and L. L. Harms. 1974. Quantification of Pollutants in Agricultural Runoff. EPA-660/2-74-005. U.S.G.P.O. Washington, D. C.

Doyle, R. C., G. D. Stanton and D. C. Wolf. 1977. Effectiveness of Forest and Grass Buffer Strips in Improving the Water Quality of Manure Polluted Runoff. ASAE Paper 77.2501. American Society of Agricultural Engineers, St. Joseph, MI.

Duda, A. M. and D. S. Finan. 1983. Influence of Livestock on Nonpoint Source Nutrient Levels of Streams. Transactions of the ASAE. pp. 1710-1716.

Duke, H. R., et al. 1978. Groundwater Contamination by Fertilizer Nitrogen. J. of Irrigation and Drainage Division. American Society of Civil Engineers 104:283-291.

Ebelhar, S. A., W. W. Frye and R. L. Blevins. 1984. Nitrogen from Legume Cover Crops for No Tillage Corn. Agronomy Journal 76:51-55.

Edwards, W. M., R. R. van der Ploeg, and W. Ehlers. 1979. A Numerical Study of the Effects of Noncapillary-Sized Pores Upon Infiltration. Soil Sci. Soc. Am. J. 43:851-856.

Edwards, W. M. 1982. Predicting Tillage Effects on Infiltration. IN Predicting Tillage Effects on Soil Physical Properties and Processes. ASA, Madison, WI.

Edwards, W. M., L. B. Owens and R. K. White. 1983. Managing Runoff from a Small Paved Beef Feedlot. J. Environ. Qual. 12(2):281-286.

Edwards, W. M. and C. R. Amerman, 1984. Watershed Evaluations of Infiltration Under Conventional and No-Till Corn on Two Ohio Soils. USDA-ARS, Coshocton, OH.

Ehlers, W. 1975. Observations on Earthworm Channels and Infiltration on Tilled and Untilled Loess Soil. Soil Sci. 119:242-249.

El-Haris, M. K., V. L. Cockran, L. F. Elliot and D. F. Bezdicek. 1983. Effect of Tillage, Cropping and Fertilizer Management on Soil Nitrogen Mineralization Potential. Soil Science Soc. Am. J. 47:1157-1161.

Elliot, L. F., G. E. Schuman and F. G. Viets, Jr. 1971. Volatilization of Nitrogen Containing Compounds from Beef Cattle Area. Soil Sci. Soc. Amer., Proc. 35:752-755.

Elliot, L. F., T. M. McCalla, L. N. Mielke and T. A. Travis. 1972. Ammonium, Nitrate and Total Nitrogen in the Soil Water of Feedlot and Field Soil Profiles. Appl. Microbio. 23:810-813.

Ellis, B. G., A. J. Gold and T. L. Loudon. 1985. Soil and Nutrient Runoff Losses with Conservation Tillage. IN: D'Itri, F. M. (ed.) A Systems Approach to Conservation Tillage. Lewis Publishers, Inc., Crelsea, MI.

Endelman, F. R., R. R. Hughes, G. E. P. Box, D. R. Keeney, J. R. Boyle, M. L. Northrup, and P. G. Saffigna. 1974. The Mathematical Modeling of Soil-Water Nitrogen Phenomena. Rept. Oak Ridge Nat. Lab. EDFB-1BP-74-8:66.

Engman, E. T. 1986. Roughness Coefficients for Routing Surface Runoff. ASCE Journal of Irrigation and Drainage Engineering, 112(1):39-53.

EPA. 1976. Quality Criteria for Water. Environmental Protection Agency, Washington, D. C.

Fenn, L. B. and D. E. Kissel. 1974. Ammonia Volatilization from Surface Applications of Ammonium Compounds on Calcareous Soils. II. Effects of Temperature and Rate of Ammonium Nitrogen Application. Soil Sci. Soc. Am. Proc. 38:606-610.

Fisher, D. W., et al., Atmospheric Contributions to Water Quality of Streams in the Hubbard Brook Experimental Forest, New Hampshire. Water Resources Research, (4)1115, 1968.

Forster, D. L., T. J. Logan, S. M. Yaksich and J. R. Adams. 1985. An Accelerated Implementation Program for Reducing the Diffuse-Source Phosphorus Load to Lake Erie. J. Soil and Water Cons. 40(1):136-140.

Frederick, L. R. 1956. The Formation of Nitrate From Ammonium Nitrogen in Soils. II. Effect of Temperature. Soil Sci. Soc. Am. Proc. 20:496-500.

Gast, R. G., W. W. Nelson, and G. W. Randall. 1978. Nitrate Accumulation in Soil and Loss in Tile Drainage Following Nitrogen Application to Continuous Corn. J. Env. Qual. 7: 258-262.

Gerhart, J. M. 1986. Ground-water recharge and its effects on nitrate concentration beneath a manured field site in Pennsylvania. Ground Water 24(4):483-489.

Gerretsen, F. C. 1942. Enkele waarmemingen betreffende den invloed van de temperatuur op de nitrificatie en vast legging van de stikstof. Landbouwk, Tijdschr. 54:573-582.

Gerwing, J. R., A. C. Caldwell and L. L. Goodroad. 1979. Fertilizer Nitrogen Distribution Under Irrigation Between Soil Plant and Aquifer. J. of Env. Qual. 8:281-284.

Gianessi, L. P., H. M. Peskin, P. Crosson, and C. Puffer. 1986. Non-Point Source Pollution: Are Cropland Controls the Answer?. J. Soil and Water Cons. 41(4):215-218.

Gilbertson, C.B., T.M. McCalla, J. R. Ellis, O. E. Cross and W. R. Woods. 1970. The Effect of Animal Density and Surface Slope on Unpaved Feed Yards. Nebraska Agr. Exp.Sta. Bull. 508.

Gilbertson, C. B., J. R. Ellis, J. C. Nienaber, T. M. McCalla, and T. J. Klopfenstein. 1975. Physical and Chemical Properties of Outdoor Beef Cattle Feedlot Runoff. Res. Bull. 271. Agr. Expt. Stn. Univ. of Nebraska, Lincoln, NB. 16 pp.

Gilley, J. E., S. C. Finkner and G. E. Varvel. 1986. Runoff and Erosion as Affected by Sorghum and Soybean Residue. Transactions of the ASAE 29(6):1605-1622.

Gilliam, J.W., R. W. Skaggs and S.B. Weed. 1979. Drainage Control to Diminish Nitrate Loss from Agricultural Fields. J. Env. Qual. 8: 137-142.

Gorham, E., F. B. Martin, and J. T. Litzau. 1984. Acid Rain: Ionic Correlations in the Eastern United States, 1980-1981. Science 225:407-409.

Griffith, D. R., J. V. Mannering and W. C. Moldenhauer. Conservation Tillage in the Eastern Corn Belt. J. of Soil and Water Cons. 32(1):20-28.

Grumbs, F. A. and J. I. Lindsay. 1982. Runoff and Sediment Loss in Trinidad Under Different Crop and Soil Management. Soil Sci. Soc. Am. J. 46:1264-1266.

Guest, R. W., S. D. Klausner, and R. P. March. 1973. 1973 Guidelines for Dairy Manure Management in New York State.

Agricultural Engineering Ext. Bull. No. 400. Dept of Agr. Eng., Cornell University, Ithaca, N.Y.

