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HYDROLOGIC AND WATER QUALITY IMPACTS OF PEAT MINING IN NORTH CAROLINA

by

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DISCLAIMER STATEMENT

ABSTRACT

The surface and subsurface hydrologic impacts of peat mining were studied at a pocosin site in the lower Coastal Plain of Northeastern North Carolina. Runoff and water quality data were collected for discharge from field ditches draining sites being actively mined and sites with natural vegetation. A water management model (DRAINMOD) was adapted to simulate surface hydrology. The finite element method was used to evaluate the effects of peat mining on vertical seepage to a deep aquifer and lateral seepage from a nearby lake.

Volume, duration, and peak flow of storm discharge from field ditches was greater from the mining sites than from those having natural vegetation. Baseflow between storm events was greater from vegetated sites than from the mined sites. Reduced evapotranspiration, reduced infiltration capacity and grading and sloping of the surface are most likely responsible for those differences.

Relatively high concentrations of organic sediment in field ditch outflows resulted from the highly erodible state of the mined surface and the field ditch channels. However, much of the sediment load settled in the weir stilling ponds and concentrations decreased downstream in collector and main canels due to setting and dilution. Concentrations of nitrogen and phosphorus varied little among sampling sites but were considerably higher than those reported in outflow from similar sites with natural vegetation in several other studies. Mining appeared to have no significant impact on concentrations of K, Ca, Mg, and Cl in outflows.

The water management model, DRAINMOD, was coupled with a numerical flood routing model and utilized to predict the long-term hydrologic effects of mining the peat bogs and reclaiming them to agriculture. During actual mining, the volume of water entering the canals will be increased by an annual average of about 29% over that from a well-vegetated area. Simulated comparisons of pre- and post-mining hydrology revealed that post-mining hydrology of land under cultivation will not be much different from pre-mining hydrology except that improved water management can be exercised

Data from drill holes to a depth of 17.8m at 22 locations at the peat mining site were used to describe the underlying sediments. Utilizing the finite element method, the deep seepage rate to the Castle Hayne aquifer was estimated to be 0.039 cm/yr and was not altered by mining. Mining had no effect on total seepage out of Lake Phelps.

TABLE OF CONTENTS

	Page
ACKNOWLEDGMENTS	iii
ABSTRACT	v
TABLE OF CONTENTS	vii
LIST OF FIGURES	xi
LIST OF TABLES	xix
SUMMARY AND CONCLUSIONS	xxi
RECOMMENDATIONS	xxv
I. INTRODUCTION	1
LITERATURE CITED	3
II. PEAT RESOURCES AND THE POTENTIAL HYDROLOGIC IMPACTS OF PEAT MINING AND RECLAMATION ACTIVITIES - A LITERATURE REVIEW	4
WHAT PEAT IS AND IS NOT	4
TYPES OF PEAT DEPOSITS	5
NORTH CAROLINA PEAT DEPOSITS	5
WORLD PEAT RESOURCES AND USE	7
PEAT MINING FOR ENERGY PRODUCTION	8
HYDROLOGY OF NATURAL AND ALTERED PEATLAND	9
Water Movement in Natural Peatlands	10
Water Quality in Natural Peatlands	13
Changes in Water Movement with Artificial Drainage. .	15
Changes in Water Movement with Land Clearing.	16
Changes in Water Movement with Artificial Drainage and Land Clearing	17
Changes in Water Movement with Peat Mining	17
Changes in Water Movement with Reclamation	17
Changes in Water Quality with Artificial Drainage . .	18
Change in Water Quality with Artificial Drainage and Land Clearing	20
Changes in Water Quality with Peat Mining	20
Changes in Water Quality with Agricultural Reclamation	20
Changes in Water Quality with Forest Reclamation. . .	22
LITERATURE CITED	23

III. EFFECTS OF PEAT MINING ON RUNOFF AND WATER QUALTIY . . 29

INTRODUCTION 29

PROCEDURES 29

Site Description 29
Sampling Site Selection 40
Hydrologic Measurements 42
Water Quality Measurements. 47

RESULTS AND DISCUSSION 49

Runoff 49
Water Quality 56

SUMMARY AND CONCLUSIONS 77

LITERATURE CITED 78

IV. COMPUTER MODELS FOR ANALYSIS OF HYDROLOGIC EFFECTS OF PEAT MINING 80

INTRODUCTION 80

MODEL DEVELOPMENT 82

Field Hydrology Model 82
Flood Routing Model. 87

MODEL TESTING 93

Introduction 93
Input data 93
Results and Discussion 98
Conclusions 110

ANALYSIS OF HYDROLOGIC EFFECTS OF PEAT MINING 112

Comparison of Pre-Mining and Post-Mining Hydrology . . 112
Hydrologic Effects of Alternative Post-Mining Drainage Systems 132
Hydrologic Effects of Alternative Post-Mining Crop Rotations 145
Analysis of Hydrologic Effects During the Mining Process 153
Analysis of Hydrologic Effects of Mining Strategy . . 163

RECOMMENDATIONS 168

SUMMARY 171

LITERATURE CITED. 173

V. EFFECTS OF PEAT MINING ON GROUNDWATER PROCESSES 175

INTRODUCTION. 175

PHYSICAL PROPERTIES OF MINERAL SEDIMENTS 175

Introduction 175
Procedures 176
Results and Conclusions 178

ANALYSIS OF GROUNDWATER MOVEMENT 193

Introduction 193
Procedures 193
Results and Discussion 196

SUMMARY AND CONCLUSIONS 212

LITERATURE CITED 214

LIST OF FIGURES

		Page
Figure 1.	General location of permitted peat mining area	30
Figure 2.	Permitted peat mining area	31
Figure 3A.	Natural vegetation on the mine site . . .	33
3B.	White cedar logs removed from the peat during canal construction.	33
Figure 4.	Drainage system characteristics	35
Figure 5.	Drainage flow pattern in the peat mine area.	36
Figure 6A.	Field ditch on site with natural vegetation.	37
B.	Field ditch on site prepared for peat harvesting	37
Figure 7.	Location of pilot mining area and sampling sites	38
Figure 8.	Location of sampling sites for runoff and water quality in field ditches on the pilot mining area	39
Figure 9.	Portion of permitted mine area burned by wildfire in 1981.	41
Figure 10.	Aftermath of 1981 wildfire.	43
Figure 11A.	Installation of weir in flashboard riser. .	44
11B.	Completed weir installation with trash screen.	44
Figure 12A.	Complete runoff measurement installation. .	45
B.	Silt screen installed in field ditch. . . .	45
Figure 13.	Organic sediment and wood accumulated in field ditch and stilling pond, (A) without silt screen (B) with silt screen.	46
Figure 14.	Weir installation on DeHoog Canal	48
Figure 15.	Runoff from field ditches draining cleared and vegetated sites (4 ha in size) for a storm in December, 1981	50

Page

Figure 16. Runoff from field ditches draining cleared and vegetated sites (4 ha in size) for a storm in January, 1982 51

Figure 17. Runoff from field ditches draining cleared and vegetated sites (4 ha in size) for a storm in February 1982 52

Figure 18. Runoff from field ditches draining cleared and vegetated sites (4 ha in size) for a storm in August 1982. 53

Figure 19. Monthly rainfall at the peat mining site. . 55

Figure 20. Permitted peat mining area on First Colony Farms 81

Figure 21. Layout of a typical stretch of main canal on First Colony Farms permitted mining area 88

Figure 22. Effect of limiting channel depth. 91

Figure 23. Steady upward flux as a function of the water table depth for the pre- and post-mining soils. 96

Figure 24. Drainage volume as a function of the water table depth for the before, during, and after mining soils. 97

Figure 25. Observed and simulated water table depth comparison from site 103 for 1977 99

Figure 26. Observed and simulated water table depth comparison from site 103 for 1978 100

Figure 27. Observed and simulated water table depth comparison from site 103 for 1979 101

Figure 28. Observed versus simulated monthly flow volumes per unit area for 24 months 104

Figure 29. Observed and simulated hydrograph comparison from site 103 for model validation 105

Figure 30. Observed and simulated hydrograph comparison from site 103 for model validation 106

Page

Figure 31.	Observed and simulated hydrograph comparison from site 103 for model validation. Also shown is the simulated hydrograph forced to fit the observed by using model calibration	107
Figure 32.	Observed and simulated hydrograph comparison from site 103 for model validation. . .	109
Figure 33.	Observed and simulated flow duration curves of collector canal outflows from site 103 for model validation	111
Figure 34.	Before and after mining water table depth comparison for a typical First Colony Farms field for a year of average rainfall . . .	115
Figure 35.	Comparison of actual evapotranspiration for before and after mining. Values are monthly averages from 20 years of simulation	116
Figure 36.	Comparison of flow volumes per unit area for before and after mining. Volumes are monthly averages from 20 years of simulation	118
Figure 37.	Comparison of collector canal outflow with main canal outflow on a per unit area basis	119
Figure 38.	Comparison of hourly, routed collector canal outflows with average daily flows from the field edge as determined by DRAINMOD, converted to equivalent units for the before mining condition	120
Figure 39.	Comparison of before and after mining flow duration curve. Values are average daily flows, as determined by DRAINMOD, converted to collector canal outflows. Twenty years of daily values are represented for each condition	122
Figure 40.	Comparison of before and after mining hydrographs for a 200 day return period peak flow..	124
Figure 41.	Comparison of before and after mining hydrographs for a 100 day return period peak flow..	125

Page

Figure 42. Comparison of before and after mining hydrographs for a 75 day return period peak flow.. 126

Figure 43. Comparison of before and after mining hydrographs for a 50 day return period peak flow.. 127

Figure 44. Comparison of before and after mining hydrographs for a mid-summer storm . 129

Figure 45. Comparison of before and after mining hydrographs for a growing season storm producing no surface runoff. 131

Figure 46. After mining water table depth comparison for a typical First Colony Farm mining site for a year of average rainfall. Comparison is between good and poor subsurface drainage. 133

Figure 47. Comparison of monthly flow volumes per unit area between good and poor subsurface drainage conditions. Values are monthly averages from twenty years of simulation. 134

Figure 48. Flow duration curve comparison between good and poor subsurface drainage treatments. Values are average daily flows simulated by DRAINMOD. 135

Figure 49. Flow duration curve comparison between good and poor subsurface drainage treatments with the horizontal scale limited to less than 6% time equaled or exceeded 136

Figure 50. Average monthly flow volume from subsurface drainage and surface runoff for the post-mining condition with poor subsurface drainage 138

Figure 51. Average monthly flow volume from subsurface drainage and surface runoff for the post-mining condition with good subsurface drainage.. 139

Figure 52. Good and poor subsurface drainage hydrograph comparison for post-mining for a 200 day return period flow. Eleven blocks (1430 ha) contribute flow to each hydrograph 140

Page

Figure 53.	Good and poor subsurface drainage hydrograph comparison for post-mining for a 100 day return period flow. Eleven blocks (1430 ha) contribute flow to each hydrograph.	141
Figure 54.	Good and poor subsurface drainage hydrograph comparison for post-mining for a 75 day return period flow. Eleven blocks (1430 ha) contribute flow to each hydrograph.	142
Figure 55.	Good and poor subsurface drainage hydrograph comparison for post-mining for a 50 day return period flow. Eleven blocks (1430 ha) contribute flow to each hydrograph.	143
Figure 56.	Good and poor subsurface drainage hydrograph comparison for the post-mining condition. Eleven blocks (1430 ha) contribute flow to each hydrograph.	144
Figure 57.	Average monthly flow rates per unit area for five different crop rotations for good subsurface drainage	147
Figure 58.	Average monthly flow rates per unit area for five different crop rotations for poor subsurface drainage.	148
Figure 59.	Average monthly evapotranspiration for five different crop rotations for the post-mining condition with good subsurface drainage	150
Figure 60.	Average monthly evapotranspiration for five different crop rotations for the post-mining condition with poor subsurface drainage..	151
Figure 61.	Average monthly rainfall from Elizabeth City.	152
Figure 62.	Before and during mining water table depth comparison for 1960	155
Figure 63.	Before and during mining average monthly flow volumes per unit area..	158
Figure 64.	Comparison of subsurface drainage and surface runoff during mining	159

Page

Figure 66. Comparison of flow duration curves for before and during mining. Values are average daily flows from DRAINMOD converted to collector canal outflows. 160

Figure 67. Comparison of flow duration curves for before and during mining. Values are average daily flow rates from DRAINMOD converted to collector canal outflows. The horizontal scale is limited to less than 6% time equaled or exceeded. 162

Figure 68. Hydrograph comparison of effects of one, three or six actively mined blocks contributing to flow in DeHoog Canal. All other blocks are in before mining condition. This is for a 75 day return period peak flow. 164

Figure 69. Hydrograph comparison of effects of one, three or six actively mined blocks contributing to flow in DeHoog Canal. All other blocks are in before mining condition. This is for a 50 day return period peak flow. 165

Figure 70. Comparison of DeHoog hydrographs between having six blocks in active mining condition and having the same six blocks in reclaimed condition with good subsurface drainage for a 75 day return period peak flood flow. 166

Figure 71. Comparison of DeHoog Canal hydrographs between having six blocks in active mining condition and having six blocks in reclaimed condition with good subsurface drainage for a 50 day return period peak flood flow. 167

Figure 72. Location of field borings along Allen, Boerema, Clayton, DeHoog, and Evans roads . 177

Figure 73. General description of the sequence of the mineral sediments below the peat deposits . 179

Figure 74. Symbols used to describe the profile along Allen, Boerema, Clayton, DeHoog, and Evans roads 180

Figure 75. Description of the profile along Allen road 181

		Page
Figure 76.	Description of the profile along Boerema road	182
Figure 77.	Description of the profile along Clayton road.	183
Figure 78.	Description of the profile along DeHoog road.	184
Figure 79.	Description of the profile along Evans road	185
Figure 80.	Particle size distributions for samples 1 through 5	188
Figure 81.	Particle size distributions for samples 6 through 10	189
Figure 82.	Particle size distributions for samples 11 through 15	190
Figure 83.	Particle size distributions for samples 16 through 21.	191
Figure 84.	Typical cross section of the mining area. Drainage canals divide the surface of the land into rectangular blocks	194
Figure 85.	Location of First Colony Farms peat mining operation	195
Figure 86.	Profile description along transect A-B. . .	197
Figure 87.	Boundary conditions assumed along transect A-B	200
Figure 88.	Boundary conditions assumed along transect C-D	201
Figure 89.	Equipotentials lines for the case of no drainage canals with the water table ponded at the surface	204
Figure 90.	Equipotentials in meters referenced to the top of the Castle Hayne aquifer for the pre-mined case along transect A-B	205
Figure 91.	Equipotentials in meters referenced to the top of the Castle Hayne aquifer for Case 4 where one block is mined in transect A-B. .	206

Page

Figure 92. Equipotentials in meters referenced to the top of the Castle Hayne aquifer for Case 5 where four adjacent blocks are mined in transect A-B. 207

Figure 93. Equipotentials in meters referenced to the top of the Castle Hayne aquifer for Case 7 where all blocks are mined in transect A-B. 209

LIST OF TABLES

Page

Table 1. Water budget for the Albemarle-Pamlico Peninsula. 11

Table 2. Average concentrations of nonfiltrable solids in drainage waters. 58

Table 3. Average turbidity of drainage waters 59

Table 4. Comparison of nonfiltrable solids concentration in a weekly grab sample vs. a stormflow sample collected the same week 62

Table 5. Average concentration of NH_4-N in drainage waters 63

Table 6. Average concentration of $NØ_3$-N in drainage waters 64

Table 7. Average concentration of total Kjeldahl N in drainage waters 65

Table 8. Average concentration of total phosphorus in drainage waters. 68

Table 9. Average concentration of orthophosphate in drainage water 69

Table 1Ø. Average pH in drainage waters. 7Ø

Table 11. Average concentration of K in drainage waters 71

Table 12. Average concentration of Ca in drainage waters 72

Table 13. Average concentration of Mg in drainage waters 73

Table 14. Average concentration of Cl in drainage waters 75

Table 15. Average concentration of dissolved oxygen in drainage waters 76

Table 16. Summary of soil property, drainage system and crop parameter inputs to DRAINMOD for organic soil (site 1Ø3) prior to mining. . . 94

Table 17. Collector canal specifications for input to flood routing model 95

Page

Table 18. Monthly totals of observed and simulated flow volumes from site 103. 102

Table 19. Summary of soil property and drainage system and crop parameter inputs to DRAINMOD for post mining analysis 113

Table 20. Main canal (DeHoog Canal) specifications for input to flood routing model 123

Table 21. Julian day (JULD) - rooting depth (ROOTD) relationship for the five different crop rotations in the form required as input to DRAINMOD. 146

Table 22. Yearly average water loss for five different crop rotations and two different drainage treatments 149

Table 23. Yearly average evapotranspiration for five different drainage treatments. 149

Table 24. Summary of soil property and drainage system parameter inputs to DRAINMOD for during mining analysis 153

Table 25. Bulk density values of mineral layers described below the peat deposits 186

Table 26. Saturated hydraulic conductivities measured by the piezometer method 192

Table 27. Aquifer hydraulic conductivity obtained from USGS. 192

Table 28. Soil layers and their hydraulic conductivities 198

Table 29. Boundary conditions considered in evaluating the effect of mining on groundwater flow . . 202

Table 30. Groundwater flow rates to the Castle Hayne aquifer for the various boundary conditions assumed. 203

Table 31. Flow rates of water seeping out of Lake Phelps for various boundary conditions assumed. 211

SUMMARY AND CONCLUSIONS

A review of the literature on the peat resources of North Carolina and the potential hydrologic impacts of peat mining and reclamation activities was conducted. About half of the estimated 260,000 ha of peatland in North Carolina occurs in relatively large deposits (1,500 - 93,00 ha in size) in pocosins. Most of that area and many of the smaller deposits in floodplains and Carolina Bays are potentially exploitable as an energy source. Mining of peat will require intensive surface drainage and maintenance of the surface in a fallow condition. Adverse hydrologic changes may result. Reclamation of the site to agriculture or other uses will result in hydrologic conditions similar to those currently found for those land uses on pocosin mineral or shallow organic soils.

Field studies of the effects of peat mining on runoff and water quality were conducted at the pilot peat mine of First Colony Farms, Creswell, North Carolina. The 6000 ha site that has been permitted for mining is located on the Albemarle - Pamlico Peninsula south of Lake Phelps. Preparation for surface mining of peat included drainage system installation and chopping of the vegetation and mixing it into the peat surface. Runoff and water quality data were collected at the ends of field ditches draining fields being actively mined and fields in which the surface vegetation between the field ditches was left undisturbed.

Volume, duration, and peak flow of storm discharge from field ditches was greater from the mining sites than from those having natural vegetation. Complete, accurate hydrographs could not be obtained for the largest storms because of flooding of the weirs. However, peak flows from the smaller storms were generally 5-10 times higher from the mined sites than from the vegetated sites. Baseflow between storm events was greater from vegetated sites than from the mined sites. The hydrologic changes noted resulted from a combination of factors. Removal of the vegetation reduced evapotranspiration. Grading and sloping of the surface associated with reduced infiltration capacity increased the volume and rapidity of overload flow to the ditches.

The milling and mixing processes used to prepare the peat surface for harvest produced a surface of high erodibility. Concentration of nonfiltrable solids and turbidity of water flowing over the weirs was considerably higher in the cleared fields than in those remaining in natural vegetation. The weir stilling ponds rapidly filled with organic sediment and wood fragments, much of it the result of erosion from the channels themselves. Silt fences placed in the ditches were ineffective in controlling sediment discharge but the stilling ponds effectively served as sediment basins and prevented much of the sediment from leaving the field ditch. Sediment concentrations were significantly lower downstream in collector and main canals, the result of both settling and dilution.

Chemical quality of field ditch outflow varied little among sites. Outflows were characterized by low pH except where outflow from a recently burned area was part of the total flow. The concentrations of nitrogen and phosphorus from all sites were considerably higher than those reported in other studies on similar sites with natural vegetation.

Concentrations of K, Ca, Mg and Cl were somewhat higher in the main canals than in the field ditches, particularly where outflow from the burned area was present. Concentrations in the field ditches were similar in magnitude to those reported in other studies.

A water management computer model (DRAINMOD) was adapted for use on the First Colony Farms permitted peat mining area. Water loss from the fields predicted by DRAINMOD was taken as lateral flow input to a numerical flood routing model which routed the water down the collector canals. The outflows from the collector canals were, in turn, routed down the main canals to the watershed outlet.

The models were tested against measured water table depths and collector canal flow rates and were found to perform very favorably.

DRAINMOD was used to predict the long-term hydrologic effects of mining the peat bogs and reclaiming them to agriculture on a per unit area basis. The DRAINMOD - flood routing model combination was used to determine actual canal flow rates at the mining area boundary for storms of varying magnitudes for a series of different land conditions.

Simulated comparisons of "pre" and "post" mining hydrology revealed that the post-mining hydrology will not be substantially different from pre-mining hydrology except that improved hydrologic control will be available to the farmer. On average, the water lost from the fields, down the canals, should be less after mining. This is because increased rooting depths, from commonly used crop rotations on the reclaimed lands, would increase evapotranspiration and thus increase average water table depths and reduce average subsurface drainage. Adding good subsurface drainage to the reclaimed areas could increase base flow rates and slightly increase total annual flow but would decrease peak flood flows.