Hallberg, G. R. 1986. Nitrates in Groundwater. In: Proceedings of Nitrogen and Groundwater Conference. The Iowa Fertilizer and Chemical Association, Ames, Iowa.

Hallberg, G. R. 1986. From Hoes to Herbicides: Agriculture and Ground Water Quality. J. of Soil and Water Cons. 41(6):357-364.

Hamlett, J. M., J. L. Baker, and R. Horton. 1986. Anion Movement under Ridge Tillage: A Field Study. ASAE Paper 86- 2509. American Society of Agricultural Engineers, St. Joseph, MI.

Hamlett, J. M., R. Horton and J. L. Baker. 1987. Water and Nitrate Movement under Ridge Tillage. ASAE Paper 87-2102. American Society of Agricultural Engineers, St. Joseph, MI.

Hargrove, W. L., J. T. Reid, J. T. Touchton, and R. N. Galllaher. 1982. Influence of Tillage Practices on the Fertility Status of an Acid Soil Double-cropped to Wheat and Soybeans. Agron. J. 74:684-687.

Harris, J. R. 1987. Broiler Manure Management and Groundwater Quality. ASAE Paper 87-4048. American Society of Agricultural Engineers, St. Joseph, MI.

Hashimoto, A.C. 1974. Aeration of Poultry Wastes for Odor and N-Control. TRANSACTIONS of the ASAE 17(5):978-986.

Haslam, R. T., R. L. Hershey, and R. H. Keen. 1924. Effect of gas Velocity and Temperature on Rate of Absorption. Ind. Eng. Chem. 16:1224-1230.

Hayes, J. C.; and J. E. Hairston, 1983. Modeling the Long-Term Effectiveness of Vegetative Filters on On-Site Sediment Controls. Am. Soc. Agric. Eng., Paper No. 83-2081, St. Joseph, Mich.

Hsieh, Y. P., L. A. Douglas, and H. L. Motto. 1981. Modeling Sewage Sludge Decomposition in Soil: II. Nitrogen Transformations. J. Environ. Qual. 10(1)59-64.

Heatwole, C. D., A. B. Bottcher and L. B. Baldwin. 1986. Basin Scale Model for Evaluating Best Management Practice Implementation Programs. Transactions of the ASAE. (29)2:439-444.

Hegg, R. O., C. L. Barth, V. L. Quisenberry and W. H. Livingston. 1982. Terraced pasture for disposal of dairy yard runoff. Water Resources Research Institute, Report No. 99, Clemson University, Clemson, SC.

Hensler, R. F., R. J. Olden, O. J. Attoe, W. H Paulson and R. F. Johannes. 1970. Effect of Method of Manure Handling on Crop Yields, Nutrient Recovery and Runoff Losses. Transactions of the ASAE 13(6):726-731.

Hoeft, R. G., D. R. Keeney, and L. M. Walsh. 1972. Nitrogen and Sulfur in Precipitation and Sulfur dioxide in the Atmosphere in Wisconsin. J. Environ. Qual. 1: 203-208.

Holt, R. F. 1979. Crop Residue, Soil Erosion and Plant Nutrient Relationships. J. of Soil and Water Cons. pp. 96- 98.

House, G. J., B. R. Stinner, D. A. Crossley, Jr., E. P. Odum, and G. W. Landale. 1984. Nitrogen Cycling in Conventional and No-tillage Agroecosystems in the Southern Piedmont. J. Soil and Water Cons. (May-June):194-200.

Hubbard, R. K., A. G. Erickson, B. E. Ellis and A. R. Wolcott. 1982. Movement of Diffuse Source Pollutants in Small Agricultural Watersheds of the Great Lakes Basin. J. of Environ. Qual. 11:117-123.

Hubbard, R. K. and J. M. Sheridan. 1983. Water and Nitrate-Nitrogen Losses from a Small, Upland, Coastal Plain Watershed. J. Environ. Qual. 12:291-295.

Hubbard, R. K., L. E. Asmussen and H. D. Allison. 1984. Shallow Ground Water Quality Beneath an Intensive Multiple Cropped System Using Center Pivot Irrigation, J. of Env. Qual. 13:156-161.

Hutchinson, G. L., and F. G. Viets, Jr. 1969. Nitrogen Enrichment of Surface Water by Absorption of Ammonia Volatilized from Cattle Feedlots. Science, 166:514-515.

Johnson, H. P., J. L. Baker, W. D. Shrader and J. M. Laflen. 1979. Tillage System Effects on Sediment and Nutrients in Runoff from Small Watersheds. Transactions of the ASAE 22(5):1110-1114.

Jones, R. E., J. C. Nye, and A. C. Dale. 1973. Forms of Nitrogen in Animal Waste. ASAE Paper no. 73-439, ASAE, St. Joseph, MI.

Joshi, R. C. and D. K. Das. 1977. Study on the Effect of Initial Soil Water Content and Bulk Density on Horizontal Infiltration in a Sandy Loam Soil. J. Indian Soc. Soil Sci. 25: 351-359.

Juo, A. S. R. and R. Lal. 1977. The Effect of Fallow and Continuous Cultivation on the Chemical and Physical Properties of an Alfisol in Western Nigeria. Plant and Soil. 47:567-584.

Kanwar, R. S. J. L. Baker, J. M. Laflen. 1985. Nitrate Movement Through the Soil Profile in Relation to Tillage System and

Fertilizer Application Method. TRANSACTIONS of the ASAE 28(6):1802-1807.

Kanwar, R. S., J. L. Baker, and H. P. Johnson. 1986. Agricultural Chemical Losses through Subsurface Drains from Agricultural Watersheds. Proceedings of International Seminar on Land Drainage. J. Saavalainen, and P. Vakkilainen, EDS; Helsinki University of Technology, Helsinki, Finland. 366-381.

Kanwar, R. S., J. L. Baker, and D. G. Baker. 1987. Tillage and N-Fertilzer Management Effects on Ground Water Quality. ASAE Paper No. 87-2077. St. Joseph, MI.

Khaleel, R., G. R. Foster, K. R. Reddy, M. R. Overcash and P. W. Westerman. 1979a. A Nonpoint Source Model for Land Areas Receiving Animal Wastes III. A Conceptual Model for Sediment and Manure Transport. TRANSACTIONS of the ASAE. 22(6):1353-1361.

Khaleel, R., G. R. Foster, K. R. Reddy, M. R. Overcash and P. W. Westerman. 1979b. Animal Wastes: IV. Model Inputs and Verification for Sediment and Manure Transport. TRANSACTIONS of the ASAE. 22(6):1362-1368.

Khaleel, R., K. R. Reddy and M. R. Overcash. 1979c. Transport of Potential Pollutants in Runoff Water from Land Areas Receiving Animal Wastes: A Review. Water Research 14:421-436.

Khaleel, R. G., K. R. Reddy, and M. R. Overcash. 1979d. Estimating the Changes in Soil Physical Properties Due to Manure Applications: A Review. North Carolina Agricultural Research Service Journal Series Paper No. 6041. 41 pp. Raleigh, NC.

Kilmer, V. J., J. W. Gilliam, J. G. Lutz, R. T. Joyce, and C. D. Eklund. 1974. Nutrient Losses from Fertilized Grassed Watersheds in Western North Carolina. J. Environ. Qual. 3:214-219.

King, L. D. 1973. Mineralization and Gaseous Losses of Nitrogen in Soil Applied Liquid Sewage Sludge. J. Environ. Qual. 2:356-358.

Kitur, B. K., M. S. Smith, R. L. Blevins and W. W. Frye. 1984. Fate of ^{15}N-Depleted Ammonium Nitrate Applied to No-Tillage and Conventional Tillage Corn. Agronomy Journal 76:240-242.

Klausner, S. D., P. J. Zwerman and D. F. Ellis. 1974. Surface Runoff Losses of Soluble Nitrogen and Phosphorus under Two Systems of Soil Management. J. Env. Qual. 3:42- 46.

Klausner, S. D., P. J. Zwerman and D. F. Ellis. 1976. Nitrogen and Phosphorus Losses from Winter Disposal of Dairy Manure. J. Environ. Qual. 5:47-49.

Koelliker, J. K. and J. R. Miner. 1973. Desorption of Ammonia from Anaerobic Lagoons. TRANSACTIONS of the ASAE 16(1):148-151.

Kreis, R. D., R. M. Scalf, and J. F. McNabb. 1972. Characteristics of Rainfall Runoff from a Beef Cattle Feedlot. EPA-R2-72-061. Robert S. Kerr Environ. Res. Lab. U.S.E.P.A. Ada, OK.

Krivak, J. A. 1978. Best Management Practices to Control Non-Point Source Pollution from Agriculture. J. of Soil and Water Conservation. 3(4):161-168.

Laflen, J. M., J. L. Baker, R. O. Hartwig, W. F. Buchele, and H. P. Johnson. 1978. Soil and Water Loss from Conservation Tillage Systems. Trans. Am. Soc. Agric. Eng. 21:881-885.

Laflen, J. M., W. C. Moldenhaur, and T. S. Colvin. 1981. Conservation Tillage and Soil Erosion on Continuously Row-Cropped Land. p121-123. IN Crop Production with Conservaiton in the 80's. ASAE Publication 7-81.

Laflen, J. M. and T. S. Colvin. 1981. Effect of Crop Residue on Soil Loss from Continuous Row Cropping. Transactions of the ASAE 24(3):605-609.

Laflen J. M. and M. A. Tabatabai. 1984. Nitrogen and Phosphorus Losses from Corn-Soybean Rotations as Affected by Tillage Practices. Transactions of the ASAE: 58-63.

Langdale, G. W., H. R. Perkins, A. P. Barnett, J. C. Reardon, and R. W. Wilson, Jr. 1983. Soil and Nutrient Runoff Losses with In-Row, Chisel-Planted Soybeans. J. Soil and Water Cons. 38(3): 297-301.

Langdale, G. W., R. A. Leonard, and A. W. Thomas. 1985. Conservation practice effects on phosphorus losses from Southern Piedmont watersheds. J. Soil and Water Cons. 40(1)157-161.

Landstrom, M. J. and C. A. Onstad. 1984. Influence of Tillage Systems on Soil Physical parameter and Infiltration after Planting. J. Soil and Water Cons. 39:149-152.

Lari, M. B. 1987. Effect of Crop Residues Management on Soil Water Conservation. Research Review Submitted to the Graduate School of University of Maryland in Partial Fulfillment of the Requirement for the Degree of M.S., College Park, MD.

Lauer, D. A., D. R. Bouldin, and S. D. Klausner. 1976. Ammonia Volatilization from Diary Manure Spread on the Soil Surface. J. Environ. Qual. 5:134-140.

Lemon, E. 1977. Gaseous Ammonia Implications to Agriculture and Ecology (unpublished results).

Lietman, P. L., J. R. Ward, and T. E. Behrendt. 1983. Effects of Specific Land Uses on Nonpoint Sources of Suspended Sediment, Nutrients, and Herbicides - Pequea Creek Basin, Pennsylvania 1979-1980. USGS Water Resources Investigations Report 83-4113.

Linden, D. R., C. E. Clapp and W. E. Larson. 1984. Quality of Percolate Water After Treatment of a Municipal Wastewater Effluent by a Crop Irrigation System. J. Env. Qual. 13(2):256-264.

Lindstrom, M. J. and W. B. Voorhees. 1980. Planting Wheel Traffic Effects on Interrow Runoff and Infiltration. Soil Sci. Soc. Am. J. 44:84-88.

Lindstrom, M.H., W. B. Voorhees, and G. W. Randall. 1981. Long-term Tillage Effects in Interrow Runoff and Infiltration. Soil Sci. Soc. Am. J. 45:945-948.

Lindstrom M. J., W. B. Voorhees and C. A. Onstad. 1984. Tillage System and Residue Cover Effects on Infiltration in Northwestern Corn Belt Soils. J. of Soil and Water Cons. 39(1):64-68.

Livingston, W. H. and R. O. Hegg. 1981. Terraced Pasture for Disposal of Dairy Yard Runoff. Proc. 4th International Livestock Waste Symposium, ASAE Publication 2-81, ASAE. St. Joseph, MI 49085. p.270-273.

Loehr, R. C. 1970. Control of Nitrogen from Animal Wastewaters. Proc. of the 12th Sanitary Eng. Cong., Univ. of Illinois. Feb. 1970.

Loehr, R. C. 1974. Characteristics and Comparative Magnitude of Non-Point Sources. J. Water Pollution Control Fed. 46(8):1849-1870.

Loehr, R. A. 1977. Pollution Control For Agriculture. Academic Press, Inc., N.Y. The chapter on animal waste disposal and water quality.

Long, F. L., Z. F. Lund, and R. E. Hermanson. 1975. Effect of Soil-Incorporated Dairy Cattle Manure on Runoff Water Quality. J. Environ. Qual. 4:163-166.

Long, F. L. 1979. Runoff Water Quality as Affected by Surface Applied Dairy Cattle Waste. J. of Env. Qual. 8:215- 218.

Lysyk, T. J., E. R. Easton and P. D. Evenson. 1985. Seasonal Changes in Nitrogen and Moisture Content of Cattle Manure in Cool-Season Pastures. J. of Range Management 38(3)251-254.

Madden, J. M., and J. C. Dornbush. 1971. Measurement of Runoff and Runoff Carried Waste from Commercial Feedlots. IN Livestock Waste

Management and Pollution Abatement. ASAE Proc-271:44-47. ASAE St. Joseph, MI.

Magdoff, F. R., J. F. Amadon, S. P. Goldberg and G. D. Wells. 1977. Runoff from a Low Cost Manure Storage Facility. Transactions of the ASAE. pp 658-660.

Magette, W. L., R. B. Brinsfield, R. E. Palmer, J. D. Wood, T. A. Dillaha, and R. B. Reneau. 1987. Vegetated Filter Strips for Agricultural Runoff Treatment. USEPA Region III, CBP/TRS 2/87, Philadelphia, PA.

Magette, W. L., Palmer R. E. and Wood J. D. 1987. Nitrogen Movement to Ground Water and Surface Water from Poultry Litter. ASAE Paper 87-2022. American Society of Agricultural Engineers, St. Joseph, MI.

Magette, W. L. and R. A. Weismiller. 1985. Nutrient Management for Water Quality Protection. Bay Fact Sheet 3. Agricultural Engineering Dept., University of Maryland, MD.