During actual mining the volume of water entering the canals by overland flow will be substantially increased by an annual average of approximately 140%. Total water lost to the canals by subsurface as well as surface drainage will be increased by an annual average of about 29%.

Data from drill holes to a depth of 17.8 m at 22 locations on the peat mining site on First Colony Farms were used to describe the sediments beneath Allen, Boerema, Clayton, DeHoog, and Evans roads. Five mineral layers were continuous between most of the drill holes and were considered to have the major influence on groundwater movement. The sediments found below the peat deposits were a transition zone consisting of organic and sandy material, sandy loam, sandy loam with fossils, and two types of silty clay loam.

The finite element method was used to evaluate the effects of peat mining on groundwater flow. Deep seepage to the Castle Hayne aquifer and horizontal seepage from Lake Phelps were determined by solving the Laplace equation. The deep seepage rate to the Castle Hayne aquifer for the pre-mining condition was 0.039 cm/year. Three mining situations (mining occurring on a single block, four adjacent blocks, and on all blocks along a transect) were simulated to determine the effect of mining on deep seepage. Due to deep beds of slowly permeable lenses, the predicted deep seepage rates were found not to be significantly altered by the mining process.

The horizontal seepage rates from Lake Phelps were used in a mass balance to calculate a recharge rate over the entire lake area. The total predicted seepage out of Lake Phelps, under post-mining, agricultural conditions was 0.193 cm/year. The lateral seepage rates from the lake were approximately 2% of the vertical seepage rates through the bottom of the lake. Deepening the canals by 1 m increased the total predicted seepage out of Lake Phelps by less than 0.2%, and had no effect on the deep seepage to the Castle Hayne aquifer.

RECOMMENDATIONS

Surface Hydrology

1. Recommendations for the Mining and Reclamation Process:

   (a) To minimize total and peak flows from a mining site, the active mining area should be kept to a minimum. Also mined areas should be reclaimed as quickly as possible (i.e. revegetated).

   (b) Installing good subsurface drainage in the reclaimed fields will distribute the water loss more evenly over the year reducing the high flows and potentially increasing crop yields.

   (c) Planting a cover crop over winter months on the reclaimed fields will increase the evapotranspiration over this period and, on average, reduce runoff.

2. Recommendations for Water Quality Management:

   (a) Additional research utilizing flow-proportional automatic samples should be conducted to compare nitrogen and phosphorus concentrations with that from similar undisturbed areas and to determine the influence of a mining operation on the nitrogen and phosphorus concentrations downstream.

   (b) Utilize sediment basins in field ditches and collector canals draining a site being actively mined to control discharge of organic sediment.

   (c) Keep active mining area to a minimum and revegetate as quickly as possible.

3. Recommendations for improvement of Surface Hydrologic Model:

   (a) The computations for water storage in the canal banks and its subsequent release to the canals needs to be improved.

   (b) The linkage between the field scale model and the flood routing model should be up-graded to allow for downstream effects in the canals to influence the field scale processes.

   (c) Field experiments to improve the understanding of the infiltration characteristics of the peat soils need to be conducted.

4. Recommendations for Future Analyses using the Hydrologic Models:

   (a) Modeling of the hydrology of forested areas on reclaimed land would assist in more accurately describing the after mining conditions.

   (b) The models' capability should be expanded to model blocks without any drainage ditches. This is required in order to model the large areas on the Albemarle-Pamlico Peninsula, containing harvestable peat, essentially still in a native condition.

Groundwater Processes

5. Recommendations for Groundwater Model Improvement:

   (a) The modeling process should be expanded to three dimensions to consider lateral seepage out of the mining area.

   (b) Boundary conditions for the model should be altered to consider the piezometric head of the aquifers at the boundaries.

   (c) Improved estimates of the conductivities, depths, and thicknesses of the aquifers should be obtained.

   (d) Further hydraulic conductivity tests should be conducted for the surficial layers to more intensively cover the mining area.

   (e) Due to the very low conductivities of the surficial sediments, the recharge to the deep aquifers and the seepage from Lake Phelps were found to be very small. The effects of peat mining on these factors are essentially negligible indicating little impact of peat mining itself on the groundwater processes. Note, however, that this recommendation does not cover groundwater pumping for process water or site dewatering.

I. INTRODUCTION

Though it has occurred for centuries in other parts of the world, peat mining is new to North Carolina. Peat is only one of many possible energy supplements receiving greater attention in recent years with the unprecedented rise in conventional fuel costs. Peat deposits are located in many different areas of the eastern Coastal Plain of North Carolina (Otte and Ingram, 1980). However, the largest deposits are found in the pocosin wetlands of the Albemarle-Pamlico peninsula, an area already under considerable pressure from other development interests such as recreational home development, agriculture, and forestry. Artificial drainage and clearing for agriculture and timber production have occurred thoughout the region since the 1700's but never before at today's intensity. Because the deep peats have proven to be relatively poor for agriculture and are somewhat difficult and expensive to prepare for timber production, many pocosin areas have remained relatively undeveloped. However, in addition to providing an energy source, peat mining provides an economically feasible way of exposing the more fertile mineral soils underlying the peatlands for improved crop and timber production.

The estuarine system surrounding the Albemarle-Pamlico Peninsula is the largest in any single state on the Atlantic Coast (Street and McClees, 1981). These highly productive ecosystems are thought to be particularly sensitive to the hydrologic changes caused by wetland development. Many feel that water quality degradation in the form of reduced surface water salinities, nutrient overenrichment and bacterial contamination has already had a serious impact on the coastal fishing industry in the region (Pate and Jones, 1981; Street and McClees, 1981; NCDNRCD, 1982 b). These problems have stimulated much research and controversy since the mid-1970's.

In 1981, the Governor appointed a Coastal Water Management Task Force to study this issue and recommend solutions. Among other things, this group recommended (in December, 1982) that comprehensive water management plans be developed for the area and that additional research and monitoring efforts be carried out. Another group, the Peat Mining Task Force, was organized within the North Carolina Department of Natural Resources and Community Development in 1980 to address the peat mining issue. In its report, released in 1983, this Task Force concluded that existing permit programs, if properly coordinated, are generally adequate for managing peat mining operations. However, the need for continuing monitoring and research, as a basis for permit decisions, is highlighted in the report.

In 1980, First Colony Farms, Inc. received a permit to mine peat on a 6,000 ha tract located in Hyde, Washington, and Tyrrell Counties, North Carolina. The research presented here was conducted to evaluate the hydrologic impacts of that mining operation. Objectives of the project were to:

(1) Measure drainage, water quality, and quantity parameters for a peat-mining site and compare the results to similar parameters for drained sites with natural vegetation or other potential land uses.

(2) Develop a model to predict the effects of peat mining and subsequent agricultural activities on the hydrology of the area. Then use the model to evaluate the hydrologic effects of peat mining on the 6,000 ha First Colony Farms mining site.

(3) Evaluate the effects of peat mining at the First Colony Farms site on groundwater recharge and water movement from Lake Phelps.

(4) Design strategies to prevent, reduce, or ameliorate any adverse hydrologic or water quality impacts.

The research has been conducted in three different phases to accomplish those objectives: studies of runoff and water quality, analysis of groundwater movement and development of simulation models.

Field measurements of hydrologic parameters included runoff, rainfall, temperature, and on-site and laboratory analyses of standard water quality parameters. Hydrologic changes in surface soils resulting from mining were determined by comparing runoff (in field ditches) from fields being actively mined to runoff (in field ditches) from fields with natural vegetation. Runoff was also measured in a main canal to provide data for model testing. Water quality sampling was conducted at field ditch outlets and at several different locations throughout the drainage system.

The objectives of the groundwater study were to classify the mineral sediment layers beneath the peat deposits and to evaluate the effects of peat mining on groundwater flow. First, field borings were made to determine the characteristics of the sedimentary strata in the mining area. Bulk densities and hydraulic conductivities of various mineral layers were measured. Next, the effects of mining and subsequent reclamation on groundwater processes in the mining site were analyzed by solving the Laplace equation for steady state flow conditions. A numerical finite element model was used to characterize groundwater movement for various mining scenarios and conditions. The effects on subsurface flow to drainage canals, groundwater movement from Lake Phelps, and seepage to deep aquifers were studied.

Mining the peat in an area the size of the First Colony Farms site would take place over many years and it would be extremely difficult and time consuming to measure all of the hydrologic effects directly. Also, by the time a detrimental effect was observed in the field, preventative measures would be too late. Computer models were developed to simulate the water balance in the soil profile and then route the surface and subsurface runoff through the canal network to an outlet. By simulating the different soil profiles (before, during, and after mining), over the same period of climatological record, the effects of mining on the hydrology were predicted. The effects of mining on the water yield from individual storms of varying magnitudes (eg. 50, 100, and 200 day recurrence intervals) can also be easily compared and differences related to such factors as water table depth. Various mining strategies and after mining reclamation practices may also be compared for hydrologic effects and recommendations made in advance of the actual occurrence.

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II. PEAT RESOURCES AND THE POTENTIAL HYDROLOGIC IMPACTS OF MINING AND RECLAMATION ACTIVITIES-A LITERATURE REVIEW

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WHAT PEAT IS AND IS NOT

The terms peat, organic soil, and Histosol often seem to be used interchangeably in the literature. According to Soil Conservation Service terminology, organic soils contain a minimum of 12 percent or 20 percent organic carbon depending on the annual period of saturation and clay content (Soil Survey Staff, 1981). Histosols contain over 50 percent organic matter in the surface 80 cm (Soil Survey Staff, 1975). To receive serious consideration for energy production, peat must be almost 100 percent organic matter and have relatively high bulk density. Fibrous peats that are very low in bulk density are not suitable. Increasing mineral fractions decrease heating values and increase ash production. North Carolina peat is highly decomposed, usually with less than 10 percent and often with less than 5 percent ash (Ingram and Otte, 1981b). Thus, when discussing peat for energy production, many organic soils and Histosols are not included.

There is also a problem with using some of the standard definitions for peat. The Soil Conservation Service defines peat as a "raw undecomposed organic material in which the original fibers constitute almost all the material" (Soil Survey Staff 1981). Most North Carolina peats are fine-grained and highly decomposed (Otte and Ingram, 1980) and would be called muck by the standard Soil Conservation Service terminology for organic soil materials (Soil Survey Staff, 1981).

Classification methods for Histosols have received less attention than those of other soil types. The present system used to classify Histosols in the U.S. is provisional, having undergone relatively little testing compared to other soil orders (Soil Survey Staff, 1975). In general, Histosol suborders are defined by moisture regime and the degree to which the organic matter has decomposed. Under this system, most North Carolina peats would be included in the Saprist (most decomposed) suborder and the Medisaprist great group (Lilly, 1981). However, some peats in the state would qualify as Fibrists (least decomposed) or Hemists (intermediate decomposition).

Histosols can form in virtually any climate as long as sufficient water is present (Soil Survey Staff, 1975). Histosols are saturated or nearly saturated with water part of the year. Saturation with water results in slow rates of organic matter decomposition and allows organic materials to accumulate over time. Because this is much the same way that coal bed evolution begins, peat is often referred to as "young coal".

In conclusion, the term peat, as used in this report, refers to fuel quality organic deposits and highly decomposed Histosols composed of at least 80 percent organic matter.

TYPES OF PEAT DEPOSITS

The most commonly recognized peat varieties were described as follows by Punwani (1980), based upon the vegetation from which they were formed and the extent of decompostion. These varieties are similar to the Histosol suborders described by the Soil Conservation Service (Soil Survey Staff, 1975). Fibric peats (peat moss) are composed of sphagnum and other mosses; are the least decomposed and most fibrous; and though they make good soil conditioners, are poor fuels. Hemic peats (reed-sedge), derived from reeds, sedges, swamp plants, and trees, have a lower fiber content and are more decomposed. Sapric peats (humus) are the most decomposed, least fibrous type and make the best fuels. From a hydrologic standpoint, Moore and Bellamy (1974) listed three main types of peat formations: (1) ombrotrophic peats, which are recharged exclusively by rainfall, with no influence from flowing groundwater; (2) rheotrophic peats which are influenced by flowing groundwater originating from outside the immediate catchment area; and (3) mesotrophic peats, which represent a transitional stage between the first two types. Much of the literature refers to only two types: the ombrotrophic type or bog and the minerotrophic type or fen, which includes all peats influenced by groundwater flow. These hydrologic terms may also be applied to wetlands that do not have significant peat deposits.

NORTH CAROLINA PEAT DEPOSITS

Otte and Ingram (1980) identified three main geologic types of peat deposits in coastal North Carolina: (1) pocosins (an Indian term for swamp-on-a-hill) which are formed in broad shallow depressions or broad flat interstream divides on "uplifted" sea floors; (2) river floodplain deposits; and (3) Carolina Bay deposits, which are formed in eliptical depressions of unknown origin. The major peat deposits in North Carolina are in pocosins (Otte and Ingram, 1980).

There appears to be some controversy in the literature concerning the definition of a pocosin. The term pocosin usually refers to a fairly well-defined type of plant community. However, pocosin vegetation can occur in a variety of geological settings in the Coastal Plain on mineral or organic soils. The dominant peat type associated with pocosins is the ombrotrophic type formed on blocked drainage systems on interstream divides (Otte, 1982). However, pocosin vegetation, associated with mineral or organic soils, may also form in Carolina Bays; in ridge and swale systems associated with old dune or beach ridge systems on the lower Coastal Plain terraces; and in seeps, springs and margins of slow-flowing streams in the Sandhills region (Christensen, et al., 1981; Otte, 1982). In the latter two cases, the pocosins may be rheotrophic or minerotrophic in that they may be influenced by groundwater (Christensen, et al., 1981). The Carolina Bays appear, in many cases, to be in typical successional stages somewhere between lakes and bogs (Christensen, et al., 1981).

Pocosins are characterized by a very dense growth of mostly broadleaf evergreen shrubs and scattered pond pine (*Pinus serotina*) growing on wet, acid, nutrient poor, mineral and organic soils (Richardson, et al., 1981). Under natural conditions, the soils in these areas are often saturated or flooded for extended periods, particularly in the winter and early spring, and are burned on an average of every 15 to 20 years (Richardson, et al., 1981). The soils of undisturbed pocosins on the Albemarle-Pamlico peninsula are rarely saturated all the way to the surface except during periods of high rainfall. Surface water seeps rapidly through the porous root mat to natural drainage outlets or to ditches and canals in the area. Relatively high evapotranspiration rates, even in winter, quickly remove excess soil water from the zone of dense roots near the surface. Water table depths on deep organic soils normally vary from about 10 cm to about 60 cm in winter and may drop to 1 m or more in late summer (Skaggs et al., 1980). Under the wetland classification system developed by Cowardin, et al. (1979) and now in use by the U.S. Fish and Wildlife Service, pocosins are classified as Palustrine systems of the scrub-shrub class and a mixed broad-leaved and needle-leaved evergreen subclass. Other modifiers under this classification scheme would include a saturated water regime, acid waters, and organic or mineral soils.

Pocosins occur on wetlands. There is also general disagreement about the types of ecosystems to which the term wetlands should be applied. Wetlands are extremely diverse and the boundary between wet and dry environments is diffuse. Regulations on dredge and fill activities in wet environments that are based on the legal definition and the demarcation of wetlands have resulted in much controversy among the factions on either side of the issue.

To minimize confusion, in this report we shall use a broad, ecologic definition of wetland that is readily available in the literature (Cowardin, et al., 1979). In terms of the concept outlined by the U.S. Fish and Wildlife Service, "Wetlands are lands where saturation with water is the dominant factor determining the nature of soil development and the types of plant and animal communities living in the soil and on its surface." The wetland definition refers to hydrophytic plants and hydric soils and lists of those plants and soils are under development for the wetlands of North Carolina.

> Wetlands are lands transitional between terrestrial and aquatic systems where the water table is usually at or near the surface or the land is covered by shallow water. For purposes of this classification wetlands must have one or more of the following three attributes: (1) at least periodically, the land supports predominantly hydrophytes; (2) the substrate is predominantly undrained hydric soil; and (3) the substrate is nonsoil and is saturated with water or covered by shallow water at some time during the growing season of each year (Cowardin et al., 1979).

According to Richardson, et al. (1981), pocosins covered more than 800,000 ha of the lower Coastal Plain of North Carolina in 1962, representing more than 15 percent of the total land area of the counties involved. The largest pocosin areas are on the Albemarle-Pamlico peninsula. Peat deposits in this region reach almost 5 m in depth, with an average depth of about 1.2 m (Ingram and Otte, 1982).

Ingram and Otte (1982) have found that the peats on the Albemarle-Pamlico peninsula are dominated by two types of sapric peat. A black sapric peat, most prevalent in the upper 90 to 120 cm of the deposit, is fine grained and highly decomposed. Dominated by colloidal sized material, the peat resembles axle grease when wet. In the deeper deposits, a brown, more fibrous sapric peat is usually found beneath the black sapric peat. Many logs, roots, and stumps of fallen Atlantic white cedar (*Chamaecyparis thyoides*) and cypress (*Taxodium distichum*) trees, well preserved under anaerobic conditions, may be found at all depths in the peat deposit. The advanced decomposition and large quantity of buried wood distinguish these peats from many other deposits in this country and elsewhere (Peat Methanol Associates, 1983). A transition zone, normally less than 30 cm thick, separates the peat from the underlying mineral soil (Ingram and Otte, 1982).

Otte (1982) concluded that peat formation on the North Carolina Coastal Plain began between 10,000 and 12,000 years ago with the blockage of local drainage systems, the cause of which is not yet known. The altered position of sea level at the time (due to global climatic variations) is not thought to have been a major factor in peatland development. This conclusion is based upon ^{14}C dating of the deposits, the apparent remoteness of these deposits from the probable ocean shoreline at the time formation began, and the fact that peat formation did not occur over the entire Coastal Plain.

WORLD PEAT RESOURCES AND USE

Though estimates of world peat resources vary considerably, it appears that peatlands cover between 1 and 2 percent of the earth's surface, with approximately 95 percent of the total occurring in Europe, Asia, and North America (Punwani, 1980). The Soviet Union contains approximately half of the world's potential peat supply, and the United States ranks second or third with roughly 10 percent of the total. Among the states, Alaska contains over 50 percent and Minnesota, Michigan, and Wisconsin combined, account for more than 25 percent of the nation's peat resources. North Carolina ranks seventh among the states with about 2 percent of the nation's resources.

Peat has long been used for energy production in the U.S.S.R., Finland, and Ireland. Until recently, use by most other nations has been limited to horticulture and agriculture.

PEAT MINING FOR ENERGY PRODUCTION

Recent oil price increases have stimulated interest in mining peats in the United States for energy production. Some of the proposed mining operations in this country have been quite large in scope. The Minnesota Gas Company (Minnegasco) once proposed mining 80,000 ha of Minnesota peat for conversion to methanol (Boffey, 1975). A current proposal by Peat Methanol Associates (PMA) would convert 2,000 metric tons of North Carolina peat per day, from a 6,000 ha tract, into 747 m^3 of methanol (NCDNRCD, 1983b).

More than 200,000 ha of North Carolina's peat resource appear to be exploitable as an energy source (NCDNRCD, 1983b). Peats on the Albemarle-Pamlico Peninsula have an average heating value of around 24 million J/kg thus comparing favorably with eastern coal at 27 million to 30 million J/kg (Campbell, 1981; Ingram and Otte, 1982). Peat produces less sulphur and ash than coal but requires over four times the volume of coal to produce the same heat (Campbell, 1981). Due to its greater bulk, peat becomes most competitive with coal when transport distances are minimized.

Over the past three years, five mining permits have been issued for approximately 10,500 ha of peatland in North Carolina, representing about 4 percent of the total area of exploitable peat deposits in the state (NCDNRCD, 1983b). First Colony Farms holds a permit (NCDNRCD, 1980) for the largest of these tracts (6,000 ha) but has thus far conducted only small scale, experimental mining operations. The new proposal by PMA would greatly intensify mining operations on this 6,000 ha tract and construct a facility for converting the peat into methanol (PMA, 1983).

The unique character of North Carolina peats has dictated many modifications to previously accepted mining techniques used elsewhere. First Colony Farms has pioneered much of this work over the past few years with its experimental mining operation. PMA (1983) has drawn heavily on this experience in developing the proposed mining approach outlined in the following paragraphs:

Construction of an adequate artificial drainage system is the initial phase of peat bog preparation. The bog is first divided into 130 ha blocks, approximately 1.6 km long and 0.8 km wide. A series of small field ditches 50 m apart would subdivide these blocks into 4 ha mining strips and drain into larger collector canals. These collector canals, in turn, drain perpendicularly into still larger main canals which conduct drainage waters toward the Pamlico Sound. This is similar to the existing drainage system in the pilot mining area.

Following drainage system installation, an assortment of machinery is used to: (1) grind up surface vegetation and incorporate it into the surface peat layer; (2) grind up wood imbedded in the peat surface; and (3) shape the mining strips such that there is approximately a 1 percent slope from the center of the strip to the field ditch. Careful field shaping and smoothing is needed to encourage maximum surface water runoff and eliminate surface ponding.

With favorable weather conditions, the bog should then be ready for mining. First Colony Farms has experimented with both sod and milled peat mining methods (Campbell, 1981). The sod peat method utilizes a tractor attachment which excavates relatively wet peat material by cutting a series of small furrows in the surface and extruding the excavated material into compressed cylinders approximately 8 cm in diameter. After drying sufficiently in the sun, these cylinders are collected and stockpiled. Careful monitoring of surface moisture conditions is essential to determining when the peat has the proper consistency for sodding.