Manges, H. L., R. I. Lipper, L. S. Murphy, W. L. Powers, and L. A. Schmid. 1975. Treatment and Ultimate Disposal of Cattle Feedlot Wastes. EPA-660/2-75-013. Natl. Environ. Res. Ctr. U.S. E.P.A. Corvallis, OR.

Martin, G. W. and J. T. Touchton. 1983. Legumes as a Cover Crop and Source of Nitrogen. J. of Soil and Water Cons. 38(3):214-216.

Maryland Geological Survey. 1978. Water Resources Basic Data Report No. 10. Maryland Ground-Water Information: Chemical Quality Data. MGS, Baltimore, MD.

Maryland State Soil Conservation Committee. 1987. Maryland Agricultural Water Quality Management Program. MDA 229-87. Maryland Department of Agriculture, Annapolis.

Mathers, A. C. and B. A. Stewart. 1974. Corn Silage Yield and Soil Chemical Properties as Affected by Cattle Feedlot Manure. J. Environ. Qual. 3:143-147.

Mathers, A. C. and B. A. Stewart. 1983. Manure Effects on Crop Yield and Soil Properties. Paper No. 83-2120, ASAE, St.Joseph, MI.

McCaskey, T. A., H. G. Rollins, and J. A. Little. 1971. Water Quality of Runoff from Grassland Applied with Liquid, Semi-Liquid, and Dairy 'Dry' Waste. IN Livestock Waste Management and Pollution Abatement. ASAE Proc-271. ASAE, St. Joseph, MI. pp.239-242.

McDowell, L. L. and L. C. McGregor. 1980. Nitrogen and Phosphorus Losses in Runoff from No Till Soybeans. Transactions of the ASAE 23(3):643-647.

McGregor, K. C. and J. C. Green. 1982. Erosion Control with No-till and Reduced-till Corn for Silage and Grain. Trans. Am. Soc. Agric. Eng. 25:154-159.

McIssac, G. F., J. K. Mitchell and M. C. Hirschi. 1986. The Effect of Contour and Conservation Tillage on Soil Loss from the Tama Silt Loam Soil. ASAE Paper 86-2600. American Society of Agricultural Engineers, St. Joseph, MI.

McMahon, M. A. and G. W. Thomas. 1976. Anion Leaching in Two Kentucky Soils Under Conventional Tillage and a Killed-Sod Mulch. Agron. J. 68:437-442.

Metropolitan Washington Council of Governments, 1978. Land Use/Runoff Quality Relationships in the Washington Metropolitan Area, Occoquan/Four Mile Run Non-Point Source Correlation Study, Prepared by Northern Virginia Planning District Commission and the Department of Civil Engineering, Virginia Polytechnic Institute and State University.

Metropolitan Washington Council of Governments, 1979. Guidebook for Screening Urban Nonpoint Pollution Management Strategies. Northern Virginia Planning District Commission, Falls Church, VA.

Meyer, L.D. and J. V. Mannering. 1961. Minimum Tillage Corn: Its Effect on Infiltration and Erosion. Trans. ASAE 4:72-75 and 86-87.

Meyer, L. D., G. R. Foster, and M. J. M. Römkens. 1970. Soil Detachment, Transportation, and Deposition by Rainfall and Runoff. Amer. Soc. Agr. Eng., Paper No. 70-240, St. Joseph, MI.

Midwest Plan Service. 1985. Livestock Waste Facilities Handbook, 2nd Ed. MWPS-18, Iowa State University, Ames, Iowa 50011.

Miller, R. H. and D. D. Johnson. 1964. The Effect of Soil Moisture Tension on Carbon-Dioxide Evolution, Nitrification, and Nitrogen Mineralization. Soil Sci. Soc. Am. Proc. 28:644-646.

Miller, M. H., J. B. Robinson and D. W. Gallagher. 1976. Accumulation of Nutrients in Soil Beneath Hog Manure Lagoons. J. Environ. Qual. 5:279-282.

Miner, J. R. and R. J. Smith. 1975. Livestock Waste Management with Pollution Control. North Central Regional Research Publication 222, Midwest Plan Service, Iowa State University, Ames.

Minshall, N. E., S. A. Witzel, and M. S. Nichols. 1970. Stream Enrichment from Farm Operations. ASCE J. Sanitary Eng. Div. 96(SA2):513-524.

Mitchell, W. H. and M. R. Teel. 1977. Winter-Annual Cover Crops for No Tillage Corn Production. Agronomy Journal. 69:569-573.

Moldenhawer, W. C., W. G. Lovely, N. P. Swanson and H. D. Currenie. 1971. Effect of Row Grades and Tillage Systems on Soil and Water Losses. J. Soil and Water Cons. 26:192-195.

Möller, E. and H. Schieferdecker. 1985. A Relationship Between Agricultural NH_3 Emissions and the Atmospheric SO_2 Content Over Industrial Areas. Atmospheric Environment. 19(5):695-700.

Moore, I. D. and C. L. Larson. 1979. Estimating micro-relief surface storage from point data. Trans., ASAE 22:1073-1077.

Moore and White. 1983. Criteria for Selecting Manure Handling and Land Application Systems, Dairy Housing II. Am. Soc. Agric. Eng.

Motavalli, P. P., S. D. Confort, K. A. Kelling and J. C. Converse. 1982. Changes in Soil Profile N, P, and K from Injected Liquid Dairy Manure of Fertilizer. Agricultural Waste Utilization and Management. Proc. 5th Int. Sym. on Agricultural Wastes, pp 200-210. American Society of Agricultural Engineers, St. Joseph, MI.

Muck, R. E., R. W. Guest, and B. K. Richards. 1981. Effects of Manure Storage Design on Nitrogen Conservation. ASAE Paper No. 81-2578, ASAE, St. Joseph, MI.

Muck, R. E., and B. K. Richards. 1983. Losses of Manurial Nitrogen in Free-Stall Barns. Agricultural Wastes. 7:65-79.

Muck, R. E., and T. S. Steenhuis. 1982. Nitrogen Losses from Manure Storages. Agricultural Wastes. 4:41-54.

Mueller, M. H., J. B. Robinson and D. W. Gallagher. 1976. Accumulation of Nutrients in Soil Beneath Hog Manure Lagoons. J. Env. Qual. 5:279-282.

Mueller, D. H., T. C. Daniel, B. Lowery and B. Adraski. 1982. The Effect of Conservation Tillage on the Quality of Runoff Water. Paper for Amer. Soc. of Ag. Eng.

Mueller, D. H., R. C. Wendt and T. C. Daniel. 1984a. Soil and Water Losses as Affected by Tillage and Manure Application. Soil Sci. Soc. Am. J. 48:896-900.

Mueller, D. H., R. L. Wendt and T. C. Daniel. 1984b. Phosphorus Losses as Affected by Tillage and Manure Application. Soil Sci. Soc. Am. J. 48:901-905.

Mutchler, C. K., L. L. Mc. Dowell and J. D. Green. 1985. Soil Loss from Cotton with Conservation Tillage. Transactions of the ASAE 28(1):160-163, 168.

Negi, S. C., G. S. U. Raghavan, and F. Taylor. 1982. Hydraulic Characteristics of Conventionally and Zero-Tilled Field Plots. Soil Tillage Research 2:218-292.