In contrast, the milling method involves removing a thin layer of dried granular peat from the peat surface. Drying and collection are facilitated by first grinding, fluffing, and turning the thin surface layer. This is the mining method selected by PMA for future mining operations.

PMA has estimated that weather conditions will be favorable for mining on an average of 142 days per year (from mid-March through mid-November). Approximately six contiguous 130 ha blocks will be mined at a time. Upon completion of mining activities on a six-block area, that area will be reclaimed and mining will commence on a new area.

At the anticipated pace of mining activities, mining of the 6,000 ha peat deposit would be completed in approximately 28 years.

HYDROLOGY OF NATURAL AND ALTERED PEATLANDS

Much concern has arisen in recent years relative to the potential environmental impacts of converting wetlands in the lower Coastal Plain of North Carolina to other uses. Other uses are normally not possible without some hydrologic modification achieved with surface and/or subsurface drainage systems. Research has shown that these modifications and the particular activities that follow will cause changes in both the quantity and quality of local water resources.

It must be noted that wetland development is not a new idea. Lilly (1981) described the extensive development which has occurred in North Carolina swamp lands beginning as early as 1700. He observed that "essentially all the swamp lands in the state have been logged at least once and have had some drainage imposed, either for agricultural development or to aid logging and reforestation."

The following discussion of the potential impacts of peatland development, based upon the available literature, will be prefaced by a characterization of natural peatland hydrology.

Water Movement in Natural Peatlands

Before proceeding further, it will be helpful to review the basic elements of the hydrologic cycle in the form of a water budget equation:

$$P = ET + GW + R$$

where P is precipitation, ET is evapotranspiration (evaporation plus transpiration), GW is groundwater outflow to deep aquifers and R is runoff. Runoff includes water entering ditches, canals, or streams from overland flow, unsaturated flow of soil water, and saturated flow from the near-surface ground water. Heath (1975) emphasized the fact that precipitation was the only element which had been accurately measured in the Albemarle-Pamlico region (before 1975) and that the other elements must be estimated from other areas or from basic hydrologic principles. In view of the yearly variability of all the water balance elements, Dooge (1975) stressed the need for long-term data when attempting to characterize the hydrology of any system.

Knowledge of natural peatland hydrology in the southeast is relatively limited due to the inaccessibility of these areas and the fact that most were altered long before a significant interest in their natural processes arose (Heath, 1975). Also, most of the available literature on peatland hydrology comes from research in northern latitudes where peat has formed under considerably different conditions.

Based on data from Heath (1975), the estimated average water budget for the Albemarle-Pamlico peninsula under natural conditions indicates that ET is a dominant factor in removal of water from pocosin ecosystems (Table 1).

Table 1. Average Water Budget for the Albemarle-Pamlico Peninsula

Element	Amount (cm)	Percent of Precipitation
Precipitation	130	100
Evapotranspiration	91	70
Groundwater outflow	1	1
Runoff (by Subtraction)	38	29

Precipitation

Precipitation is the only source of recharge for the ombrotrophic peatlands of the Albemarle-Pamlico peninsula. Due to the high variability of the rainfall component, hydrologic conditions within the peatlands might be expected to vary accordingly. The isolation of these systems from the stabilizing influence of regional groundwater flow is well documented and will be discussed later. Though annual rainfall averages about 130 cm in the region, the longest precipitation records available (for New Holland, N.C.) show that annual rainfall may vary up to 50 cm above or below this average in any given year (Heath, 1975).

Evapotranspiration (ET)

The actual ET occurring in natural peatlands of the Albemarle-Pamlico region may be difficult to predict. Heath (1975) suggested that, for all practical purposes, actual ET is the same as potential ET, except during extreme droughts. From his study of Minnesota bogs, Verry (1982) reported ET losses near potential values, based on 3 year averages. However, others have reported that ET may be significantly less than potential values in a given year, depending upon variations in rainfall and near surface groundwater levels (Bay, 1967; Dooge, 1975; Daniel, 1981). Daniel (1981) concluded that both the amount of rainfall and its distribution within the year are important. Higher annual ET can be expected if most of the annual rainfall occurs during the summer and fall, when higher potential ET and lower groundwater levels reduce the amount of runoff. If winter and spring are the wetter seasons, lower potential ET and higher groundwater levels will result in more runoff and less annual ET. Daniel (1981) reported that annual ET was only about one-half of potential for his study watersheds on the Albemarle-Pamlico peninsula during 1978. During that year, ET amounted to roughly 40 percent of precipitation while around 60 percent was lost as runoff. Total precipitation for the year was approximately average.

Based upon his studies in the U.S.S.R., Dooge (1975) concluded that ET from bogs is less than that from non-marsh areas. However, others have indicated that ET from peat may be higher than that from mineral soils (Eggelsmann, 1975b; Clark and Clark, 1979). Boelter and Verry (1977) predicted higher ET rates in peatlands of the Northern Lake States covered by scrub and sedge vegetation than in forested peats due to higher surface winds and the generally greater biomass of transpiring plants.

Groundwater

The hydrologic behavior of peatlands is closely related to the position of the groundwater table beneath the surface (Bay, 1967; Dooge, 1975; Eggelsmann, 1975b; Skaggs, et al., 1980; Daniel, 1981). The extremely low hydraulic conductivity of peat and some of the fine textured mineral layers beneath it essentially isolates the shallow groundwater in the peat horizons from the deeper groundwater system in underlying mineral layers (Bay, 1967; Maki, 1974; Heikurainen, 1975; Eggelsmann, 1975b; Clark and Clark, 1979; Skaggs, et al., 1980; Daniel, 1981). In a Minnesota study, Bay (1967) found no correlation between bog water tables and fluctuations in deeper underground storage. Clark and Clark (1979) concluded that "recharge is generally less from wetland areas than from other areas." Eggelsmann (1975b) stated that peats in general have low water retention capacity since the pore spaces are almost always filled with water. Downward water movement is further inhibited by a relatively impermeable mineral horizon underlying the organic soils of many pocosins (Maki, 1974; Daniels, et al., 1977).

The extent of decomposition, as measured by bulk density and fiber content, determines the hydraulic conductivity and storage capacity of a peat material (Baden and Eggelsmann, 1963; Boelter, 1964, 1969; Dooge, 1975; Verry and Boelter, 1975; Skaggs et al., 1980; Daniel, 1981). In North Carolina, Badr and Skaggs (1978) reported hydraulic conductivities of only about 0.02 m/day in the decomposed subsoil, while Badr (1978) estimated the hydraulic conductivity of the undecomposed surface root mat to be 240 m/day. Boelter (1964) found that slightly decomposed surface peats in Minnesota bogs hold 95 to 100 percent water by volume at low tensions while more decomposed peats deeper in the soil profile hold 80 to 90 percent water under much greater tensions.

Daniel (1981) observed the artesian character of wells driven through the peat into underlying mineral soil aquifers. Water levels in these wells, representing a potentiometric surface, rose well above the mineral soil contact with the peat. This potentiometric surface rose immediately during storm events, apparently in response to pressure from the additional weight of the rainwater in the overlying peat layer. Actual recharge of these mineral soil aquifers did not occur for some time.

Skaggs, et al. (1980) and Daniel (1981) observed that water tables in North Carolina peatlands are normally near the surface (10 to 60 cm deep) in the winter and early spring when rainfall tends to exceed ET. As ET increases during the summer months, water tables decline to lower levels (120 to 150 cm deep), though short-term fluctuations still occur in response to individual storm events (Skaggs, et al., 1980; Daniel, 1981).

Surface Water Runoff

The Albemarle-Pamlico peninsula lacks an extensive system of natural stream channels. Under natural conditions it appears that much overland flow occurs as sheet-like or diffuse flow through the root mat of surface vegetation, eventually entering streams or other water bodies at many points along banks and shorelines (Heath, 1975; Daniel, 1981). These characteristics, together with those previously mentioned, make natural runoff conditions difficult to quantify.

Runoff delivery from peatlands can be quite high when water tables are in the undecomposed surface horizons with high hydraulic conductivities. However, when water tables are in the lower, more decomposed horizons, water yield is greatly reduced (Boelter, 1964; Chapman, 1965; Bay, 1969; Heinselman, 1970; Clark and Clark, 1979; Skaggs, et al., 1980; Daniel, 1981). The effectiveness of peatlands in storing storm waters is thus largely determined by the water table depth at the time of a particular storm. Boelter (1966) concluded that bogs are not the effective long-term storage reservoirs they are often thought to be because undecomposed surface peats release water very quickly and deeper, more decomposed peats release very little water at all. Others agree that peatlands may provide some short-term storage and runoff delay particularly when water tables are low but do not appear to function effectively as long-term storage areas and stream flow regulators (Boelter, 1966; Bay, 1969; Verry and Boelter, 1975; Ayre, 1977; Clark and Clark, 1979; Daniel, 1981). Natural peatlands, like most wetlands, do act as storm buffers in spreading storm waters over large, flat land areas, thereby delaying delivery to receiving streams and estuaries (Verry and Boelter, 1975; Clark and Clark, 1979; Daniel, 1981).

Water Quality in Natural Peatlands

Being isolated from regional groundwater, ombrotrophic peatlands are almost totally dependent upon rainfall for nutrient and mineral inputs (Heinselman, 1970; Moore and Bellamy, 1974; Boelter and Verry, 1977; Daniel, 1981). Natural drainage waters from these areas in North Carolina are typically low in nutrients, conductivity, and pH, and high in humic acids and color (Kuenzler, et al., 1977; Kirby-Smith and Barber, 1979; Daniel, 1981; Richardson, et al., 1981)

Heinselman (1970) stated that ombrotrophic peat waters may be distinguished from others by measuring pH alone. Peats are typically low in calcium, which normally reacts with carbonic acid in rainwater to create bicarbonate, the only strong base found in abundance in natural waters. This, combined with the prevalence of hydrogen cations, produces acid soils and drainage waters with pH's generally less than 4 (Moore and Bellamy, 1974; Daniel, 1981; Richardson, et al., 1981). Sparling (1966) reported that acid conditions are favored by the slow water movement in these systems.

Anaerobic, acid conditions slow the rate of organic matter decay and the subsequent release of organically bound nutrients. Nutrient inputs from precipitation are, for the most part, utilized by living plants or adsorbed by the peat, which has a high cation exchange capacity. Thus, few nutrients are normally available for transport by drainage waters (Moore and Bellamy, 1974; Richardson, et al., 1981). Kirby-Smith and Barber (1979) observed that waters draining from pine forests and pocosins in coastal North Carolina are almost devoid of dissolved inorganic nutrients (nitrogen and phosphorus) when compared to rainwater inputs. Kuenzler, et al. (1977) and Skaggs, et al. (1980) also observed that nitrogen (N) and phosphorus (P) concentrations in streams draining these peatlands are considerably lower than concentrations found in local precipitation. These areas are obviously accumulating P, but Skaggs, et al. (1980) noted that the possibility of N volatilization and fixation makes an accurate N balance impossible. Verry (1982) concluded that Minnesota bogs are nutrient traps, retaining 30-60 percent of all nutrient inputs. However, work by Burke (1980) and Skaggs, et al. (1980) indicates that the capacity of acid peatlands to store P on a long-term basis may be limited (Richardson, et al., 1978).

Mineral concentrations in natural drainage waters are limited to the relatively small amounts delivered by precipitation. There appears to be a net efflux of elements such as calcium, magnesium, sodium, and chlorine from natural systems (Kuenzler, et al., 1977; Skaggs, et al., 1980).

Dissolved oxygen (D.O.) levels in natural streams are sometimes low, particularly during summer low flows when sediment and root metabolism rates are high (Kuenzler, et al., 1977). Biological oxygen demand (B.O.D.) is relatively low in these waters due to the tendency of the soils to have a higher C/N ratio than upland soils, a condition which restricts microbial decomposition (Skaggs, et al., 1980).

Changes in Water Movement with Artificial Drainage

Typical drainage systems installed in the Albemarle-Pamlico region provide mostly surface drainage. The low hydraulic conductivity of these soils limits lateral water movement such that the influence on the peat water table extends little beyond the ditch banks (Skaggs, et al., 1980; Daniel, 1981). Heikurainen (1975) reported that drainage of a forested peatland in Finland lowered the water table by only 200 to 300 mm, an amount which he stated is probably negligible. In Minnesota, Boelter (1972) found that drainage hastened the drainage of the surface root mat but that drainage effects did not extend more than 5 m from the ditch when water tables were in the lower, more decomposed horizons. Evapotranspiration (ET), rather than artificial drainage, is the primary mechanism for water table drawdown in these organic soils (Baden, 1976; Skaggs, et al., 1980).

From studies of North Carolina streams draining predominantly mineral soils, Heath (1975) and Daniel (1981) concluded that the most immediate effects of artificial drainage in wetlands areas were changes in stream flow characteristics. They observed that natural systems have little or no flow for significantly greater periods than drained systems. Because artificial drainage canals are deeper than natural channels, flow can continue at lower water table levels (Heath, 1975; Daniel, 1981). Skaggs, et al. (1980) reported that the deeper collector canals had a greater effect on water table depth than the shallow field ditches and continued to flow after ditch flow had stopped. As discussed previously, the rate of water movement into canals will be greatly reduced as the water table falls from the relatively undecomposed peats at the surface into the deeper, more decomposed peats.

Heath (1975) and Daniel (1981) stated that artificial drainage in the Albemarle-Pamlico region has also increased the hydraulic efficiency of stormwater removal, as evidenced by increased maximum discharges in local streams. This effect has also been reported in northern peatlands (Heikurainen, 1975; Brooks and Predmore, 1978; Clark and Clark, 1979; Mundale, 1981). The capacity of the artificial drainage system is often not sufficient to accommodate these higher storm flows, thus reducing the potential discharge from a drained area (Skaggs, et al., 1980). It appears that increases in base and maximum flows in drained areas are balanced by a decrease in mid-range flow, indicating that artificial drainage alone does not significantly increase total annual runoff (Heath, 1975; Daniel, 1981).

Drainage may also cause subsidence of the peat surface and changes in its hydrologic properties. Subsidence is caused by the cumulative effects of erosion, shrinkage, oxidation, compression, and compaction (Stephens, 1969; Brooks and Predmore, 1978; Skaggs et al., 1980). Peat shrinks with moisture loss and compresses as the buoyant force of water is removed (Stephens, 1969). Oxidation occurs either by the acceleration of aerobic decomposition or by fire losses, fires being more likely in drained areas (Stephens, 1969). Compaction occurs with the use of heavy machinery.

Researchers have found much smaller subsidence rates in North Carolina peats than those reported elsewhere in the literature (Maki, 1974; Skaggs, et al., 1980). Stephens (1969) reported subsidence rates in Florida of from 1.5 cm to 5.8 cm per year, increasing as water table depth increased. Schothorst (1976), from studies of peat drainage in the Netherlands, found subsidence rates of 0.1 to 0.6 cm per year with shallow drainage and 0.5 to 0.6 cm per year with deeper drainage. Skaggs, et al. (1980) reported subsidence rates of only 0.07 cm per year, due most likely to the ability of the more decomposed peats found in North Carolina to hold water against drainage forces.

In some fibrous peats, peat permeability decreases (bulk density increases) rapidly immediately after drying due to shrinkage but later levels off at some minimum value (Eggelsmann, 1975a). This generally has the effect of decreasing ET and increasing runoff (Brooks and Predmore, 1978; Mundale, 1981). In North Carolina peats, drying results in shrinkage but hydraulic conductivity and macroporosity increases upon drying due to irreversible aggregation. Such changes appear to be limited to the thin surface layer disturbed by tillage in agricultural fields and the narrow corridors along canal and ditch banks where vegetation has been destroyed and water table levels are consistently depressed by drainage.

Changes in Water Movement with Land Clearing

Previous discussions have emphasized the importance of ET in regulating groundwater levels and surface runoff, particularly during the summer. Clearing land of vegetation reduces transpiration and increases surface evaporation as wind speeds accelerate and solar radiation strikes the dark soil surface more directly. Because transpiration reductions exceed evaporation increases, ET would be reduced overall (Brooks and Predmore, 1978). Evapotranspiration is further reduced with the elimination of interception losses, the evaporation of rainfall clinging to plants. Boelter and Verry (1977) reported interception losses of 15-20 cm per year in closed canopies of black spruce growing in bogs. This reduction in the ET component will tend to raise groundwater tables and total annual runoff (Bulavko and Drozd, 1975; Brooks and Predmore, 1978; Skaggs, et al., 1980; Daniel, 1981). If clearing includes removal of the surface vegetation by scraping, which removes the surface root mat, runoff will be further facilitated (Brooks and Predmore, 1978; Skaggs et al., 1980)

Changes in Water Movement with Artificial Drainage and Land Clearing

The combination of peatland clearing and drainage has increased maximum flows and total annual runoff in some studies (Mustonen and Suena, 1975; Brooks and Predmore, 1978; Daniel, 1981; Carpenter and Farmer, 1981). This was due mainly to reductions in the ET but was also influenced by improved drainage efficiency of artificial drainage systems, reductions in soil water storage caused by subsidence of the surface layer, or rises in the water table and changes in the hydraulic properties of the peat itself. Annual runoff from a bog in Finland increased by an average of 43 percent in the 9 years following installation of artificial drainage. This was attibuted to the mortality of wetland vegetation caused by a reduction of about 30 cm in average water table depth (Mustonen and Suena, 1975).

Changes in Water Movement with Peat Mining

Peat harvesting exposes more decomposed, less permeable peats which retain water under high suction. This has the effect of increasing surface runoff and maximum discharge while reducing ET, available storage, and groundwater recharge (Brooks and Predmore, 1978). Repeated movement of heavy mining equipment across the surface increases the compaction and bulk density of the peat material, further increasing surface runoff potential. Skaggs, et al. (1980) found that developed sites (agriculture) with deep organic soils had higher bulk densities than undeveloped sites to a depth of 0.4 meters. Mined fields are also shaped for improved surface drainage toward the field ditches.

Peat removal lowers the elevation of the land surface and exposes underlying mineral soils. The hydrologic consequences of this are examined in detail later in this report. In general, it appears that flooding hazards will not be significantly increased as long as the depth of the drainage system is lowered concurrently to maintain adequate drainage gradients and volume capacities. Exposure of the underlying mineral soil and subsequent reclamation activities may actually increase ground water recharge over natural conditions. Due to the relatively small changes in hydraulic head caused by the peat mining in relation to the depth to major aquifers, the rate of saltwater intrusion should not be significantly increased.

Changes in Water Movement with Reclamation

After the removal of surface peat deposits, former peatlands are likely to be used for agriculture or timber production. Newly exposed soils containing significant mineral fractions offer considerably more capacity for crop, livestock, and timber production than deep organic soils.

A study by Skaggs, et al. (1980) indicated that many of the previously described hydrologic patterns associated with drainage, land clearing, and peat mining would continue with agricultural reclamation. The presence of crops would be expected to increase ET over peat mining conditions, but the combination of drainage and a shallow rooting zone would still keep ET below that of natural areas. Skaggs, et al. (1980) reported that this difference in ET would be greater when several short-duration droughts occur during the growing season than when a single drought of longer duration occurs. In the longer droughts, water tables may decline such that ET from both natural and developed sites are reduced. This difference in ET may be great enough in some years to produce somewhat greater total runoff from cultivated areas.

Artificial drainage will continue to be important in any silvicultural operations which may follow peat extraction. The benefits of drainage control in improving growth and maintaining equipment operability are well documented (Klawitter and Young, 1965; Maki, 1974; Campbell and Hughes, 1981). In Coastal Plain forest management, drainage requirements vary with the stage of stand development (Unpublished report of the Forestry Work Group, Governor's Coastal Water Management Task Force, 1982). Lowering of the water table facilitates equipment operability and reduces soil damage during the harvest and regeneration phases. Drainage also prevents flooding during site preparation and regeneration when ET is very low. However, relatively high water tables can be maintained once the new stand has developed in order to maximize growth and minimize the fire hazard. Though ET reduction tends to increase annual runoff and runoff peaks during harvest, site preparation, and regeneration, only about 20 percent of the managed forest areas are in these management phases at any given time. The remainder of the area is occupied by developing or maturing stands with higher ET and thus lower maximum and annual runoff than unmanaged stands. The Governor's Coastal Water Management Task Force (1982) mentions the possibility of using forested areas with drainage controls for the temporary storage and slow release of excess waters from adjacent agricultural areas. Similar hydrologic effects have also been observed in European and Russian forests (Heikurainen, 1975; Vompersky 1975; Pelkonen, 1980; Starr and Paivanen, 1981). Starr and Paivanen (1981) concluded that any hydrologic impact caused by drainage in forest management is reduced in the long-term to the extent that runoff may even be less than that from undrained areas. From studies in the U.S.S.R., Vompersky (1975) reported that forested bog runoff was only one-third to one-half of that from an unforested bog.

Changes in Water Quality with Artificial Drainage

Kuenzler, et al. (1977) concluded that many of the water quality changes associated with drainage are due to the reduction of intimate contact of water with the wetland biota and soils. Drainage reduces rainfall residence time in the peatlands, offering less opportunity for runoff water quality to be changed before discharge. Brooks and Predmore (1978) concluded that most of the water quality impacts of peat mining are due to the influence of artificial drainage.

It appears that drainage increases the potential for transporting more organic matter and nutrients from the peatland (Crisp 1966; Heikurainen, 1975; O'Rear, 1975; Brooks and Predmore, 1978; Carpenter and Farmer, 1981). Heikurainen (1975) stated that P concentrations increase with the increase in humus in runoff waters. Crisp (1966) reported that eroded peat particulates are responsible for most N and P losses but that the availability of these nutrients to plants is probably limited. Increased aeration accelerates peat decomposition and may release organically bound nutrients for transport in drainage waters (Brooks and Predmore, 1978). A study in North Carolina by Skaggs, et al. (1980) indicated that inorganic N levels in receiving streams will not be significantly affected by peatland drainage. This study also suggested that any increases in organic N would be small and that much of this, being in particulate form, would settle out in the ditches where it would probably be removed during normal ditch cleaning operations. It appears from this study, that the drainage of organic soils will not produce significant P loadings unless such drainage is accompanied by the addition of P fertilizer to the soil.

According to Skaggs, et al. (1980) the quantity of sediment produced from draining the deep organic soils in the Albemarle-Pamlico region is not likely to be a problem. It appears that initial drainage canal installation is the primary source of sediment and that once this installation is complete, erosion and turbidity decline to much lower levels.

Increased velocities and turbulence in artificial drainage systems increase dissolved oxygen (D.O.) concentrations in drainage waters (Kuenzler, et al., 1977; Skaggs, et al., 1980). Some increase in pH may occur as a result of drainage, particularly where ditches are deep enough to intersect mineral soil. Bicarbonate in the mineral substrate may neutralize the acidity in these waters (Brooks and Predmore, 1978).

In a survey of peat mining impacts in Minnesota, Mundale (1981) reported evidence that peats tend to absorb such heavy metals as copper, nickel, and mercury, which may be released with peatland disturbance. Detectable concentrations of mercury that exceeded the current North Carolina water quality standard were found in one-time grab samples collected in November 1981 and April 1982 at several locations both within and downstream from the proposed peat mining area on First Colony Farms (Environmental Science and Engineering, Inc., 1982a,b). The State of North Carolina, Division of Environmental Management (NCDNRCD, 1983a) has also measured mercury levels in excess of the standards in both the Albemarle and Pamlico Sounds though the source of these mercury inputs is uncertain at the present time. It should be noted, however, that several other regions in North

Carolina had a higher incidence of elevated mercury levels than the Albemarle-Pamlico region.

Because the conventional surface drainage systems used in this area convert slow and diffuse stormflow discharges into more rapid, concentrated flows, with higher peak volumes the surface water salinity in receiving estuary waters may be decreased. This salinity reduction is detrimental to marine organisms using the estuaries as reproduction and growth habitat. (Kuenzler, et al., 1979; Daniel, 1981; Pate and Jones, 1981; NCDNRCD, 1982).

Changes in Water Quality with Artificial Drainage and Land Clearing

The major additional effect of vegetation removal would be to increase the potentials for sediment and nutrient inputs to drainage waters. Bare soil surfaces obviously have a greater potential for both wind and water erosion. Brooks and Predmore (1978) suggested that additional peat sediment may increase B.O.D. and thereby decrease D.O. levels. Increased speed and quantity of runoff resulting from vegetation removal increases erosion losses and reduces the opportunity for changes in rainfall quality. Since local precipitation contains 3 to 4 times as much N and P as drainage waters from natural areas (Skaggs, et al., 1980) some increase in nutrient efflux might be expected. Shade removal would increase surface temperatures and promote drying, thereby, accelerating nutrient release through organic matter decomposition (Brooks and Predmore, 1978). However, increased rates and quantities of runoff resulting from vegetation removal would tend to further reduce surface water salinities in receiving estuaries.

Changes in Water Quality with Peat Mining

As previously stated, much of the water quality impact of peat mining is due to the artificial drainage. Mining activities do, however, further enhance the potential for sedimentation and nutrient delivery. Mining activities increase the potential for peat erosion by keeping the surface loose and bare of vegetation; by increasing surface runoff; and by increasing wind transport into the ditches (Brooks and Predmore, 1978). Higher concentrations of peat sediments in surface waters may cause increased B.O.D. and reduced D.O. concentrations. Increased disturbance and drying of the peat surface would also encourage further decomposition and release of nutrients and possibly heavy metals (Brooks and Predmore, 1978).

Because mining increases surface runoff, the potential for impacts on estuary salinity would be increased.

Changes in Water Quality with Agricultural Reclamation

Water quality impacts from agricultural activities in the coastal region are primarily associated with the addition of sediment, farm chemicals, and animal wastes to the system

(NCDNRCD, 1979a). A number of investigations in recent years have examined changes in water quality associated with agricultural development in the North Carolina Coastal Plain.

Agricultural development increases the suspended sediment load and turbidity of drainage waters (Gambrell, et al., 1974; Kuenzler, et al., 1977; Kirby-Smith and Barber, 1979; NCDNRCD, 1979a; Skaggs, et al., 1980). The most significant sediment and turbidity increases occur during drainage system construction and land clearing operations.

The fact that agricultural development increases nutrient loadings in drainage waters is well documented by numerous studies across the country. Agricultural development in the North Carolina Coastal Plain has been found to increase inorganic nitrogen (primarily nitrate N) concentrations in local drainage waters (Kuenzler, et al., 1977; Kirby-Smith and Barber 1979; Skaggs, et al; 1980; Daniel, 1981). This form of N is of greatest concern due to its high availability to stream biota. Skaggs, et al. (1980) noted, however, that because inorganic N levels in the natural drainage waters of the region are very low, the actual amount represented by the increase is small compared to better drained soils. However, since coastal streams are naturally very low in nitrate, changes in N availability that are quantitatively quite small may contribute to adverse increases in algal growth. Gambrell, et al. (1974) reported that these poorly drained areas lose much of the nitrogen from the soil as gaseous nitrogen through denitrification. Losses of nutrients and sediment from agricultural fields may be significantly reduced by proper fertilization and water management (Gambrell, et al., 1974; Skaggs, et al., 1980). Organic N increases have also been measured (Kirby-Smith and Barber, 1979; Skaggs, et al., 1980) but much of it is in particulate form which would tend to settle out in drainage canals to be removed during ditch maintenance operations (Skaggs, et al., 1980).

Higher phosphorus concentrations also appear to be a consequence of agricultural development (Kuenzler, et al., 1977; Kirby-Smith and Barber, 1979; Skaggs, et al., 1980; Daniel, 1981; Richardson, 1981). This increase appears to be small except where P is applied to organic soils low in mineral content. Skaggs, et al. (1980) stated that P fertilization of these organic soils may be the greatest potential eutrophication hazard posed by development. Lilly (1981) confirmed the inability of these organic soils to absorb and fix applied P. However, much of the agricultural development in the region occurs on mineral soils where P losses are small (Skaggs, et al., 1980). Soils reclaimed for agriculture after peat mining will likely be predominantly mineral soils.

Agricultural development may also cause increases in pH, D.O., B.O.D., temperature, and concentrations of elements such as Ca, Mg, Cl, Na, and K in drainage waters (Kuenzler, et al., 1977; Skaggs, et al., 1980; Daniel, 1981). The magnitude of these changes does not appear to present a water quality problem at present. Slight increases in some heavy metal concentrations (Cu, Mg, Fe, and Mn) have been measured (Skaggs, et al.). Skaggs, et al. (1980) found fecal and total coliform bacteria counts in drainage waters from pastured areas to be considerably greater than those from natural areas. It was noted in this study that similar increases in coliform bacteria from grazed lands are common throughout the U.S. and appear to pose few water quality problems unless located immediately adjacent to shellfish waters.

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The water quality threat of pesticide use on high organic soils in North Carolina is uncertain. Skaggs, et al. (1980) found that spray applications of alachlor to corn and soybeans over a three-year period resulted in highly variable concentrations of the pesticide in discharge from the field ditches with the highest concentration approaching toxic levels for fish. The few high concentrations were thought to result from spray drift into the ditches. Since alachlor is a pesticide of relatively low toxicity to aquatic organisms, there is likely a definite water quality hazard associated with use of pesticides of higher toxicity. Proper application to preclude direct drift to water surfaces is critical.

Changes in Water Quality with Forest Reclamation

Nitrogen and phosphorus deficiencies are major factors limiting tree growth in pocosins (Woodwell, 1958). Lime and fertilizer applications are important to satisfactory timber production in the region (Maki, 1974; Campbell and Hughes, 1981). However, fertilizer applications are considerably less, both in amount and frequency of application, than in agricultural operations, with a proportionately lower water pollution potential. Lower runoff from forested areas would also lessen the potential for nutrient delivery. A statewide study concluded that silviculture is generally a minor source of water pollution in North Carolina (NCDNRCD, 1979b).

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III. EFFECTS OF PEAT MINING ON RUNOFF AND WATER QUALITY

J. D. Gregory, R. H. Culbreath, and J. R. Bailey

INTRODUCTION

Field studies of the effects of peat mining on runoff and water quality were conducted as a combined operation. The objectives were to:

(1) Determine the effects of vegetation removal and peat mining on the characteristics of runoff from field ditches.

(2) Obtain data on runoff from field ditches and collector canals for model development.

(3) Determine the effects of vegetation removal and peat mining on quality of water flowing from field ditches.

(4) Determine if the water quality measured at the end of the field ditch is altered as the water flows through collector and main canals.

Runoff was measured and water quality samples were taken at the ends of field ditches draining both fields partially mined and fields with natural pocosin vegetation and at several locations downstream in the drainage system.

PROCEDURES

Site Description

Hydrologic field studies were conducted at the pilot peat mine of First Colony Farms, Creswell, North Carolina. First Colony Farms is located on the Lower Coastal Plain on a peninsula between Albemarle and Pamlico Sounds and encompasses parts of Dare, Hyde, Tyrrell, and Washington Counties. The 6,000 ha site that has been permitted for peat mining is located astride the Hyde-Washington County line east of Pungo Lake and south of Phelps Lake (Figures 1 and 2).

The peninsula lies east of the Suffolk Scarp on the Pamlico surface. The permitted peat mine area is located on a broad flat plateau that has the highest elevation on the peninsula. Elevation across most of the mine area varies from around 5.2 to 5.5 m and slowly decreases in all directions outside the perimeter of the mine area. The 4.6 m (15 ft) contour is well outside the mine area around most of the perimeter and the land slopes away

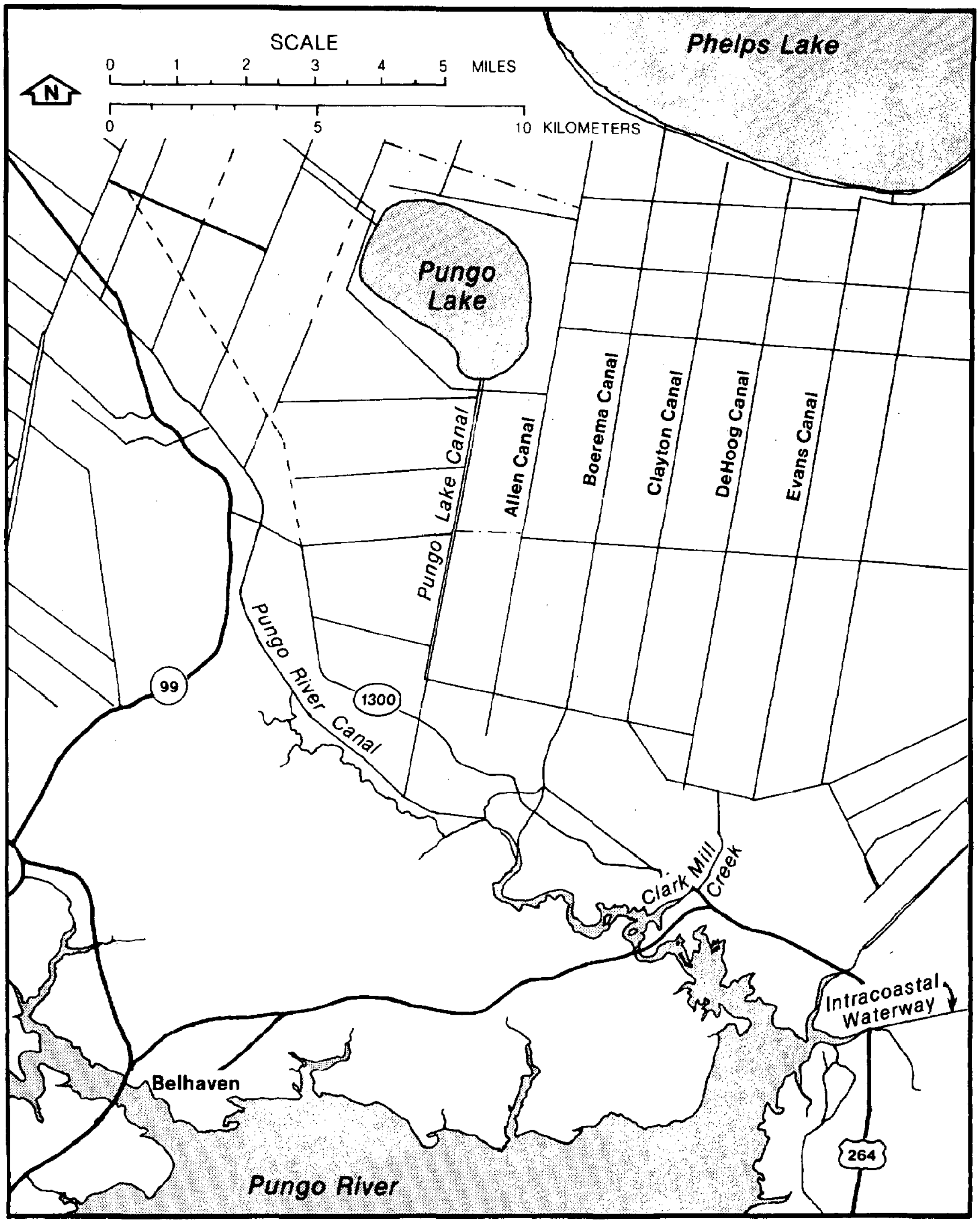


Figure 1. General location of permitted peat mining area.

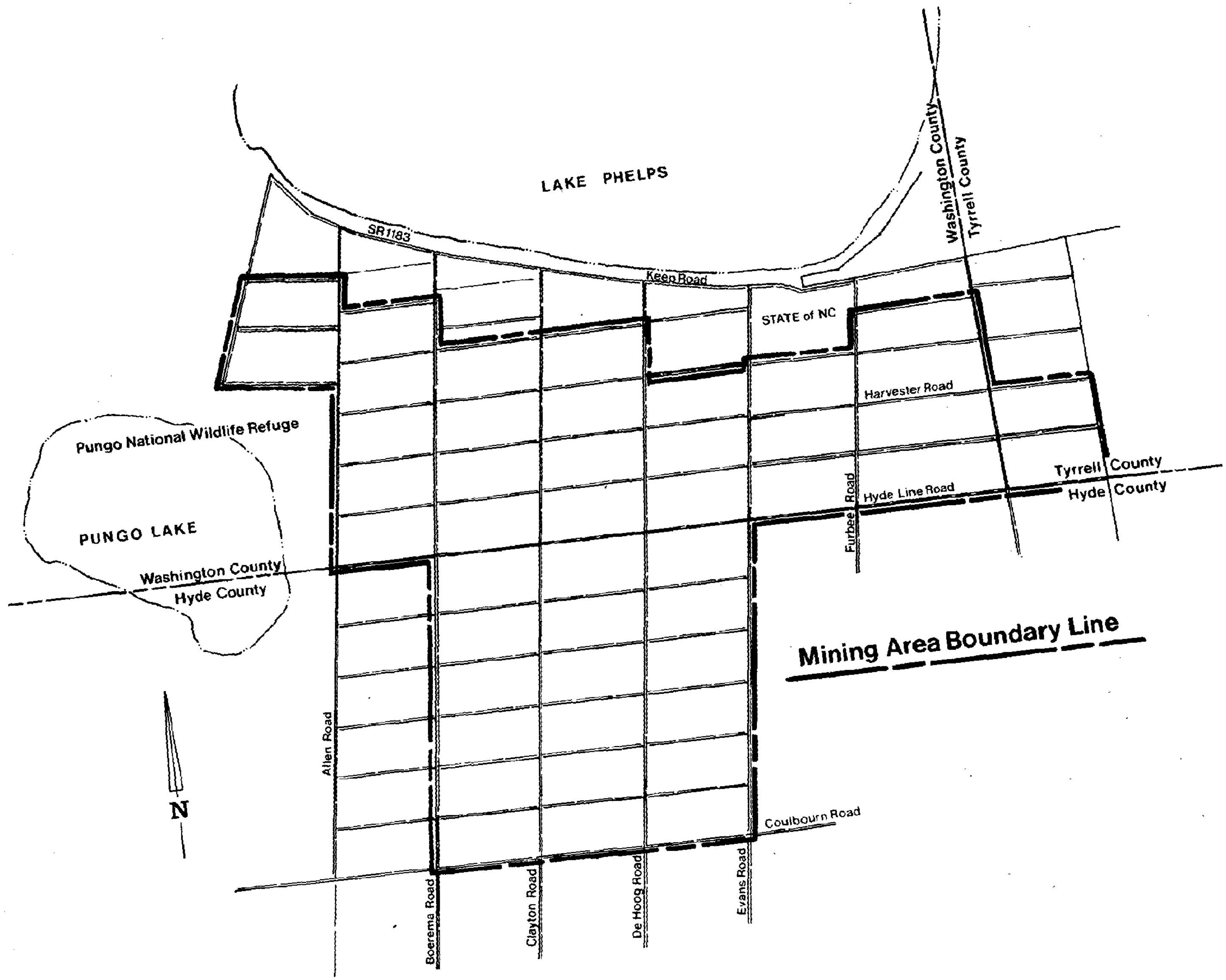


Figure 2. Permitted peat mining area.

on all sides of the area at about 0.02 percent (1 ft/mi). The area is an ombrotrophic bog formed on a broad, flat interstream divide. Before construction of the drainage system, natural drainage flowed outward from the plateau in all directions to the Albemarle Sound, Alligator River and Pungo River. All drainage water from the mine area now flows southward to the Pungo River. Average water surface elevation of Pungo Lake and Phelps Lake is about 3.0 m.

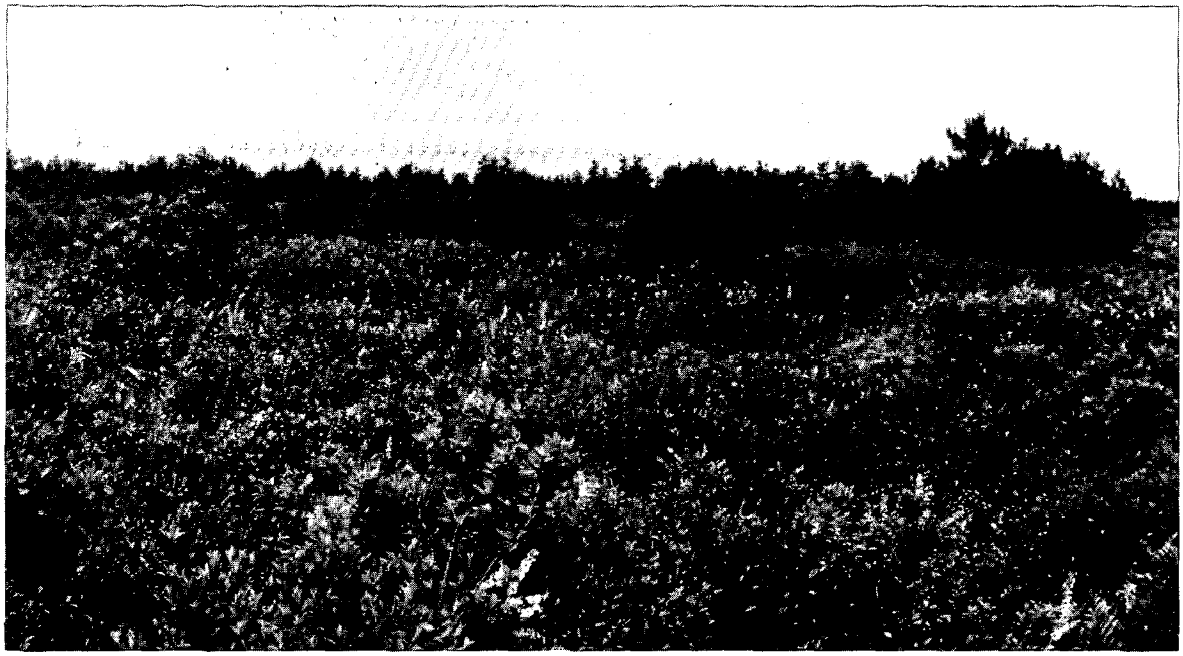
The peat mine area is a low pocosin (Otte, 1982) and in the U.S. Fish and Wildlife Service classification is a Palustrine Wetland, scrub-shrub class (Cowardin, et al., 1979). Past efforts to develop the area for agriculture have resulted in a mixure of vegetation types. The most recently disturbed areas have early successional grass and forb vegetation that is dominated by broom sedge (Andropogon spp.). Areas that have been undisturbed for longer periods have dense stands of typical pocosin shrub vegetation with a few pond pines (Pinus serotina) widely scattered among the shrubs (Otte, 1982; Sharitz and Gibbons, 1982). Vegetation on the study area is of the latter type (Figure 3A) and the soils and vegetation are quite similar to other low pocosins that have never been disturbed by man's activities. Some of the land in the study area was probably cleared in the late 1960's but was quickly abandoned when it became clear that cultivating the deep peat was impracticable.