Neibling, W. H. and E. E. Alberts. 1979. Composition and Yield of Soil Particles Transported Through Sod Strips. ASAE Paper No. 79-2065, St. Joseph, MI.

Neilson, G. H., J. L. Culley, and D. R. Cameron. 1980. Nonpoint N Runoff from Agricultural Watersheds into the Great Lakes. J. Great Lakes Res. 6(3):195-202.

Nicol, K. H., E. O. Heady and H. L. Madsen. 1974. Models of Soil Loss, Land and Water Use, Spatial Agricultural Structure and the Environment. Center for Agr. and Rural Develop. CARD Rpt. 49T. IOWA State Univ., Ames.

Norstadt, F. A. 1979. Locating Animal Feed Lots and Managing Animal Wastes. IN: Beatty, M. T., et al. (eds.) Planning the Uses and Management of Land. Agronomy 21:733- 762.

Norstadt, F. A. and H. R. Duke. 1982. Stratified Profiles: Characteristics of Simulated Soils in a Beef Cattle Feedlot. Soil Sci. Soc. Am. J. 46:827-832.

Norstadt, F. A. and H. R. Duke. 1985. Soil Profiles: etc. etc. Soil Sci. Soc. Am. J. vol 49, missing page 664 only.

NRC NewsReport. 1986. National Research Council. Protecting Water Quality from Agricultural Chemicals 26(5):9-15.

Nye, J. C. and D. D. Jones. 1978. Experiences with Runoff Control Systems in Indiana. ASAE Paper 78-2574. American Society of Agricultural Engineers, St. Joseph, MI.

Olness, A., S. J. Smith, E. D. Rhoades, and R. G. Menzel. 1975. Nutrient and Sediment Discharge from Agricultural Watersheds in Oklahoma. J. Environ. Qual 4(3):331-336.

Osuji, G. E. and O. Babalola. 1983. Soil Management as it Affects Nutrient Losses from a Typical Soil in Nigeria. J. of Environmental Management 16:109-116.

Overcash, M. R. and F. J. Humenik. 1976. State-of-the-Art: Swine Waste Production and Pretreatment Processes. EPA-600/2-76-290. pp. 170.

Overcash, M. R., S. C. Bingham, and P. W. Westerman. 1981. Predicting Runoff Pollutant Reduction in Buffer Zones Adjacent to Land Treatment Sites. Trans. ASAE 24(2): 430-435.

Owens, L. B. 1987. Nitrate leaching Losses from Monolith Lysimeters as Influenced by Nitrapyrin. J. of Env. Qual. 16:34-38.

Papendick, R. I and L. F. Elliot. 1984. Tillage and Cropping Systems for Erosion Control and Efficient Nutrient Utilization. A. S. A. Spec. Publ. Am. Soc. Agron. Madison Wis. The Society 46:69-81.

Papendick, R. I. and D. E. Miller. 1977. Conservation Tillage in the Pacific Northwest. J. of Soil and Water Cons. 32(1):49-56.

Parker, C. F. and L. E. Sommers. 1983. Mineralization of Nitrogen in Sewage Sludges. J. Environ. Qual. 12:150-156.

Parkin, T. B., and J. J. Meisinger. 1989. Denitrification Below the Crop Rooting Zone as Influenced by Surface Tillage. J. Environ. Qual.:18(1)12-16.

Paterson, J. J., J. H. Jones, F. J. Olsen and G. C. McCoy. 1980. Dairy Liquid Waste Distribution in an Overland Flow Vegetative Soil Filter System. Transactions of the ASAE 23(4):973-984.

Pennsylvania Department of Environmental Resources, 1986a. Swine Manure Management. Harrisburg, PA.

Pennsylvania Department of Environmental Resources, 1986b. Field Application of Manure. Harrisburg, PA.

Pennsylvania Department of Environmental Resources, 1986c. Dairy Manure Management. Harrisburg, PA.

Peterson, R. G., H. L. Lucas, and W. W. Woodhouse, Jr. 1956. The Distribution of Excreta by Freely Grazing Cattle and Its Effect on Pasture Fertility: I. Excretal Distribution. Agronomy J. 48:440-444.

Phillips, R. L. and M. R. Overcash. 1976. Economics and Technology for Controlling Dairy Feedlot Runoff. Paper No. 76-4032. ASAE, St. Joseph, MI.

Phillips, R. L. and M. R. Overcash. 1976b. Dairy Feedlot Hydrology-Southeastern Region. ASAE Paper No. 76-2087. ASAE, St. Joseph, MI.

Phillips, P. A. and J. L. B. Culley. 1985. Groundwater Nutrient Concentrations Below Small Scale Earthen Manure Storages. Agricultural Waste Utilization and Management. Proc. 5th Int. Sym. on Agriculture Wastes, pp. 308-320. American Society of Agricultural Engineers, St. Joseph, MI.

Pierce, F. J. 1985. A System Approach to Conservation Tillage. IN: D'Itri F. M. (ed.) A System Approach to Conservation Tillage. Lewis Publishers Inc., Chelsea and Michigan.

Power, J. F. 1970. Leaching of Nitrate-Nitrogen Under Dry Land Agriculture in the Northern Great Plains. Relationship of Agriculture to Soil and Water Pollution. Cornell University Press, Ithaca, NY.

Power, J. F., J. W. Doran and W. W. Wilhelm. 1986. Uptake of Nitrogen from Soil, Fertilizer and Crop Residues by No Till Corn and Soybean. Soil Sci. Soc. Am. J. 50:137-142.

Pratt, P. F. , S. Davis and R. F. Sharpless. 1976. A Four-Year Field Trial with Animal Manures. I. Nitrogen Balances and Yields. II. Mineralization of Nitrogen. Hilgardia 44:99-125.

Ralph, E. H. 1982. Economics and Environmental Effects of Manure Handling Systems for Northeastern Dairy Farms. J. of the Northeastern Agr. Econ. Council 11(1)45-56.

Randall, G. 1985. Nitrogen Movement. Paper Presented at the Plant Use and the Environment Symposium. Kansas City, Missouri. Agronomy Abstract, American Society of Agronomy, Madison, WI.

Reddell, D. L., W. H. Johnson, P.J. Lyerly, and P. Hobgood. 1971. Disposal of Beef Manure by Deep Plowing. IN Livestock Waste Management and Pollution Abatement. ASAE Proc-271. ASAE, St. Joseph MI. pp. 235-238.

Reddy, K. R., R. Khaleel, M. R. Overcash, and P. W. Westerman. 1977. Evaluation of Nitrogen and Phosphorus Transformations in the Soil-Manure System in Relation to Nonpoint Source Pollution. Agron. Abs. 69th Annual Meetings of ASA, Los Angeles, CA. Nov. 13-18, 1977: 35.

Reddy, K. R., R. Khaleel, M. R. Overcash, and P. W. Westerman. 1979a. A Nonpoint Source Model for Land Areas Receiving Animal Wastes: I. Mineralization of Organic Nitrogen. TRANSACTIONS of the ASAE. 22(4):863-872.

Reddy, K. R., R. Khaleel, M. R. Overcash, and P. W. Westerman. 1979b. A Nonpoint Source Model for Land Areas Receiving Animal Wastes: II. Ammonia Volatilization. TRANSACTIONS of the ASAE. 22(6):1390-1405.