Soils in the permit area are very poorly drained Histosols and are predominantly Pungo muck (Typic Medisaprist, dysic, thermic). The sapric surface layer of the Pungo series ranges from 1.3 m to more than 2.5 m in depth. The organic material is extremely acid, has up to 35% by volume of stumps and logs (Figure 3B) and commonly has charcoal in the surface meter. The wood is mainly Atlantic white cedar (Chamaecyparis thyoides) and cypress (Taxodium distichum). The underlying mineral horizon is sandy clay, silty clay, or clay and is strongly gleyed.

The depth of peat in the permit area corresponds to the depth of the sapric organic surface layer. The surface of the mineral soil is undulating and forms broad shallow basins with little evidence of the peat-filled relic stream channels that are found in other pocosins in the area. Heating value of the peat (zero moisture) averages about 24 million J/kg (10,300 btu/lb) and ash content averages 3 percent (Ingram and Otte, 1982).

Average annual rainfall in the area is 130 cm and average annual temperature is 15.7°C.

The drainage system in the permit area had been installed for agriculture some years prior to development of the mining project. Some of the field and collector ditches were probably installed when a major portion of the area was cleared in the late 1960's. The major canals and some collector ditches were installed at earlier times; some date to the mid-1800's. No exact records exist on dates that specific areas were drained and/or cleared. Main canals that are about 2-3 m deep are spaced



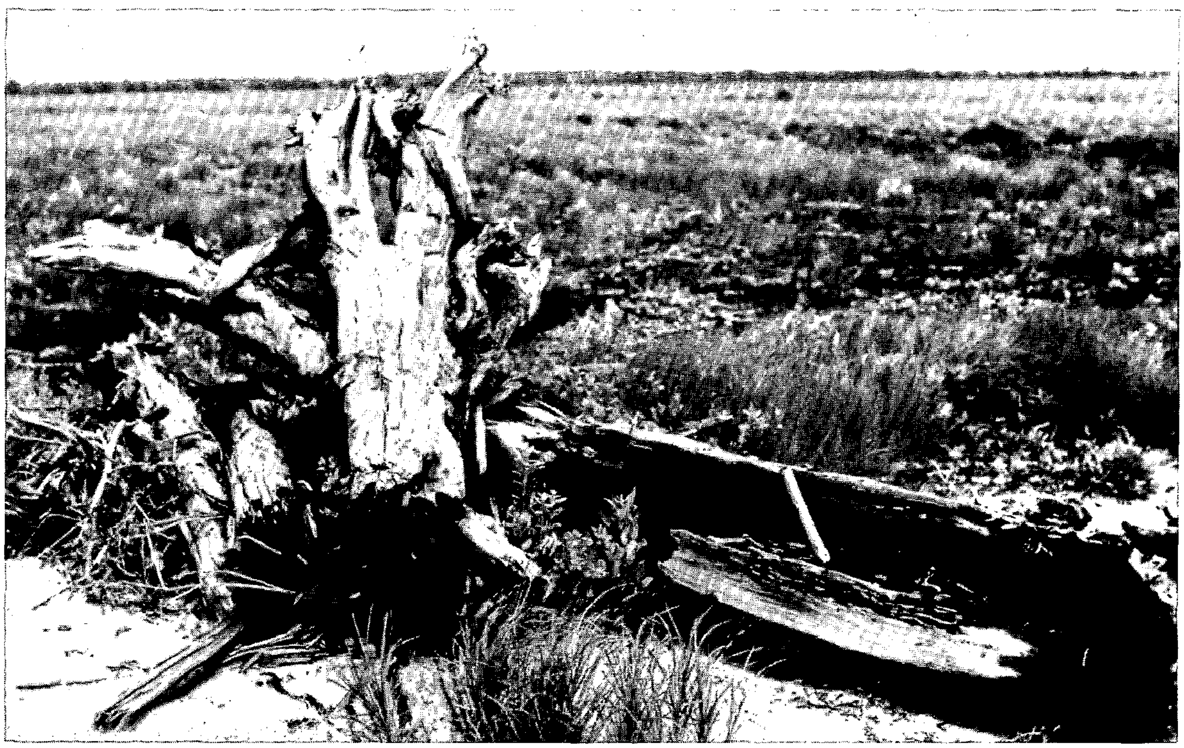


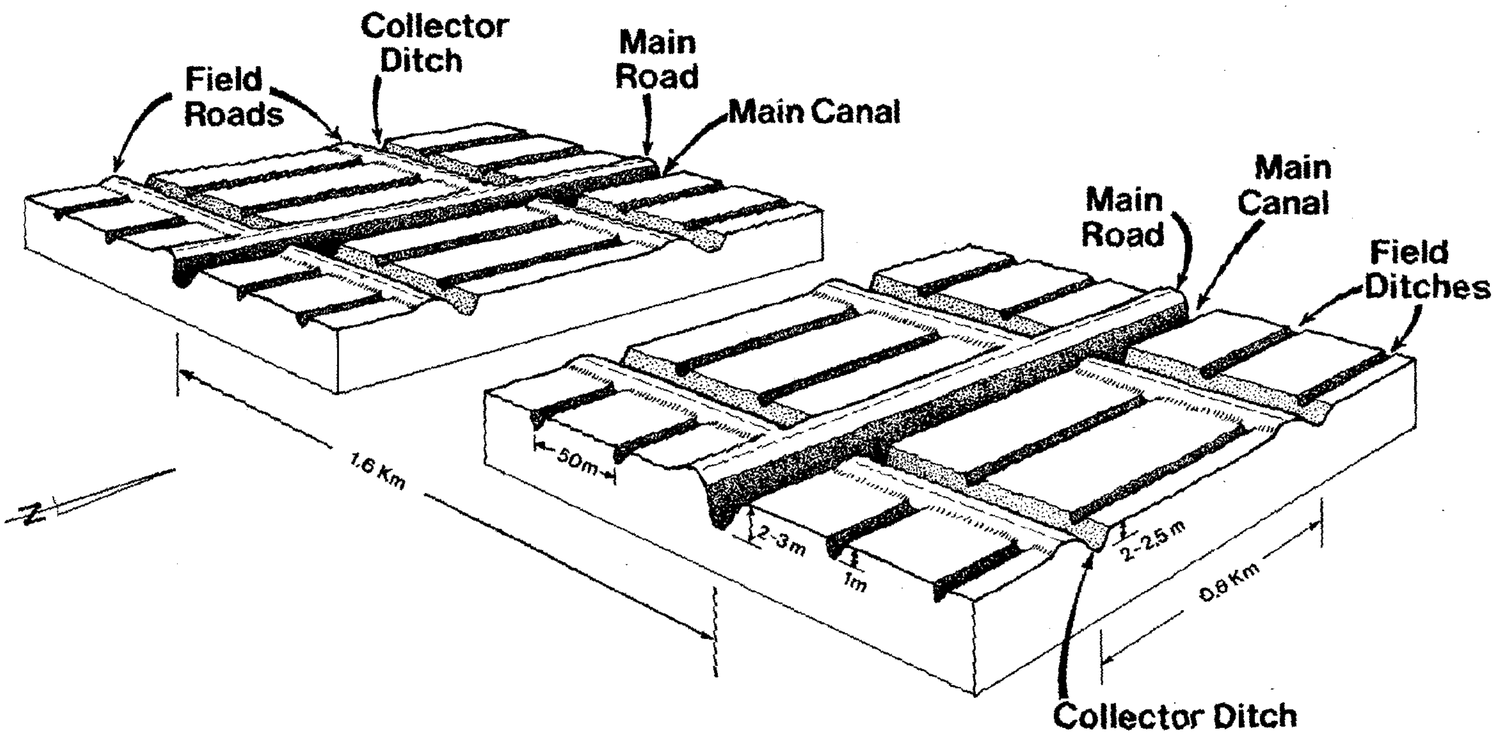
Figure 3. (A) Natural vegetation on the mine site.
(B) White cedar logs removed from the peat during canal construction.

1.6 km (1 mi) apart and are oriented north-south. Collector ditches that are about 2 m deep and are spaced at 0.8 km (0.5 mi) intervals discharge to the main canals. Field ditches originally had been installed in some parts of the area at 100 m intervals. To increase the efficiency of drainage for peat mining, additional field ditches are installed to reduce the spacing to 50 M (Figures 4 and 5).

Installation of the drainage system alters the hydrology of the pocosin but preparation of a mining strip (4 ha area defined by field and collector ditches) for peat removal further alters its surface hydrologic condition (Figure 6). The vegetation is pushed over, ground, and incorporated into the surface 10 cm by use of a modified Bros Rotor Mixer. A second pass with the grinding machine further grinds the vegetative debris, mixes it with the upper layer, and grinds any wood down to a depth of about 25 cm. The mining strip is then graded to produce a smooth surface with an outslope of about 1 percent from the center. For our purposes, a drainage field is defined as the area (4 ha) bounded by mining strip centers and collector ditches, with a field ditch dividing the field in halves. Preparation of the surface for mining involves milling and grinding with a darf miller designed to grind heavy wood to about 10 cm of depth. The top 3-4 cm of the organic material is then milled and fluffed with a shallow milling unit and is turned several times with a spoon harrow during the drying cycle (PMA, 1983). Infiltration rates are very high in the dry, shallow surface layer, but once saturated, surface runoff rates are high and the organic material at the surface is highly susceptible to erosion. The loose dry material at the surface is also highly susceptible to wind erosion.

Milling, drying, and removing a 3-4 cm layer of peat will require 1-2 weeks depending on drying conditions. The production mining season typically will extend from mid-March to mid-November and should have an average of about 142 drying days when equipment can operate in the fields. It is estimated that a maximum of about 41 cm of peat may be depleted annually, yielding about 990 metric tons/ha of 40% moisture peat (PMA, 1983).

Surface conditions varied among the mining strips during the study because of differences in duration of mining. Mining was conducted on the strips draining through Site 201 during August-November 1981, and on the strips draining through Site 202 during August-September 1981 (Figures 7 and 8). The surface remained bare but undisturbed on the other sites with the exception of some brief experimental activity with the sod peat machine. No mining was done in 1982, but the surface of the strips was milled a couple of times to prevent growth of natural vegetation.



Typical Artificial Drainage System for Mining
Sketch not to scale

Figure 4. Drainage system characteristics.

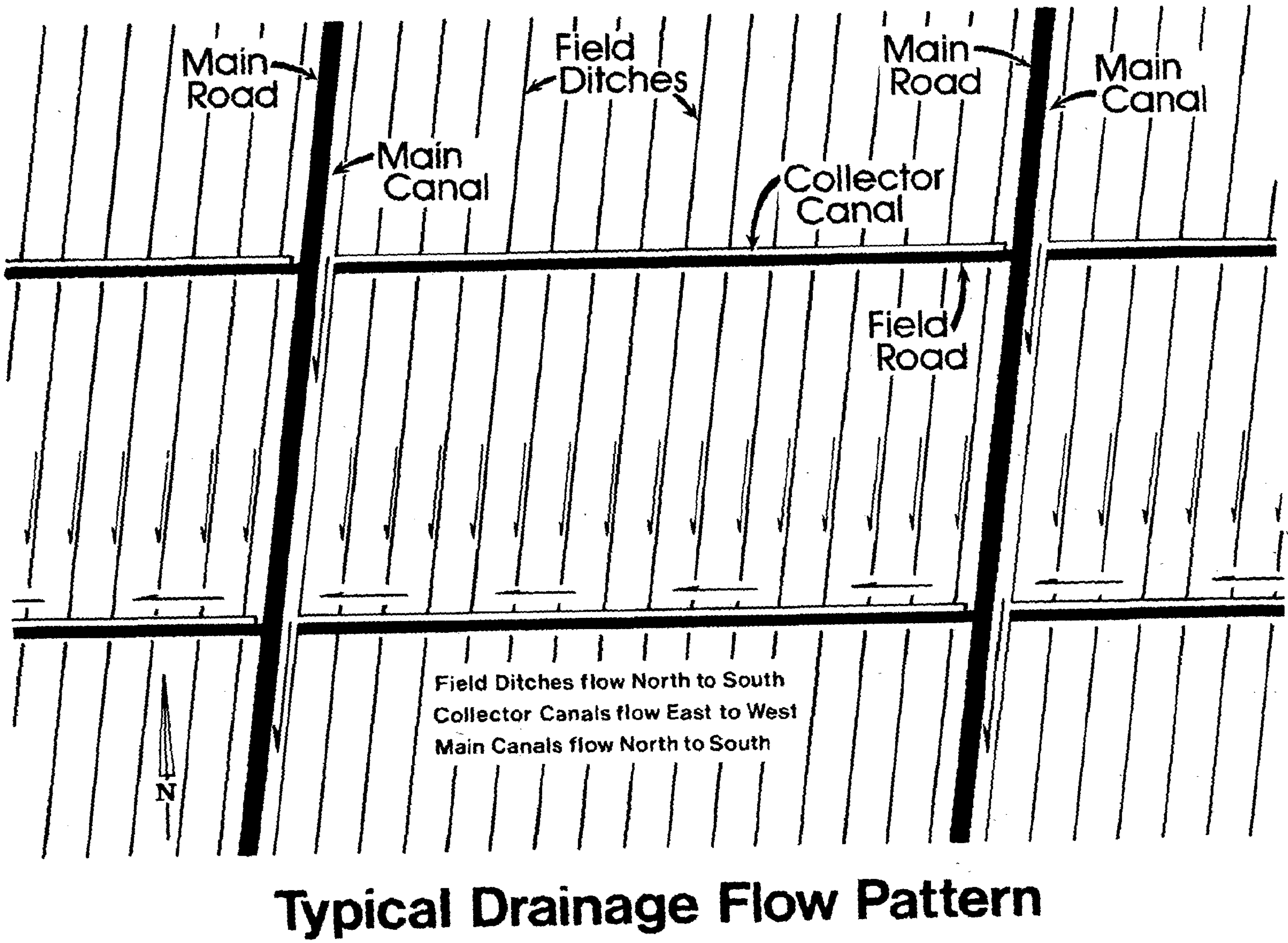


Figure 5. Drainage flow pattern in the peat mine area.

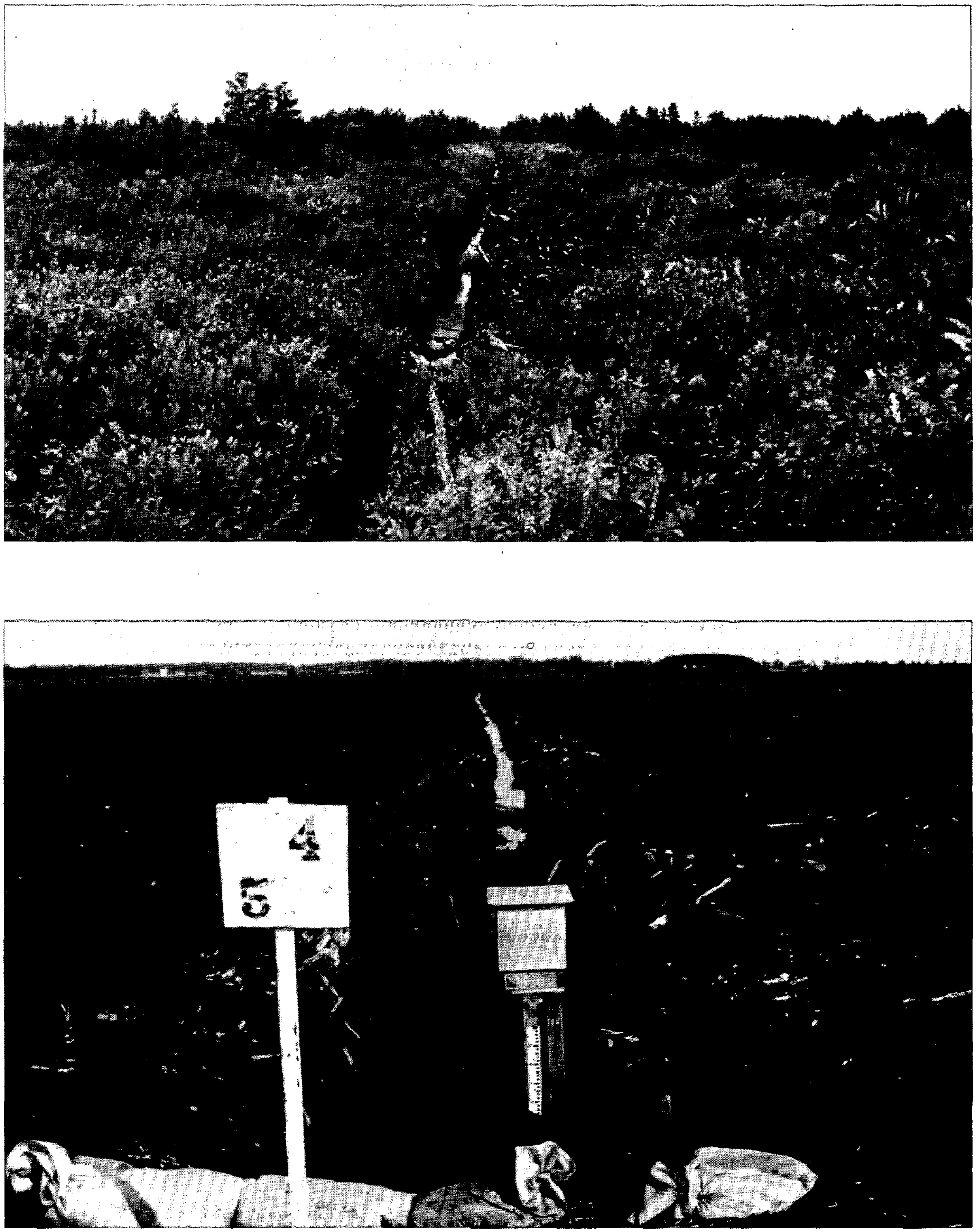


Figure 6. (A) Field ditch on site with natural vegetation.
(B) Field ditch on site prepared for peat harvesting

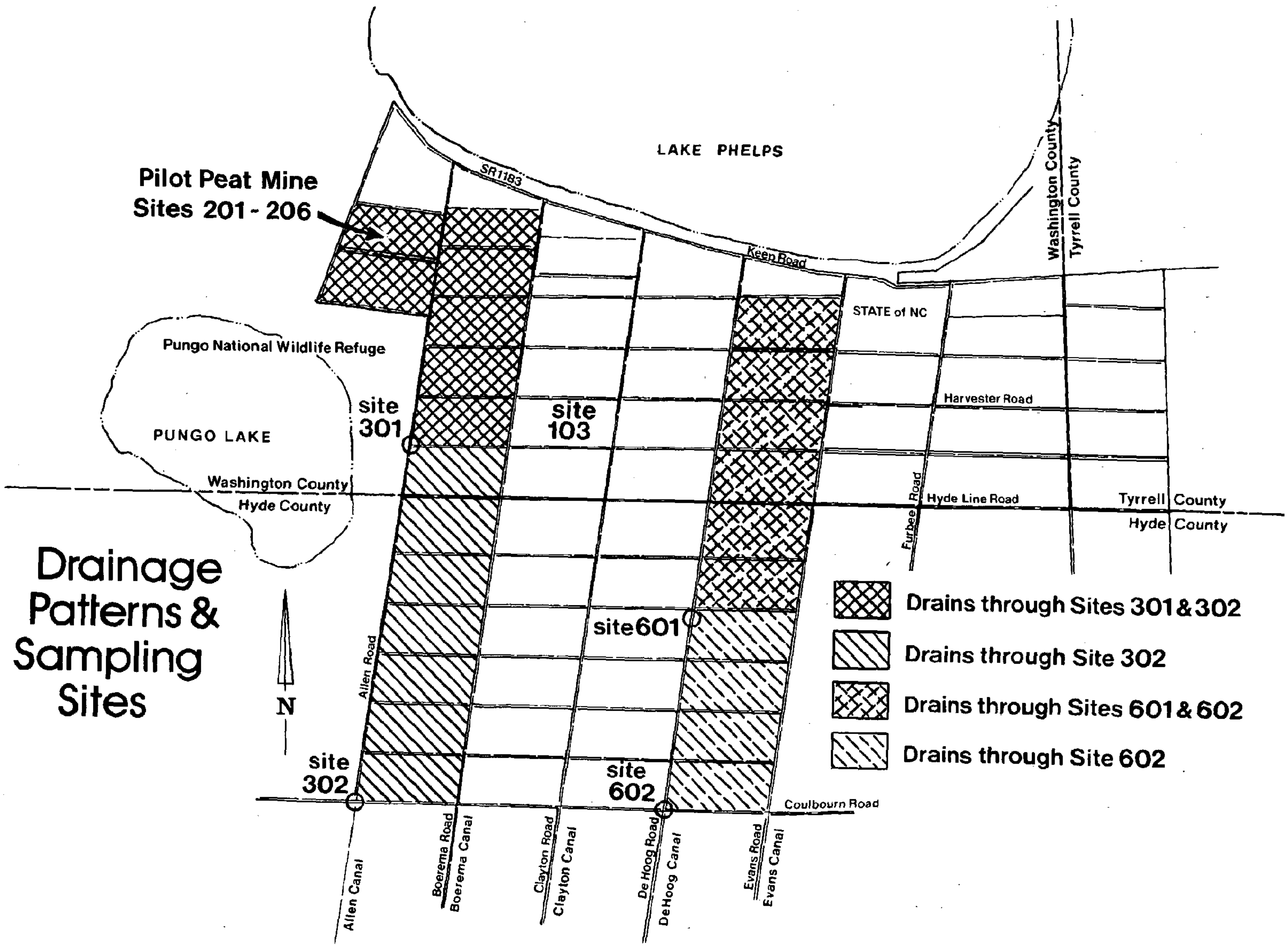
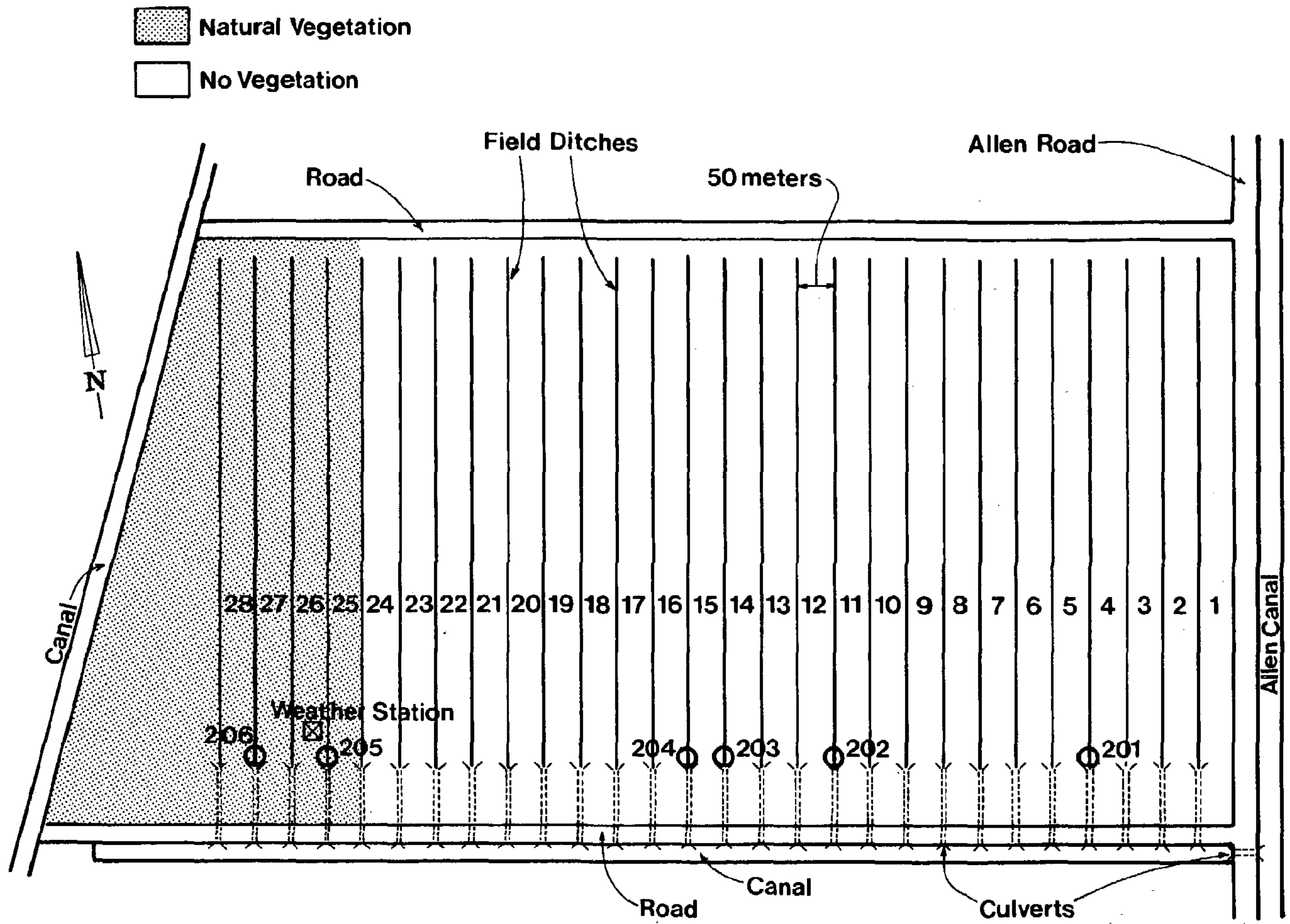


Figure 7. Location of pilot mining area and sampling sites.



PILOT PEAT MINE SITES 201-206

Figure 8. Location of sampling sites for runoff and water quality in field ditches on the pilot mining area.

Sampling Site Selection

First Colony Farms (FCF) initiated preparation for mining on the 130 ha block in the northwest corner of the permitted area in October, 1980. A major portion of the block (75 percent) was prepared for mining by incorporating the natural vegetation into the peat and shaping the surface of the mining strips to achieve a one percent slope from the center of the strip to the field ditch. The remainder of the block was left in natural vegetation. Six sites for measuring runoff and water quality in the field ditches were randomly selected representing two replications each of three treatments: (1) Standard mining practice, including a silt screen installed in the field ditch (Sites 201 and 202); (2) standard mining practice but without a silt screen (Sites 203 and 204); and (3) drained, naturally vegetated, without a silt screen (Sites 205 and 206) (Figures 7 and 8).

Initial preparation of the peat mining site with the Bros Rotor mixer was accomplished during the period October, 1980 to June, 1981. Cleaning of field ditches already present and construction of new ditches was accomplished in the area encompassing sites 201-206 during the same period. Mining was conducted on the strips draining through Site 201 during April-November 1981 and on the strips draining through Site 202 during August-September 1981. Sites 203 and 204 remained undisturbed after completion of the sloping operation in late spring 1981. No mining was accomplished on these strips during 1982. Mining strips draining through Sites 201 - 204 were milled a couple of times during 1982 to prevent regrowth of vegetation and destroy the surface crust, but the surface was much more stable than it would have been under conditions of active mining.

To determine the quality of water in the drainage system down stream from the mine area, two sampling sites were selected on Allen Canal (Figure 7). Sites 301 and 302 received drainage water from approximately 800 ha and 1700 ha respectively, including the pilot mining block.

Sites 601 and 602 on DeHoog Canal were selected as a control. These sites received drainage water from approximately 800 ha and 1300 ha (respectively) of undeveloped land with soils and vegetation similar to that at the pilot mine site.

On April 29, 1981, an intense wildfire moved onto FCF property from the Pungo Wildfire Refuge (Figure 9) and, subsequently, all field data collection was halted. In the effort to extinguish the fire, all main canals from the permitted area were blocked and the entire area was flooded with water pumped from Lake Phelps. The fire was not completely controlled until early June, 1981. Drainage canals were then reopened and dredged where necessary during the following weeks. The original flow pattern of the drainage system was restored by early August, 1981. The pilot peat mining area and the control area along DeHoog Canal were not burned and were only indirectly and

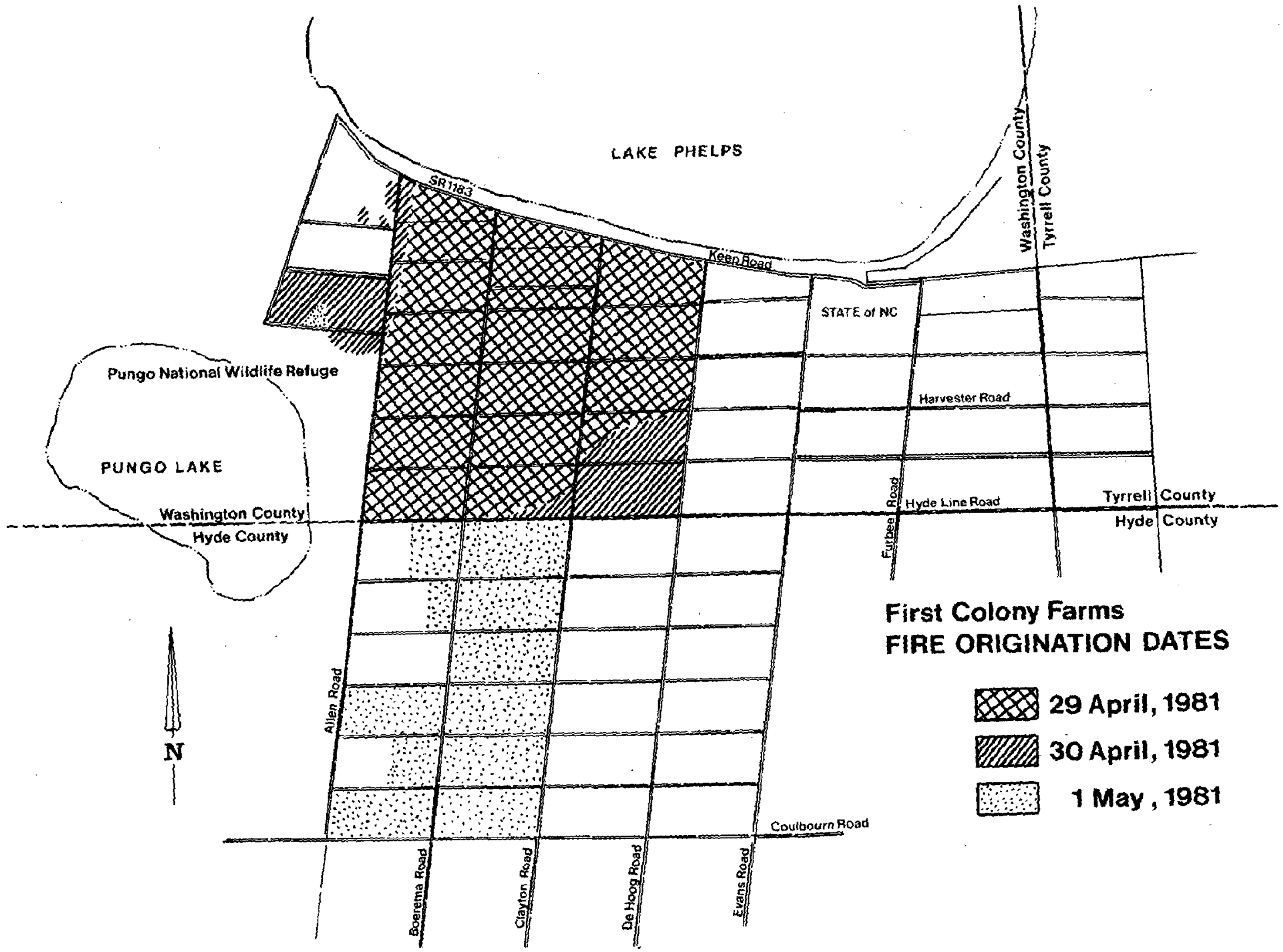


Figure 9. Portion of permitted mine area burned by wildfire in 1981.

temporarily affected by the fire due to drainage system blockage. Deposition of ash from the fire occurred throughout the study area and may have had a significant influence on water quality. Water quality and quantity were significantly affected at Sites 301 and 302, as discussed later in this paper. A total of 6,000 ha was burned by the fire, approximately half of which was within the permitted mining area. The vegetation was destroyed and the peat was consumed to an average depth of about 12 cm, leaving large quantities of ash on the surface (Figure 10).

Hydrologic Measurements

Discharge from field ditches was continually measured by recording the stage on the upstream side of 60° V-notch weirs. The weirs were mounted in flashboard risers attached to the ends of 38 cm diameter, 37 m long culverts that connected the downstream ends of the field ditches to the collector canal (Figure 11). Bottoms of the notches were set at about the same elevation as the ditch bottoms (approximately 0.6 m above bottom of stilling pond) so that the ditches would drain completely after each rainfall event. Trash screens were mounted in front of the weirs and, on two of the mined fields, silt screens were installed at the entrance to the stilling pond about 10 m upstream from the weir (Figure 12B). Stevens Type F water level recorders were utilized to record stage (Figure 12A). Recorders were serviced weekly on the same day that water quality sampling was conducted.

Accurate calibration of the weirs proved to be impossible because the volume and shape of the stilling ponds varied over time. Therefore, the standard equation for a 60° V-notch weir was used to convert stage to discharge. Large quantities of organic sediment were moved to the ditches by overland flow and were deposited in the stilling ponds by each storm. Sediment was also contributed by wind erosion (especially during a short period in early spring, 1982) and by sloughing of ditch and stilling pond banks. Large quantities of wood collected against silt screens, weir trash screens and in the stilling ponds. The ponds had to be cleaned out with a backhoe on a regular basis (Figure 13).

Accumulations of sediment and woody debris on weir trash screens during periods of high discharge sometimes blocked water flow. Submergence of the weirs due to inadequate downstream discharge capacity during periods of very high rainfall also invalidated some flow measurements. Because of these problems, the flow data presented in the results section were limited to events in which free flow over the weirs occurred during the entire period of the hydrograph.

Constant, accurate flow measurements in the main canals were impossible without some type of control structure installed. Water velocities were often below measureable speeds during low flow conditions. During high flows, unstable channel configuration and backwash when drainage capacities were exceeded

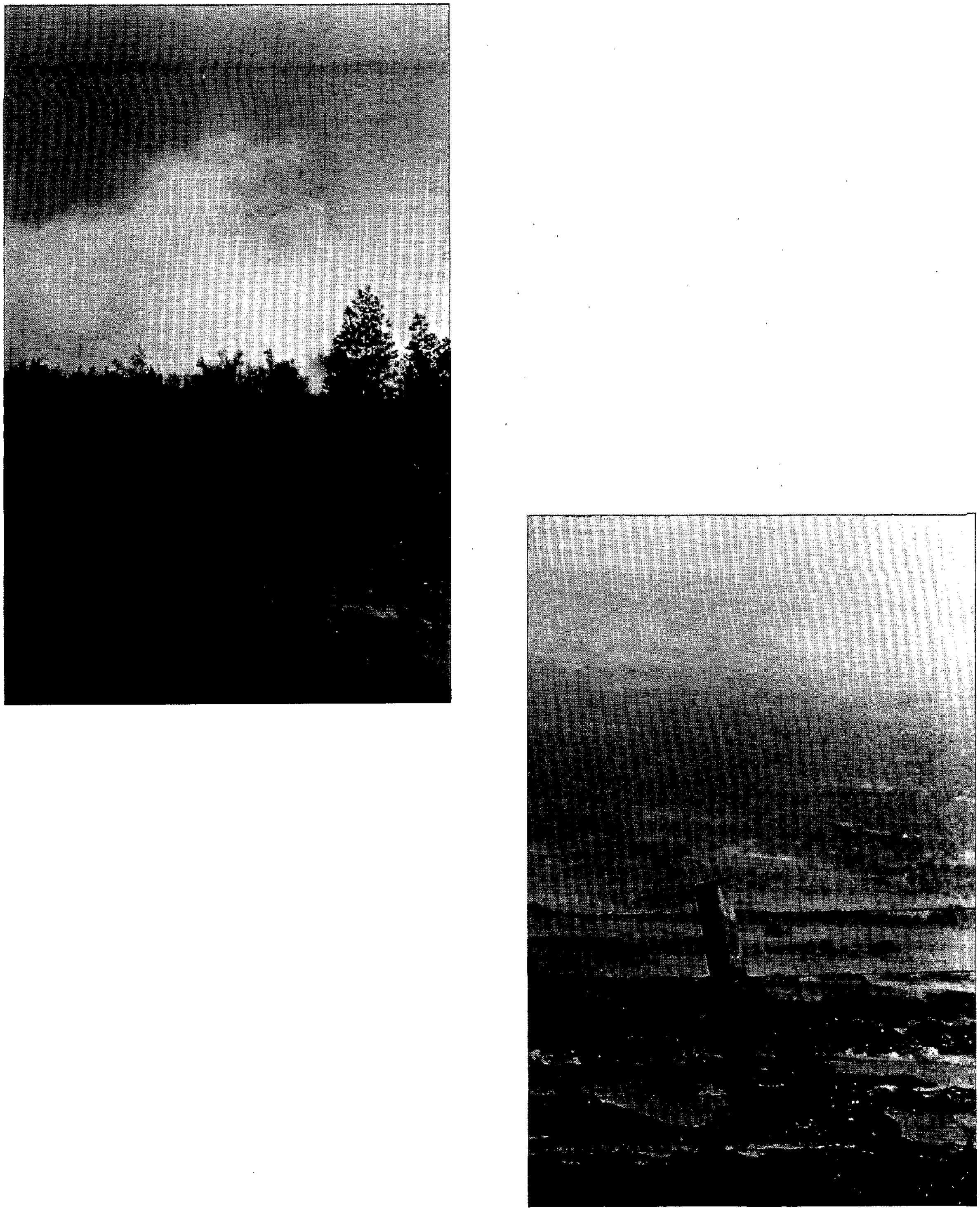


Figure 10. Aftermath of 1981 wildfire.

Figure 11. (A) Installation of weir in flashboard riser.
(B) Completed weir installation with trash screen.

Figure 12. (A) Complete runoff measurement installation.
(B) Silt screen installed in field ditch.

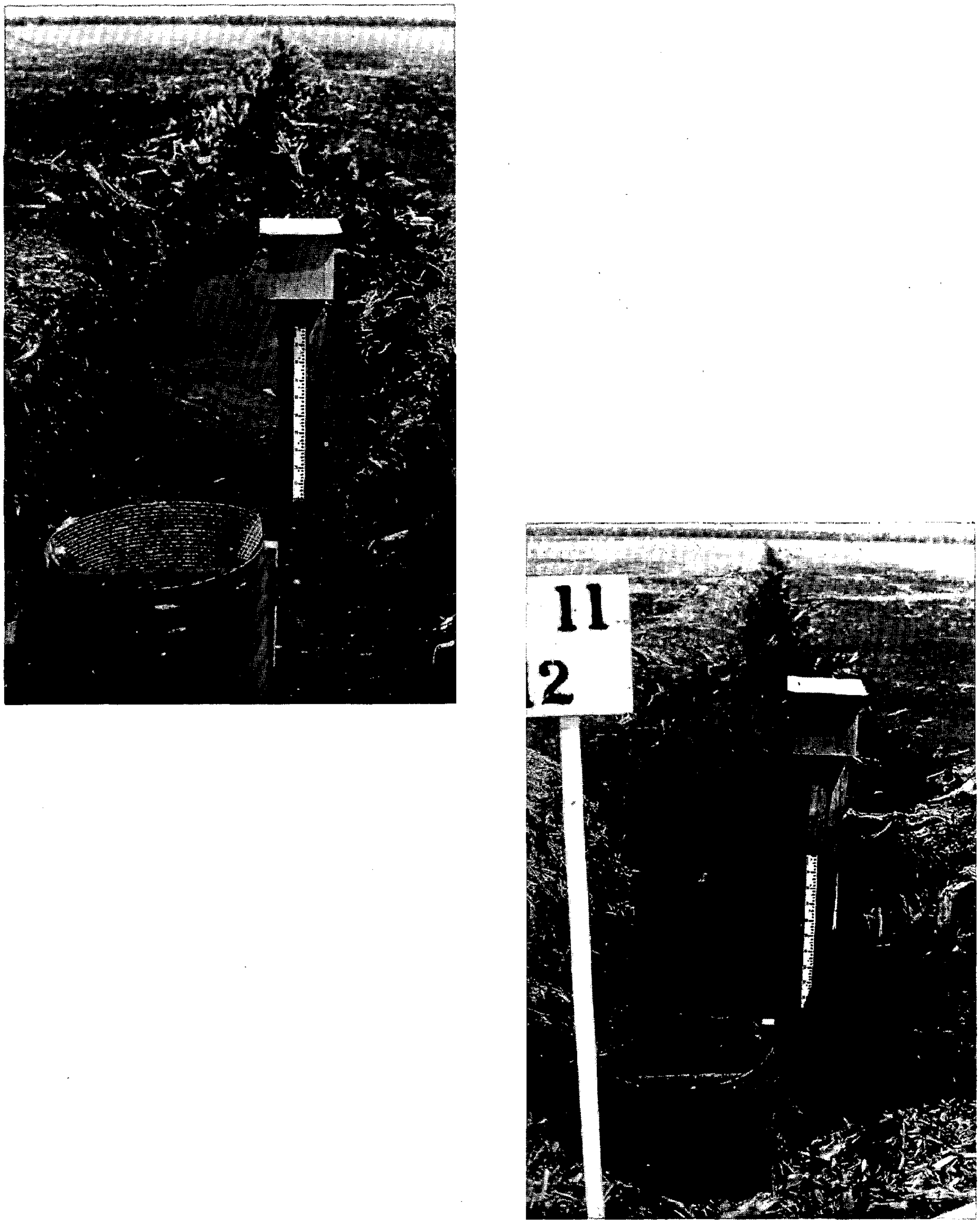


Figure 13. Organic sediment and wood accumulated in field ditch and stilling pond, (A) without silt screen (B) with silt screen.

prevented the establishment of a consistent stage-discharge relationship. The installation of flashboard risers was considered feasible only on DeHoog Canal at Coulbourn Road (Site 602) (Figure 14). A pair of 90° V-notch weirs were installed in the risers in April, 1982. Irregular pumping activity related to field testing upstream at the site of the methanol gasification plant and a ruptured earthen plug invalidated much of the flow data recorded at Site 602. Calibration tests indicated that standard equations for the combination of V-notch and retangular weirs were suitable for converting stage to discharge. One weir was mounted 5 cm lower than the other to ensure accurate measurement of low flows.

Recording tipping-bucket raingages were installed at Sites 205 and 602. A meteorograph was installed in a weather shelter at Site 205 to record temperature, relative humidity, and barometric pressure. Weather data collected at Sites 205 and 602 were supplemented with data from FCF and from the National Oceanic and Atmospheric Administration.