Reddy, K. R., M. R. Overcash, R. K. Khaleel and P. W. Westerman. 1980. Phosphorus Adsorption-Desorption Characteristics of Two Soils Utilized for Disposal of Animal Wastes. J. Env. Qual. 9:86.

Reese, L. E., R. O. Hegg and R. E. Gantt. 1982. Runoff Water Quality from Dairy Pastures in the Piedmont Region. Transactions of the ASAE. pp. 697-701.

Reese, L. E. and T. L. Loudon. 1983. Seepage from Earthen Manure Storages and Lagoons: a Literature Review. ASAE paper no. 83-4569, St. Joseph, MI.

Reicosky, D. C., D. K. Cassel, R. L. Blevins, W. R. Gill and G. R. Naderman. 1977. Conservation Tillage in the Southeast. J. of Soil and Water Cons. 32(1):13-19.

Rice, C. W., and M. S. Smith. 1982. Denitrification in No-till and Plowed Soils. Soil Sci. Soc. Am. J. 46:1168-1173.

Rice, C. W., and M. S. Smith. 1983. Nitrification of Fertilizer and Mineralized Ammonium in No-till and Plowed Soil. Soil Sci. Soc. Am. J. 47:1125-1129.

Rice, C. W., M. S. Smith, and R. L. Blevins. 1986. Soil Nitrogen Availability after Long Term Continuous No Tillage and Conventional Tillage Corn Production. Soil Science Soc. Am. J. 50:1206-1210.

Ritter, W. F., E. W. Walpole and R. P. Eastburn. 1981. An Anaerobic Lagoon for Swine Manure and its Effect on Ground Water Quality in Sandy Loam Soils. Livestock Waste: A Renewable Resource. Proc. 4th Int. Sym. on Livestock Wastes, ASAE Publication. pp. 244-251.

Ritter, W. F. 1984. Influence of Nitrogen Application and Irrigation on Groundwater Quality in the Coastal Plain. Final Report Project 14-08-0001-6833/06, Water Resources Center University of Delaware, Newark, Del.

Ritter, W. F., and A. E. M. Chirnside. 1984. Impact of Land Use on Ground Water Quality in Southern Delaware. Ground Water 22:39.

Ritter, W. F., M. and K. A. Manger. 1985. Effect of Irrigation Efficiencies on Nitrogen Leaching Losses. J. of Irrigation and Drainage Engineering 3(3).

Robbins, J. W. D., D. H. Howells, and G. J. Kriz. 1971. Role of Animal Wastes in Agricultural Land Runoff. Water Poll. Control Res. Series 13020 DGX 08171. U.S.E.P.A. Washington, D.C.

Robbins, J. W. D. 1978. Environmental Impact Resulting from Unconfined Animal Production. Publication No. EPA-600/2-78-046. Office of Research and Development, U.S.E.P.A., Ada. OK.

Rolston, D. E. 1978. Volatile Losses of Nitrogen from Soils. Proc. of National Conference on Management of Nitrogen in Irrigated Agriculture, Pratt P. E., (ed.) pp. 169-194. Department of Soil and Environmental Science, University of California, May 15-18, Riverside, CA.

Römkens, M. J. M., and D. W. Nelson. 1974. Phosphorus Relationships in Runoff from Fertilized Soils. J. Env. Qual. 3:10-13.

Römkens, M. J. M., D. W. Nelson and J. V. Mannering. 1973. Nitrogen and Phosphorus Composition of Surface Runoff as Affected by Tillage Method. J. Env. Qual. 2:292-295.

Ross, I. J., S. Sizemore, J. P. Bowden and C. T. Haan. 1978. Effects of Soil Injection of Liquid Dairy Manure on the Quality of Surface Runoff. University of Kentucky. Water Resources Research Institute. Research Report No. 113, Lexington, KY.

Ross, I. J., S. Sizemore, J. P. Bowden and C. T. Haan. 1979. Quality of Runoff from Land Receiving Surface Application and Injection of Liquid Manure. Transactions of the ASAE. pp. 1058-1062.

Ross, B. B., T. A. Dillaha, S. Mostaghimi and C. D Heatwole. 1987. Rainfall Simulator Studies for Evaluating Agricultural BMP Effectiveness. ASAE Paper 87-2070, American Society of Agricultural Engineers, St. Joseph MI.

Ryther, J. H. and W. N. Dunstan. 1971. Nitrogen, Phosphorus and Eutrophication on the Coastal Marine Environment Science 171:1008-1012.

Sabey, B. R. 1969. Influence of Soil Moisture Tension on Nitrate Accumulation in Soil. Soil Sci. Soc. Am. Proc. 33:263-266.

Safley, L, M., Jr., 1981. An Analysis of a Metal Above-Ground Storage Tank for Handling As-Produced Dairy Manure. IN Livestock Waste : A Renewable Resource. Proc. Int. Symp. Livestock Wastes, p. 410-411. ASAE, St. Joseph, MI.

Schepers, J. S. and D. D. Francis. 1982. Chemical Water Quality of Runoff from Grazing Land in Nebraska: Influence of Grazing Livestock. J. Environ. Qual. 11:351-354.

Schreiber, J. D., and L. L. McDowell. 1985. Leaching of Nitrogen, Phosphorus, and Organic Carbon from Wheat Straw Residues: I. Rainfall Intensity. J. Environ. Qual., 14(2):251-256.

Schreiber, J. D. 1985. Leaching of Nitrogen, Phosphorus, and Organic Carbon from Wheat Straw Residues: II. Loading Rate. J. Environ. Qual., 14(2):256-260.

Schwab, G. O., N. R. Fausey, and D. E. Kopak. 1980. Sediment and Chemical Content of Agricultural Drainage Water.Transactions of the ASAE, 1446-1449.

Schwab, G. O., R. K. Frevert, T. W. Edminster, and K. K. Barnes. 1981. Soil and Water Conservation Engineering. John Wiley and Sons, New York, 524 p.

Sewell, J. I., and J. M. Alphin. 1975. Effects of Agricultural Land Uses on Runoff Quality. IN Animal Waste Management Facilities and Systems. pp. 44-47. Bulletin 548. The University of Tennessee Agricultural Experiment Station, Knoxville, TN.

Sewell, J. I. 1978. Dairy Lagoon Effects on Groundwater Quality. TRANS. ASAE 21:948-952.

Sharpley, A. N., and S. J. Smith. 1989. Mineralization and Leaching of Phosphorus from Soil Incubated with Surface-Applied and Incorporated Crop Residue. Journal of Environmental Quality, 18:101-105.

Shinde, S. S., S. S. Magar and S. P. Kale. 1983. Infiltration as Influenced by Tillage Practice and Antecedent Moisture in Clay Loam Soil. J. of Maharashtra Agricultural Universities 8(2):103-106.

Siemens, J. C., and W. R. Oschwald. 1974. Effect of Tillage System for Corn on Erosion. ASAE Paper 74-2525. American Society of Agricultural Engineers, St. Joseph, MI.

Siemens, J. C. and W. R. Oschwald. 1976. Erosion for Corn Tillage Systems. TRANSACTIONS of the ASAE 19(1):69-72.

Siemens, J. C. and W. R. Oschwald. 1978. Corn-Soybean Tillage Systems: Erosion Control, Effects on Crop Production Costs. Transactions of the ASAE 21(2):293-302.