Water Quality Measurements

Weekly grab samples for laboratory analysis were collected in 500 ml polyethylene bottles, immediately placed on ice, then refrigerated at 4°C until analyzed. No preservative was used as analyses were initiated within 24 hours. At all flow-measurement stations, samples were taken from the nappe flowing over the weir. Depth-integrated samples were taken from the open channel sampling sites using a USGS model U.S. DH-75P sampler.

Water temperature and conductivity were measured in the field with a Fisher Model 152 conductivity meter. Dissolved oxygen was measured in the field with a Model 51B dissolved oxygen meter manufactured by Yellow Springs Instrument Company. Skaggs, et al. (1980) found no differences between in situ pH measurement and laboratory pH measurement on water from this area. Therefore, all pH measurements were made in the lab, with a Fisher Model 701A pH/mV meter.

Total nonfiltrable residue, or total suspended matter, and concentrations of dissolved constituents were also determined in the lab. Total nonfiltrable residue analysis was done using Whatman 934-AH glass fiber filters (.45 m pore) (American Public Health Association, et al., 1981, pg. 94). Turbidity, in nephelometric turbidity units (NTU) was determined using a model DRT 100 H/F Instrumentation Turbidimeter (American Public Health Association, et al., 1981, pg. 132).

Chloride ion concentrations were determined with a Model 4-2008 Buchler-Cotlove chloridometer.

Analyses for ammonia and nitrate were initially conducted using specific ion electrodes. Beginning on August 17, 1982, all ammonia and nitrate analyses were done with automated colori-

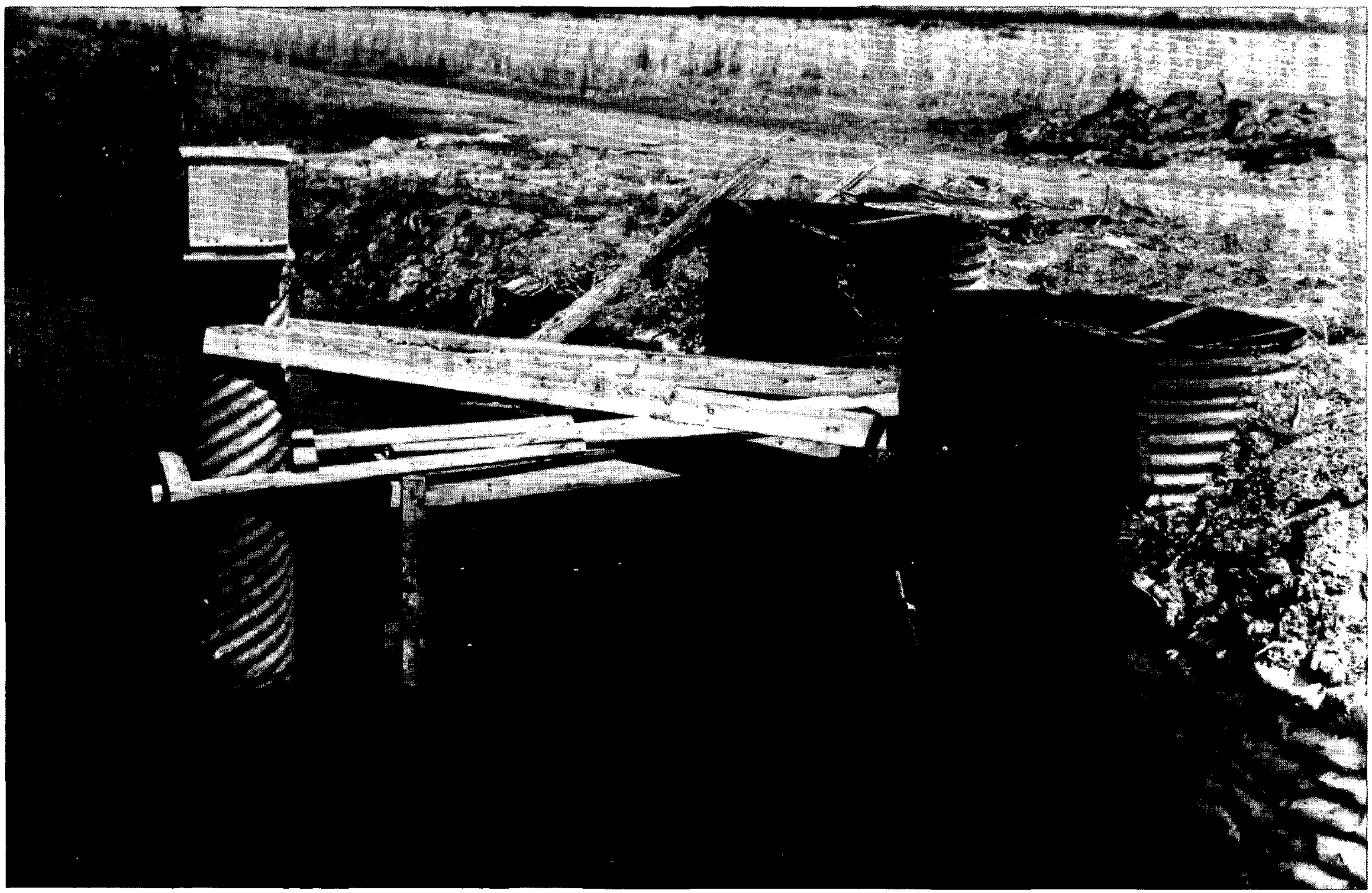


Figure 14. Weir installation on DeHoog Canal.

meteric methods (Technicon Autoanalyzer II) at the laboratory of the U.S. Forest Service Southeastern Experiment Station in Research Triangle Park, N. C.

Total Kjeldahl nitrogen and total phosphorus were simultaneously determined by an automated colorimetric method (Technicon Industrial Method No. 329-74 W/B, revised 11/78) after digestion (Technician Industrial Method No. 376-74 W/B, revised 11/78). Orthophosphate (in the filtrate) was determined as reactive phosphate by an automated colorimetric procedure, beginning August 17, 1982 (Murphy and Riley, 1962; Lennox, 1979; modified in 1982 by Dr. Carol G. Wells, Soil Scientist, U.S. Forest Service).

Samples were analyzed for total calcium, magnesium, and potassium on an atomic absortion spectrophometer (U.S. Forest Service Laboratory) after being ashed at 455°C and brought back to the original volume with a Ø.3N HNO_3 and Ø.2% Lanthanum solution. Cation analysis was conducted on a limited number of samples during the final months of the project.

All concentrations are expressed on an elemental basis.

RESULTS AND DISCUSSION

Runoff

The relative volume, duration, and peak flow rate of discharge from the field ditches were greatly influenced by the differences in conditions brought about by site preparation and/or peat mining. For all storm events, volume, duration, and peak flow were greater from the mining sites (2Ø1-2Ø4) than from those having natural vegetation (2Ø5-2Ø6) (Figures 15-18). Peak flows were generally 5-1Ø times higher from the mining sites than from the vegetated sites. The events shown for illustration are relatively small ones because flooding of the weirs prevented accurate measurement of the higher flows of larger storm events. However, the general relationships among the various hydrographs were similar for storms of all sizes. The models developed by Broadhead and Skaggs (Chapter IV) were used to analyze long term cumulative outflow volumes as well as individual storms of varying magnitudes.

Removal of the vegetation and maintaining the surface in a fallow condition greatly reduced ET. That factor alone probably accounted for a significant part of the increase in storm runoff and peak flow that resulted when the site was prepared for peat mining. Transpiration plus upward flux removed water from the soil more rapidly and to a much greater depth in the vegetated fields than upward flux and surface evaporation alone could remove water from the fallow fields. Thus, less soil water storage was available in the cleared mining fields than in the vegetated fields. A greater proportion of the water from each

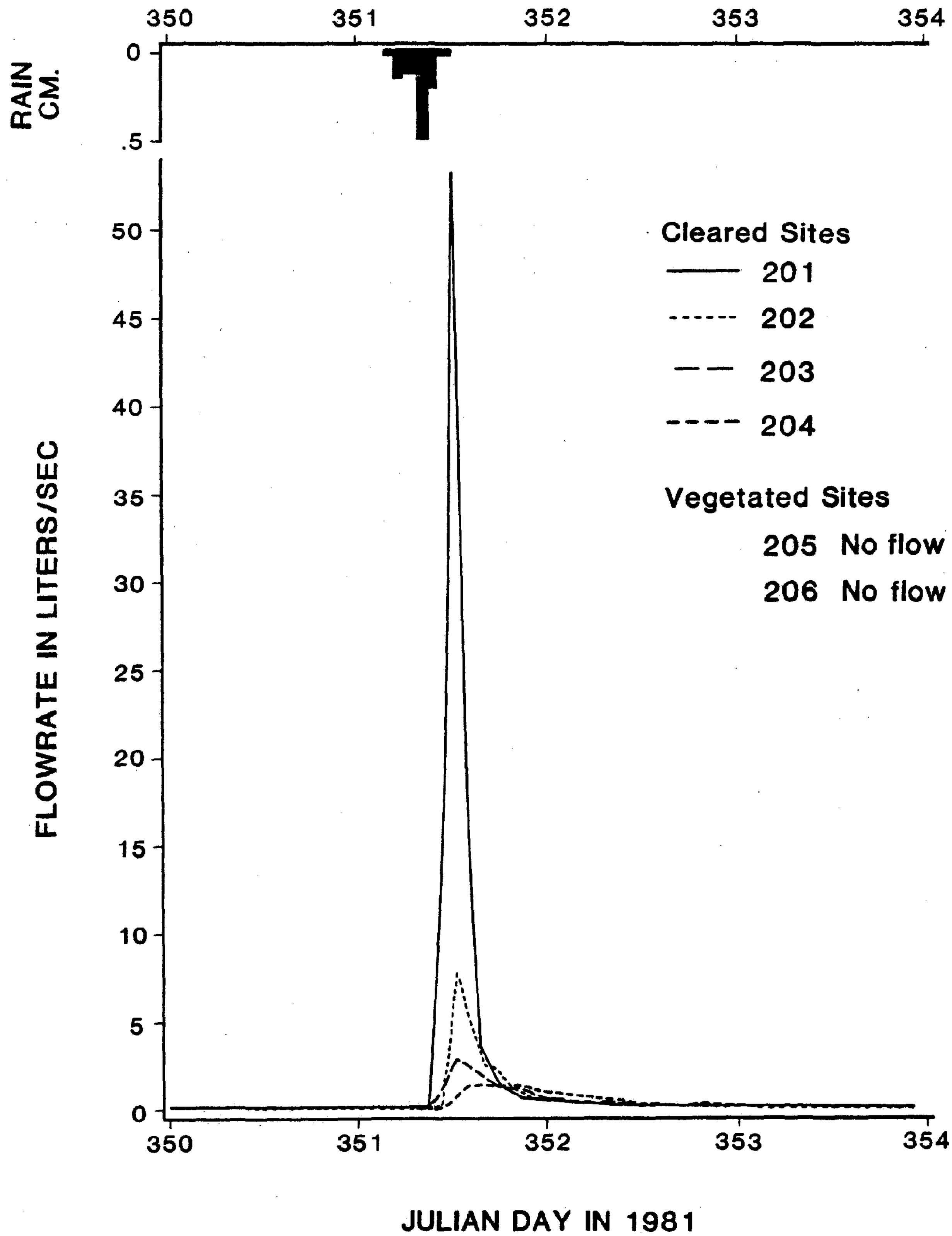


Figure 15. Runoff from field ditches draining cleared and vegetated sites (4 ha in size) for a storm in December, 1981/

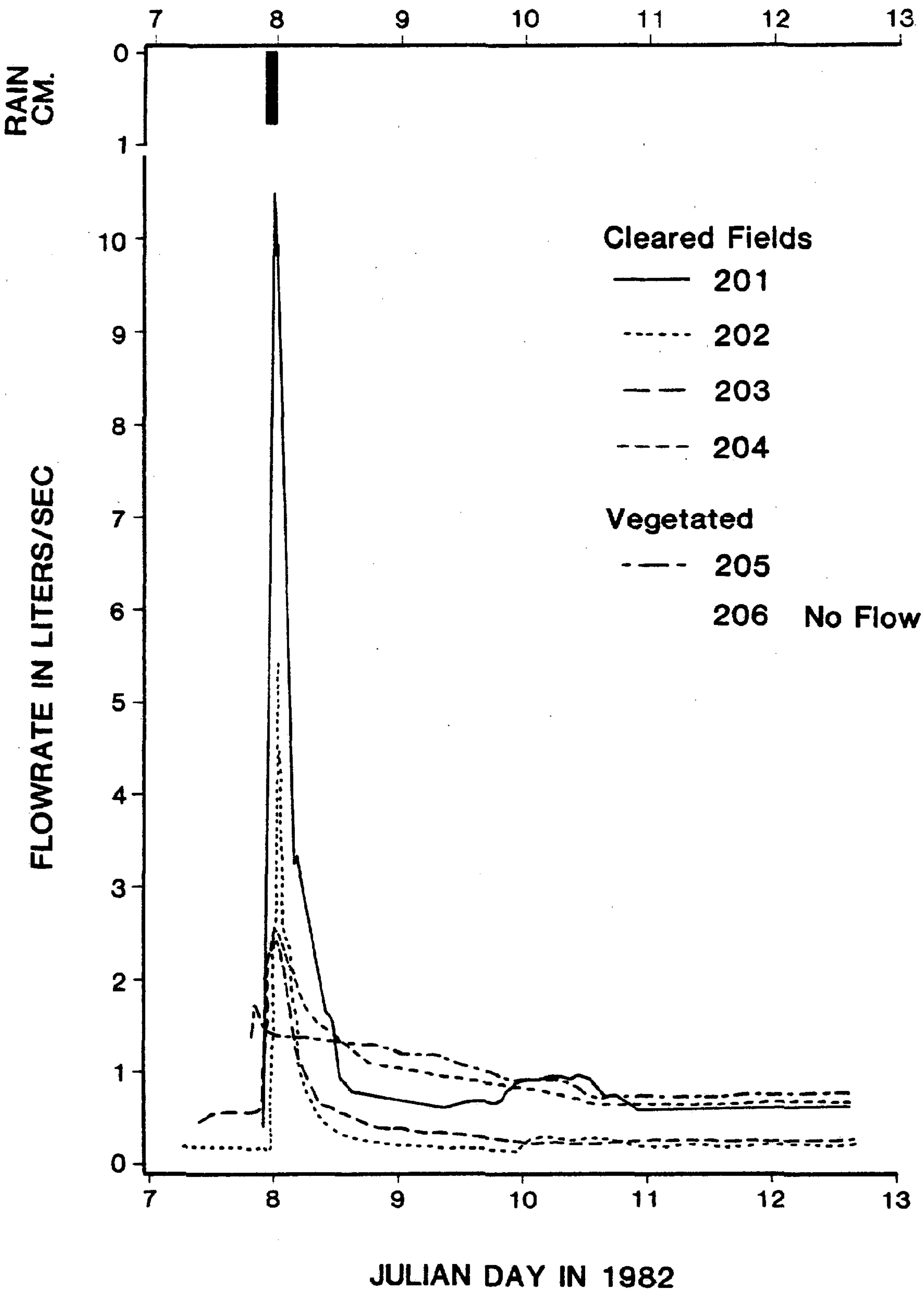


Figure 16. Runoff from field ditches draining cleared and vegetated sites (4 ha in size) for a storm in January, 1982.

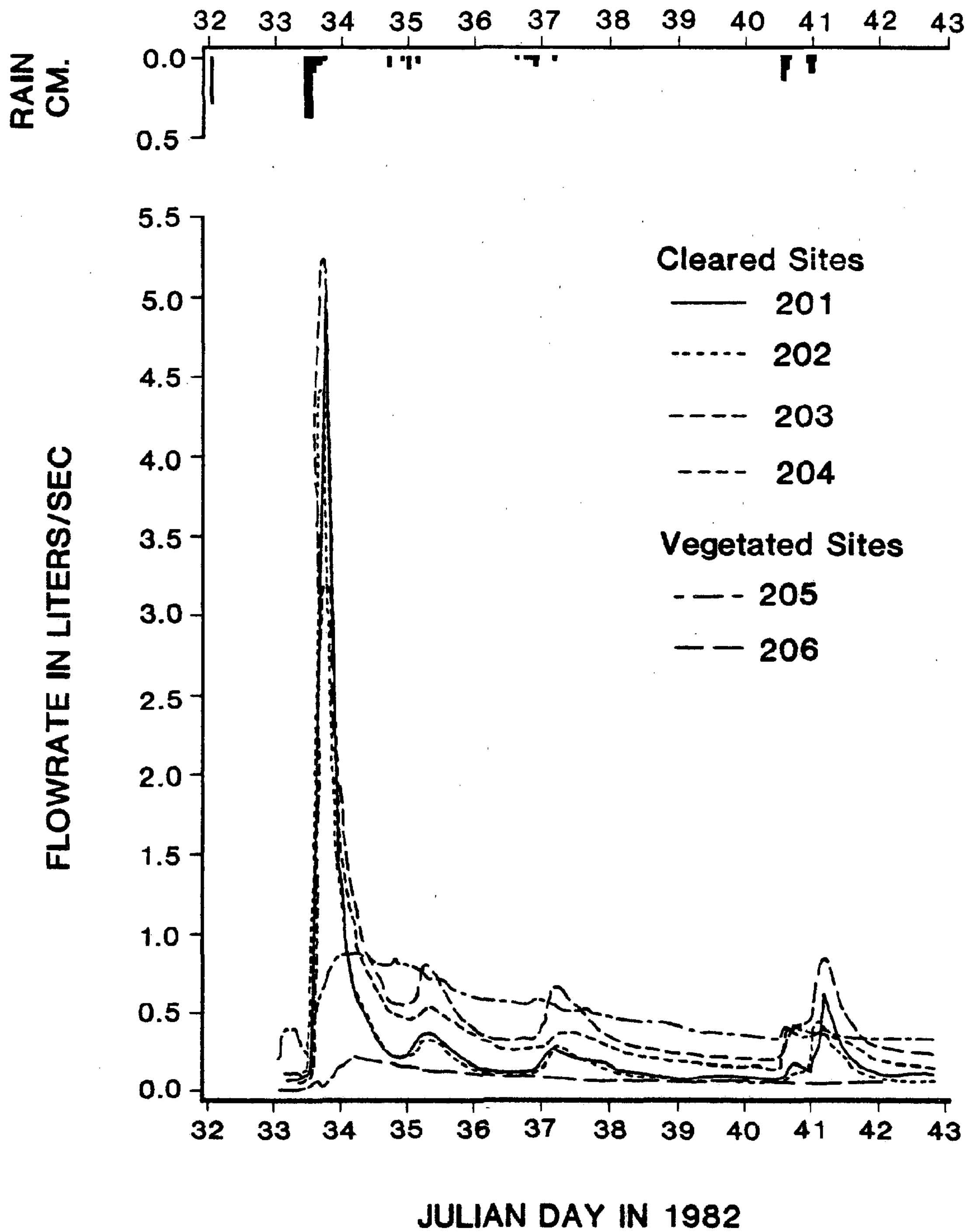


Figure 17. Runoff from field ditches draining cleared and vegetated sites (4 ha in size) for a storm in February, 1982.

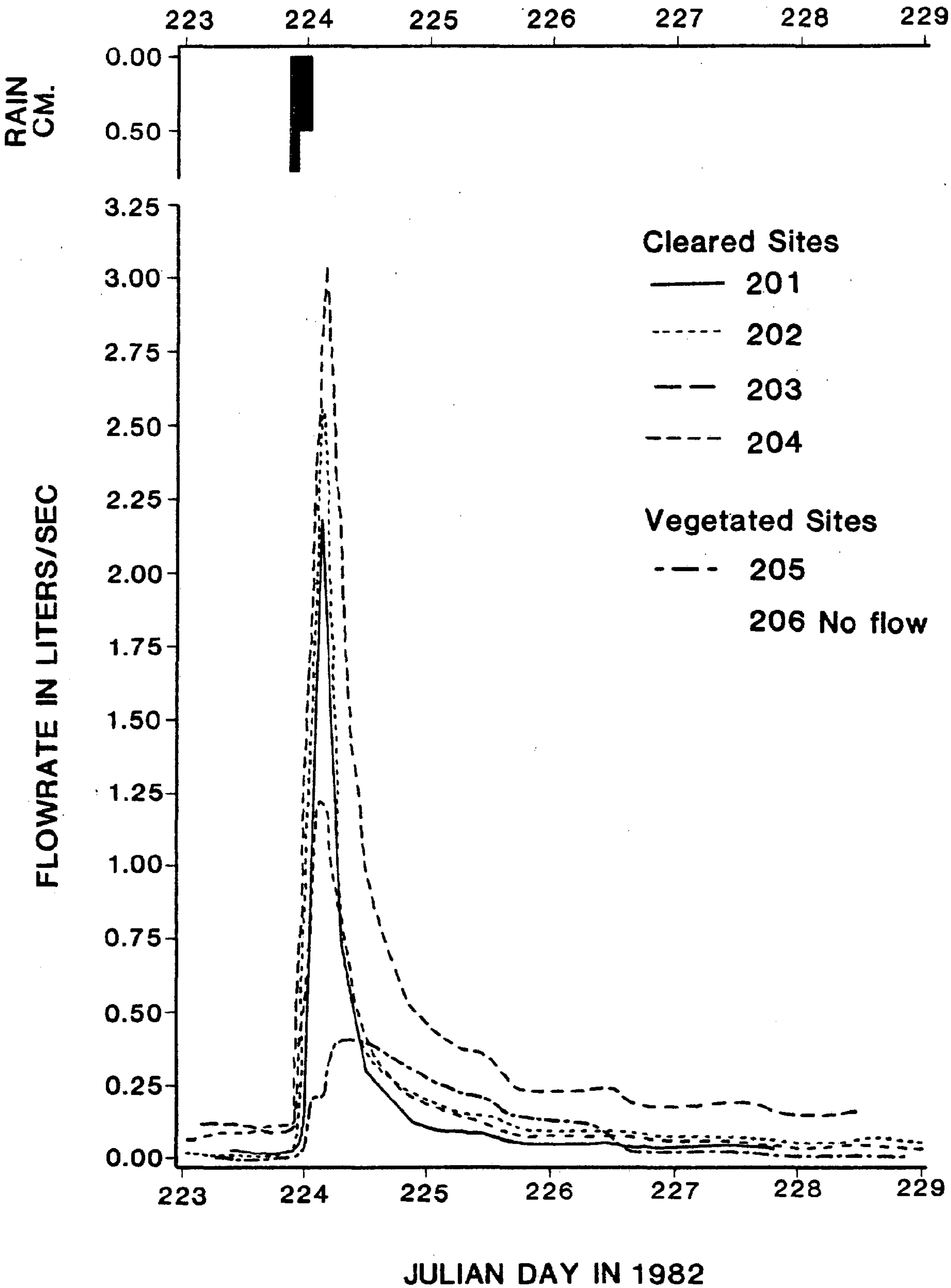


Figure 18. Runoff from field ditches draining cleared and vegetated sites (4 ha in size) for a storm in August 1982.

storm left the site as stormflow and baseflow was generally greater in the cleared fields than in the vegetated fields. No discharge occurred from Sites 205 and 206 for the storm in late December 1981 (Figure 15), and no discharge occurred from Site 206 for two of the other three storms (Figures 16-18). The consistent difference in runoff between Sites 205 and 206 was probably due in part to a difference in vegetation adjacent to the drainage ditch. The Bros Rotor Mixer was used to clear a strip of vegetation for installation of the field ditches. That cleared strip was much wider at Site 205 than at Site 206.