Sievers, D. M., G. L. Lentz and R. P. Beasley. 1970. Movement of Agricultural Fertilizers and Organic Insecticides in Surface Runoff. Transactions of the ASAE 13:323-325.

Sievers, D. M., G. B. Garner and E. E. Pickett. 1975. A Lagoon-Grass Terrace System to Treat Swine Waste. Proc. 3rd International Livestock Waste Symposium, ASAE Publication PROC-273, ASAE. St. Joseph, MI 49085. p.543-548.

Sims, J. T. 1986. Nitrogen Transformations in a Poultry Manure Amended Soil: Temperature and Moisture Effects. J. Environ. Qual., 15(1):59-63.

Smika, D. E., et al. 1977. Nitrate-N Percolation Through Irrigated Sandy Soil as Affected by Water Management. Agronomy Journal 69:623-626.

Smeins, F. E. 1976. Influence of Vegetation Management on Yield and Quality of Surface Runoff. Annual Report No. C-6310. Texas

Water Resources Institute, Texas A & M University, College Station, TX.

Smika, D. E., et al. 1977. Nitrate-N Percolation Through Irrigated Sandy Soil as Affected by Water Management. Agronomy Journal 69:623-626.

Smith, G. E., F. D. Witaker and H. G. Heinemann. 1974. Losses of Fertilizers and Pesticides from Claypan Soils. Prepared for the Office of Research and Development U. S. EPA. EPA Technology Series 660/2-74-068.

Soil Conservation Service, 1978. Agricultural Waste Management Field Manual. USDA SCS, Chapter 4.

Sommers, L. E. 1977. Chemical Composition of Sewage Sludges and Analysis of Their Potential Use as Fertilizers. J. Environ. Qual. 6:225-232.

Sonzogni, W. C., S. C. Chappa, D. E. Armstrong and T. J. Logan. 1982. Bioavailability of Phosphorus Inputs to Lakes. J. of Env. Qual. 11:555-563.

Srinath, E. C. and R. C. Loehr. 1974. Ammonia Desorption by Diffused Aeration. J. Water Poll. Control 46:1939-1957.

Stanford, G., C. B. England and A. W. Taylor. 1970. Fertilizer Use and Water Quality. ARS 41-168. Agr. Res. Serv., U. S. Dept. Agr., Washington, D. C. pp 1-19.

Stanford, G. and S. J. Smith. 1972. Nitrogen Mineralization Potentials of Soils. Soil Sci. Soc. Am. Proc. 36:465-472.

Stanford, G., M.H. Frere, and D.H. Schwaninger. 1973. Temperature Coefficient of Soil Nitrogen Mineralization. Soil Sci. 115:321-323.

Stark, S. A. and C. E. Clapp. 1977. Potentially Available Nitrogen from Sewage Sludge Applied to Soil. Agron. Abs. 69th Annual meetings of ASA, Los Angeles, CA. Nov 13-18, 1977:152.

Staver, A. W., W. Magette and R. Brinsfield. 1987. The Tillage Effects on Nutrient and Sediment Field Losses. ASAE paper No. 87-2086, American Society of Agricultural Engineers, St. Joseph, MI.

Steenhuis, T. S., G. D. Bubenzer and J. C. Converse. 1976. Ammonia Volatilization of Winter Spread Manure. ASAE Paper No. 76-4515, ASAE, St. Joseph, MI, 49085.

Steenhuis, T. S. 1979. Modeling Nitrogen and Other Nutrient Losses from Winter Spread Manure. Thesis, Department of Agricultural Engineering, University of Wisconsin, Madison, WI.

Stewart, B. A., F. G. Viets, Jr., G. L. Hutchinson, W. D. Kemper, F. E. Clark, M. L. Fairbourn, and F. Strauch. 1967. Distribution of Nitrates and Other Water Pollutants Under Fields and Corrals in the Middle South Platte Valley of Colorado. USDA ARS 41-134.

Stinner, B. R., E. P. Odum, D. A. Crossley. 1983. Nutrient Uptake by Vegetation in Relation to other Ecosystem Processes in Conventional Tillage, No Tillage and Old Field Systems. Agriculture, Ecosystems and Env. 10:1-13.

Stratton, F. E. 1968. Ammonia Nitrogen Losses from Streams. J. Sanit. Eng. Div. Proc. 1085-1092.

Sutton, A. L., D. W. Nelson, I. D. Hoff and V. B. Mayrose. 1982. Effects of Injection and Surface Applications of Liquid Swine Manure on Corn Yield and Soil Compaction. J. Env. Qual. 11:468-472.

Taraba, J. L., I. J Ross, J. D. Bottom and B. J. Barfield. 1983. Effects of Surface Application of Dairy Manure on the Infiltration Rate and Quality of Surface Runoff. Water Resources Research Institute, University of Kentucky, Research Report No. 138. Lexington, KY. NTIS PB-8-3-222422. pp 89.

Tenpas, G. H., D. A. Schlough, C. O.Cramer, and J. C. Converse. 1972. Roofed vs. Unroofed Solid Manure Storages for Dairy Cattle. ASAE Paper No. 72-949. ASAE, St. Joseph, MI 49085.

Thomas, G. W. and R. E. Phillips. 1979. Consequences of Water Movement in Macropores. J. Environ. Qual., 8:149-152.

Thomas, G. W., K. L. Wells and L. Murdock. 1981. Fertilization and Liming. IN: Phillips, R. E., G. W. Thomas and R. L. Blevins (eds.) No Tillage Research. Research Reports and Reviews. College of Agric. and Agric. Exp. Stn., Univ. of Kentucky, Lexington.

Thompson, S. W., H. F. Perkins and G. W. Langdale. 1983. Decomposition of Clover and Wheat Residues under Different Tillage Systems on Severely Eroded Soil. Agronomy Dept., University of Georgia College of Agriculture, Athens, GA.

Timmons, D. R., R. F. Holt and J. J. Latterell. 1970. Leaching of Crop Residues as a Source of Nutrients in Surface Runoff Water. Water Resources Res. 6(1):1367-1375.

Timmons, D. R., R. F. Burwell and R. F. Holt. 1973. Nitrogen and Phosphorus Losses in Surface Runoff from Agricultural Land as Influenced by Placement of Broadcast Fertilizer. Water Resource Res. 9:658-667.

Tollner, E. W. and W. C. Hargrove. 1982. Near-Surface Temperatures Under Two Tillage Treatments. ASAE Tech. Paper 82-1042. Am. Soc. Agr. Eng., St. Joseph, MI.

Tollner, E. W., W. L. Hargrove, and G. W. Langdale. 1984. Influence of Conventional and No-till Practices on Soil Physical Properties in the Southern Piedmont. J. Soil and Water Cons. Jan.-Feb. 73-76.

Triplett, G. B., F. Haghiri and D. M. VanDoren. 1979. Legumes Supply Nitrogen for No Tillage Corn. Ohio Report 64(6):83-85.

Tusneem, M. E. 1970. Nitrogen Transformations in Waterlogged Soil. PhD. Dissertation. Louisiana State University. Baton Rouge, LA.

Tyler, D. D. and G. W. Thomas. 1977. Lysimeter Measurements of Nitrate and Chloride Losses from Soil Under Conventional and No Tillage Corn. J. Env. Qual. 6:63-66.

Unger, P. W. and T. M. McCalla. 1980. Conservation Tillage Systems. Advances in Agronomy. 33:1-58.

Unger, P. W., A. F. Wiese and R. R. Allen. 1977. Conservation Tillage in the Southern Plains. J. of Soil and Water Cons. 32(1):43-48.

USGS. 1984. Nitrate in the Columbia Aquifer, Central Delmarva Peninsula , Maryland. USGS Water Resources Investigations Report 84-4322. GPO, Alexandria, VA.

Utomo, M., W. W. Frye and R. L. Blevens. 1983. Functions of Legume Cover Crops in No Till and Conventional Till Corn Production. Department of Agronomy, University of Kentucky, Lexington, KY.

Vanderholm, D. H. 1975. Nutrient Losses from Livestock Waste During Storage, Treatment, and Handling, IN Managing Livestock Wastes, Proc. Third Int. Symp. on Livestock Wastes, Am. Soc. Agric. Eng.

Vanderholm, D. H., E. C. Dickey, J. A. Jacobs, R. W. Elmere and S. L. Spahr. 1979. Livestock Feedlot Runoff Control by Vegetative Filters. EPA-600/2-79-143. U. S. Environmental Protection Agency.

Varco, J. J., W. W. Frye and M. S. Smith. 1984. Nitrogen Recovery by No Till and Conventional Till Corn from Cover Crops. Department of Agronomy, University of Kentucky, Lexington, KY.

Viets, F. G. Jr. 1971. Fertilizers. J. of Soil and Water Cons. 26:51-53.

Vitosh, M. L., J. F. Davis and B. D. Knezak, 1973. Long-Term Effects of Manure, Fertilizer and Plow Depth on Chemical Properties of Soils and Nutrient Movement in a Monoculture Corn System. J. Environ. Qual. 2:296-298.

Walter, M. F., G. D. Bubenzer, and J. C. Converse. 1974. Movement and Transformation of Manurial Nitrogen through Soils at Low

Temperatures. Sixth National Agricultural Waste Management Conference. Rochester, NY. March 25-27, 1974.

Walter, M. F., T. S. Steenhuis and D. A. Haith. 1979. Nonpoint Source Pollution Control by Soil and Water Conservation Practices. Transactions of the ASAE. 834-840.

Wanielista, M. P., Y. A. Yousef, and W. M. McLellon. 1975. Transient Water Quality Responses from Nonpoint Sources. Paper presented at the 48th Annual Conf. of the Water Pollution Control Fed. Miami Beach, FL.

Watkins, S. H., R. R. Strand, D. S. DeBell and J. Esch, Jr. 1972. Factors Influencing Ammonia Losses from Urea Applied to Northwestern Forest Soils. Soil Sci. Soc. Am. Proc. 36:354-357.

Welch, L. F. 1986. Nitrogen: Ag's Old Standby. Solutions 30:18-21.

Wells, K. L. 1984a. Slowing Nutrient Losses with Covers and Residue. Best Management Practices. Fertilizer Progress. The Fertilizer Institute. Nov./Dec.

Wells, K. L. 1984b. Nitrogen Management in the No Till System. IN: Hauck, R. D., (ed.) Nitrogen in Crop Production Madison, WI. American Society of Agronomy, Inc.

Welty, D. L., A. L. Sutton, J. C. Nye, D. D. Jones, D. I. Kelly and R. L. Vetter. 1984. Fertilizer Value and Nitrogen Losses of Swine and Dairy Manure from an Above Ground Storage System. Agr. Waste Utilization and Management. Proc. 5th Int. Sym. on Agricultural Wastes, pp. 308-321. American Society of Agricultural Engineers, St. Joseph, MI.

Westerman, P. W. and M. R. Overcash, 1980. Dairy Open Lot and Lagoon-Irrigated Pasture Runoff Quantity and Quality. TRANS. ASAE paper no. 80-2305, p. 1157-1164.

Westerman, P. W., M. R. Overcash, and S. C. Bingham. 1983. Reducing Runoff Pollution Using Vegetated Borderland for Manure Application Sites. EPA-600/2-83-022. USEPA, Robert S. Kerr ERL, Ada, OK.

Whitaker, F. D., H. G. Heinemann and R. E. Burwell. 1978. Fertilizing Corn Adequately with Less Nitrogen. J. of Soil and Water Cons. p. 28-32.

White, R. K. and M. F. Brugger. 1981. Handling Milking Center Wastewater to Avoid Pollution. Paper No. AEX706. Ohio Cooperative Extension Service, Columbus, OH.

White E. M., F. R. Gartner and R. Butterfield. 1981. Range Claypan Soil Improvement. Response from Furrowing and Ripping in Northwestern South Dakota. J. Range Manage. 34:119-125.

White, E. M. 1983. Factors Controlling Runoff Nutrient Loss from Cultivated Land in Eastern South Dakota. Water Resour. Bull. 19(5):649-653.

White E. M. 1986. Longevity and Effect of Tillage-Formed Soil Surface Cracks on Water Infiltration. J. of Soil and Water Cons. 41(5):344-346.

Williams, J. D. H., H. Shear and R. L. Thomas. 1980. Availability to Scenedesmus Quadricauda of Different forms of Phosphorus in Sedimentary Materials from the Great Lakes. Limnol. Oceanogr. 25:1-11.

Wise, G. G., and D. L. Reddell. 1973. Water Quality of Storm Runoff from a Texas Beef Feedlot. ASAE Paper 73-441. ASAE, St. Joseph, MI.

Yates, P. and J. M. Sheridan. 1983. Estimating the Effectiveness of Vegetated Floodplains/Wetlands as Nitrate, Nitrite and Orthophosphorus Filters. Agric. Ecosystems Environ. 9:303-314.

Young, R. A. 1974. Crop and Hay Land Disposal Areas for Livestock Wastes. IN Processing and Management of Agricultural Wastes. Proc. Cornell Agricultural Waste Management Conference, Rochester, NY. pp. 484-492.

Young, R. A., and C. K. Mutchler. 1976. Pollution Potential of Manure Spread on Frozen Ground. J. Environ. Qual. 5(2):174-179.

Young, R. A., and C. K. Mutchler. 1976. Pollution Potential of Manure Spread on Frozen Ground. J. Env. Qual. 5:174-179.

Young, R. A., T. Huntrods, and W. Anderson. 1978. Effectiveness of Nonstructural Feedlot Discharge Control Practices. ASAE Paper 78-2578. ASAE. St. Joseph, MI 49085.

Young C. E., B. M. Crowder, J. S. Shortle and J. R. Alwang. 1985. Nutrient Management on Dairy Farms in Southeastern Pennsylvania. J. of Soil and Water Cons. pp. 443-445.

Yuch, Ping Hsieh. 1976. A Quantitative Study of Organic Carbon Decomposition and Nitrogen Transformations in Sewage Sludge-Soil System. Ph.D. Dissertation, Rutgers University, New Brunswick, NJ.

Zwerman, P. J., S. D. Klausner, and D. F. Ellis. 1974. Land Disposal Parameters for Dairy Manure. IN Processing and Management of Agricultural Wastes. Proc. Cornell Agricultural Waste Management Conference. Rochester, NY. pp. 211-221.

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