The changes in surface conditions also contributed to the increased runoff from the cleared sites. Surface detention storage was virtually eliminated and a surface gradient to the ditches was created by the grading operation that followed grinding of the vegetation. During sequential mining operations, the grading was repeated as needed to maintain a smooth surface that maximized the rate of surface drainage. Infiltration conditions probably were not significantly altered until at least 15-20 cm of the peat (with residue from grinding of surface vegetation and buried wood) at the surface was removed. Only the fields draining through Site 201 were mined to that depth in 1981. Under natural conditions, the surface layer (0-15 cm) of the Pungo series has high infiltration capacity and high hydraulic conductivity. The peat in the surface layer has granular structure due to irreversible aggregation during drying cycles. The surface also has a high density of roots. Bog preparation with the Bros Roter Mixer grinds the vegetation above the surface and the roots and wood below the surface and mixes the chips with the granular peat. The resulting surface layer also has high infiltration capacity and hydraulic conductivity. After the granular layer is removed by mining, infiltration conditions probably deteriorate somewhat. The deep milled layer is moist, has massive structure and is highly susceptible to puddling and compaction. Impact of raindrops on the unprotected surface can cause puddling that significantly reduces infiltration capacity.

Other factors that may contribute to increased runoff from a site are the turning, drying, and the frequent removal of the 4-6 cm layer at the surface. The thin fluffy layer at the surface that is being turned and dried serves to insulate the peat beneath and inhibit evaporation and the water content there probably remains relatively high. The only layer with significant water storage capacity is removed as the moisture content decreases to 40 percent. These factors may account for the drastic difference in runoff volume and peak flow among the cleared sites for the storm of December 18, 1981 (Figure 15). Peat was mined from the fields of Site 201 during August-November and from Site 202 during August-September. No mining was done on the other two sites. About 6 cm of rain had fallen during the first two weeks of December following an extremely dry autumn (Figure 19). The soil in the fields of Sites 203 and 204 probably had more water storage capacity and greater infiltration

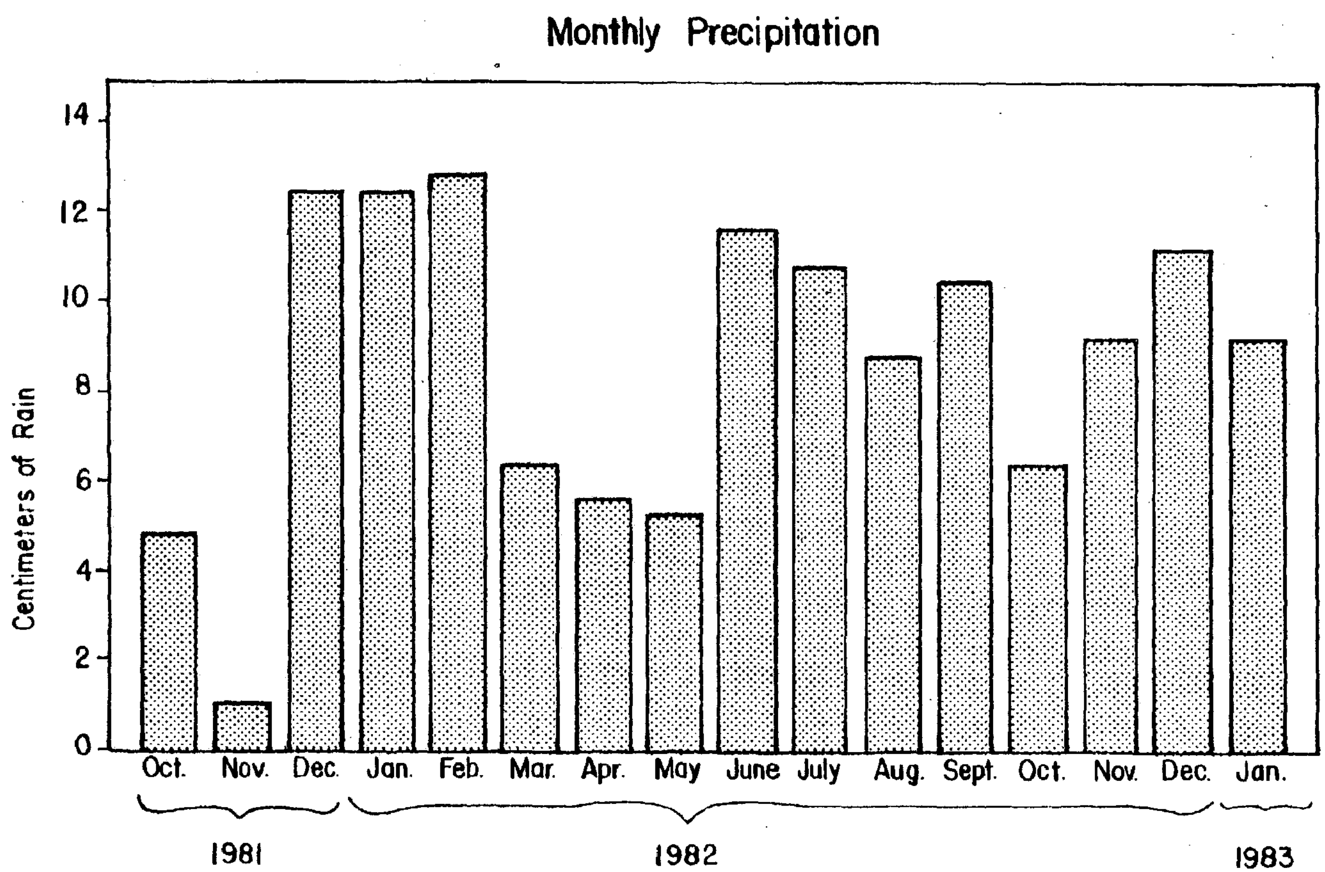


Figure 19. Monthly rainfall at the peat mining site.

capacity than Site 201, with Site 202 being intermediate. The runoff differences among the four cleared sites decreased with time after mining ceased and the normal rains of winter recharged soil moisture.

These data indicate that storm runoff and peak flow at the field edge will increase substantially when the field is first prepared for mining and a further increase may result as mining progresses and the surface layer is removed. There probably will be a slight decrease in storm discharge and peak flow when mining causes the surface to approach the peat-mineral soil interface and the field ditches have been deepened into the mineral soil. The higher conductivity of the mineral soil will result in greater rates and duration of base flow. Thus, more soil moisture storage will be available when rainfall occurs.

The hydrology at the field edge after reclamation will depend on how the land is used. The hydrology of agricultural fields will be similar to that of agricultural fields on mineral soils (Skaggs, et al., 1980; Chapter IV). Storm runoff and peak flow from forest plantations will depend on the stage of growth. Preparing the sites for planting will include bedding, a process that produces a tremendous amount of surface detention storage. Storm runoff and peak flow from young plantations (1-5 years of age) will be less than what occurred during peat mining but more than from the site prior to site preparation for peat mining. Typically, as intensively managed forest plantations on pocosin sites develop, the field ditches are not maintained and leaf area index and average rooting depth surpass that of the natural pocosin vegetation. Thus, when canopy closure occurs (5-10 years of age), drainage efficiency is decreased and ET will probably exceed that in natural vegetation. Storm runoff and peak flow will then be less than with natural vegetation.

How the changes in hydrology at the field edge that occur with site preparation and peat mining will influence downstream hydrology depends on a number of factors. The flooding of the drainage system that occurs with larger storms indicates that the discharge capacity of the collector canals and main canals is not adequte to maintain free flow from the field ditches. Thus, the size of the flow increase at the field edge that results from mining will be diminished downstream. Changes in downstream flows will also depend on the size and distribution of areas being mined at any one time, reclamation procedures, water management practices, etc. The characteristics of flow in the main canals are analyzed in detail in Chapter IV.

Water Quality

The quality of water in rivers of the Lower Coastal Plain of North Carolina has been the subject of intense debate over the last two decades as periodic massive algal blooms, indicative of eutrophication, have occurred. Unpolluted water in lower Coastal Plain streams is moderately to highly acidic, low in nutrients, and dark colored with dissolved organic matter (Kuenzler, et al., 1977; Kirby-Smith and Barber, 1979). Rivers showing symptoms of eutrophication have higher concentrations of nitrogen and phosphorus than unpolluted waters (Copeland and Hobbie, 1972; Hobbie, et al., 1972; Stanley and Hobbie, 1977). For these reasons, water quality analyses for this project concentrated on nitrogen, phosphorus, organic sediment and other factors related to biological productivity.

Concentrations of nutrient cations and heavy metals are relatively low in waters of lower Coastal Plain streams and canals draining undisturbed areas (Skaggs, et al., 1980; Kuenzler, et al., 1977). The long term land alteration caused by agricultural development can cause some increases in concentrations of nutrient cations and heavy metals in drainage waters, however, those concentrations are relatively low in terms of water quality standards, particularly in water draining from organic soils (Skaggs, et al., 1980).

Non-Filtrable Solids and Turbidity

The milling and mixing processes used to prepare the peat surface for harvest of a thin layer (3-4 cm) produces a surface of high erodibility. The surface layer of the Pungo series to a depth of about 18 cm commonly has weak medium granular structure that is very friable due to aggregation of the colloidal sized organic particles during drying cycles. The milling process tends to break down those aggregates somewhat and produces a surface layer of low bulk density with a high content of very fine textured material. Relatively high erosion rates occur when the peat becomes saturated and overland flow to the field ditches occurs. The quantity of organic sediment moved out of the field ditches and the distance of movement through the drainage system is a function of volume and velocity of flow. Relatively high gradients at the downstream ends of the field ditches result in movement of substantial quantities of sediment. Rapid filling of the weir stilling ponds with organic sediment and wood fragments was indicative of high rates of sediment movement from the ditches and the surface adjacent to the ponds.

Concentration of nonfiltrable solids (suspended sediment) and turbidity of water flowing over the weirs was considerably higher in the cleared fields than in those remaining in natural vegetation (Tables 2 and 3). The difference in mean concentrations of nonfiltrable solids and mean turbidity values between Site 201 and the other three cleared fields illustrates the effects of degree and duration of surface disturbance. Mining was conducted on the mining strips draining through Site 201 during April-November 1981 and on the mining strips draining through Site 202 during August-September 1981. The surface remained bare but undisturbed on the other sites. No mining was done in 1982 so the sediment delivery from the fields during 1982 was considerably lower than would be expected under normal mining conditions. The weir stilling ponds acted as effective sediment basins. Therefore, the sediment delivery from fields without

Table 2. Average concentration of nonfiltrable solids in drainage waters.

		Sampling Sites									
		Field Ditches						Allen Canal		DeHoog Canal	
		Cleared				Vegetated					
Year	Mo.	201*	202*	203	204	205	206	301	302	601	602
		mg/l									
1981	8	401.0	151.3	125.3	168.4	120.0	7.0	-	-	-	-
	9	147.7	156.8	20.0	65.5	63.0	71.0	91.5	14.0	-	-
	10	878.0	10.0	-	-	-	-	63.0	20.0	8.0	17.0
	11	242.5	-	-	-	-	-	27.3	13.0	34.3	15.5
	12	105.8	83.0	29.0	25.0	14.0	-	79.0	11.0	11.0	4.0
1982	1	30.7	11.0	10.5	14.0	11.5	15.5	32.6	9.5	7.5	5.0
	2	19.0	16.5	18.2	16.2	12.5	6.0	22.0	19.0	11.7	10.3
	3	17.7	10.0	7.3	6.0	7.7	4.5	5.5	2.0	1.0	1.0
	4	31.0	-	25.0	-	10.0	-	29.0	24.2	6.0	4.3
	5	137.5	-	-	-	7.0	-	20.5	23.3	3.2	4.7
	6	73.6	208.0	72.5	95.0	32.3	9.0	63.0	24.2	14.2	5.5
	7	8.5	15.5	33.7	56.5	-	-	8.0	26.0	7.2	3.7
	8	9.3	17.0	17.0	18.0	10.0	-	15.4	7.0	9.8	3.0
	9	67.0	69.0	8.0	6.0	8.0	6.0	11.7	15.2	6.5	4.5
	10	22.5	7.3	3.3	6.0	6.0	-	9.0	9.2	3.8	2.5
	11	73.0	5.8	5.8	5.4	3.8	3.0	6.8	5.4	5.4	2.4
	12	16.0	1.0	1.5	0.5	0.5	1.5	6.0	2.7	3.0	2.0
1983	1	12.3	1.0	2.0	1.7	1.8	2.0	2.5	3.8	1.0	1.5
Mean		127.39	50.9	25.3	34.6	20.5	12.6	29.0	13.5	8.4	5.4

* Had silt screen in field ditch

Table 3. Average turbidity of drainage waters.

		Sampling Sites									
		Field Ditches						Allen Canal		DeHoog	
		Cleared				Vegetated					
Year	Mo.	201*	202*	203	204	205	206	301	302	601	602
		NTU									
1981	9	155	31	17	38	-	-	20	25	-	-
	10	1100	29	-	-	-	-	42	28	20	18
	11	449	-	-	-	-	-	33	28	25	21
	12	178	187	31	31	24	-	54	24	24	9
1982	1	61	23	20	22	18	17	18	12	9	8
	2	24	20	15	18	13	14	15	12	8	8
	3	23	18	14	16	13	14	12	10	7	7
	4	58	-	20	-	23	-	24	19	7	7
	5	263	-	-	-	15	-	22	29	6	6
	6	73	149	103	112	26	14	25	26	6	5
	7	28	41	46	65	-	-	20	24	9	5
	8	29	20	16	26	12	-	20	20	4	5
	9	116	35	6	7	9	4	15	16	6	3
	10	70	15	8	9	12	-	12	16	2	2
	11	106	7	5	4	7	6	6	7	1	2
	12	50	6	2	3	6	6	8	9	1	1
1983	1	36	8	8	8	8	9	8	8	4	4
	Mean	166	42	22	28	14	11	21	18	9	7

* Had silt screen in field ditch

these basins at the ends of the field ditches is likely to be higher than measured here in runoff over the weir.

A significant portion of sediment in the field ditches appeared to have originated from erosion of the channels. The rate of erosion from the channel banks should decrease with time as the banks stabilize, particularly if natural vegetation is allowed to regenerate on the sides of the channel banks.

Comparing the monthly means of nonfiltrable solids concentrations and turbidity from Site 201 to the other three cleared sites indicated some possible hydrologic impacts of peat removal. Discharge from Site 201 was greater than from Sites 202-204 for most storm events recorded from August 1981-January 1983. The missing values in Tables 2 and 3 indicate months in which there was no flow on any of the sampling days. There was some "catchup" of sediment yield from Sites 202-204 when three relatively dry months (March-May, 1982) were followed by four months (June-September, 1982) of normal rainfall. However the surface of the mining strips at Sites 202-204 stabilized (some regrowth of forbs and grasses had started) and relative suspended sediment concentrations were about the same as for the uncleared sites during September, 1982 to January, 1983. During that period, the concentration of nonfiltrable solids and turbidity values were much higher in discharge from Site 201 than from the other field ditches. These data indicate that peat removal, probably associated with some compaction from machinery operation, tends to reduce storage capacity of the soil and increase the erodibility of the surface. The milled layer at the surface prior to initiation of mining has a high content of woodchips from the surface vegetation. The peat at the surface is a mixture of weak granules and fine, powdery material. Removal of the surface 18-24 cm exposes a surface that probably has less wood chips and a much higher proportion of very fine-textured, powdery material after drying than does the surface layer.

Suspended sediment concentrations and turbidities decrease as drainage water flows from the field ditches and through the collector and main canals. This effect results both from settling of suspended sediment and dilution as water from downstream collector canals draining undisturbed areas enters the main canal. Site 301 on Allen Canal was about 2.8 km downstream and Site 302 was about 5.6 km downstream from the entry point of the collector canal draining the field ditches in the cleared area. Allen Canal was cleaned with a dragline after the wildfire so the difference in channel stability may be a contributing factor to the differences in suspended sediment concentrations between the two canals, in addition to the input of sediment to Allen Canal from the cleared area.

The silt screens installed in the ditches at Sites 201 and 202 had no effect on movement of suspended sediment to the weir location. The screens quickly clogged with fine organic material and wood fragments collected behind them. Water then flowed around and over the screens and eroded the ditch banks. Much sediment also entered the stilling ponds from the adjacent surface and from sloughing of the banks around the ponds.

The variation in nonfiltrable solids concentration and turbidity among sampling sites and over time was influenced by sampling factors. Collection of grab samples each week extended over a period of several hours because of the time required for in situ analyses and for travel time among sampling sites. That factor would result in sampling the field ditches at different segments of the hydrograph, even when those hydrographs were similar in shape.

Collection of weekly grab samples very seldom coincided with stormflow from the mining strips that was near peak flow rates. Therefore some stormflow samples were collected to compare the relative magnitudes of suspended sediment concentration in outflow near the stormflow peak with suspended sediment concentration at lower flow rates (Table 4). These data indicate that weekly grab sampling did not adequately sample the periods of highest suspended sediment concentrations. Therefore the mean suspended sediment values are conservative compared to actual mean sediment concentrations.

Notwithstanding the conservative nature of the sampling procedure, several conclusions can be drawn from the suspended sediment data. Surface mining of milled peat results in substantial increases in suspended sediment concentration in field ditch outflow. Surface erosion from the mining strips and erosion of the ditch banks contribute to the sediment yield. Suspended sediment concentrations decrease rapidly as ditch outflow moves through the collector and main canals. This is due to settling of the suspended material as well as dilution with outflow from undisturbed areas. Silt screens in the field ditches are not effective in controlling suspended sediment movement. However, sediment basins with flashboard risers to manage flow appear to be very effective in collecting sediment discharge from field ditches. Such sediment basins would require regular cleaning to maintain effectiveness.

Nitrogen

Concentrations of nitrogen in drainage waters were extremely low and highly variable among sampling locations and over time (Tables 5-7). Flow from the field ditches was not continuous.

Table 4. Comparison of nonfiltrable solids concentration in a weekly grab samples vs. a stormflow sample collected the same week.

Site	Nonfiltrable Solids		Peak Stage	Time of Peak (1/4/82)
	Stormflow	Weekly		
	mg/l		cm	
201	78	32	118.2	11:00 AM
202	118	19	119.6	12:00 Noon
203	83	20	88.5	12:30 PM
204	57	25	108.2	12:00 Noon
205	62	15	44.4	11:00 AM
206	42	9	73.4	11:00 AM

Storm Description: 3.9 cm rain fell between 6 and 7 pm on 1/3/82, 1.2 cm rain fell between midnight and 4 am on 1/4/82 and 1.7 cm rain fell between 9 and 10 am on 1/4/82.

Sampling Times: Stormflow samples were taken between 5 and 6 pm on 1/4/82; weekly samples were taken between 10 and 11 am on 1/6/82.

Table 5. Average concentration of NH_4-N in drainage waters.

		Sampling Sites									
		Field Ditches						Allen Canal		DeHoog Canal	
		Cleared				Vegetated					
Year	Mo.	201*	202*	203	204	205	206	301	302	601	602
		mg/l									
1981	8	0.11	0.22	0.33	0.32	0.25	0.20	-	-	-	-
	9	0.18	0.35	0.26	0.42	0.24	0.10	1.95	1.80	-	-
	10	-	-	-	-	-	-	0.95	0.96	0.80	0.32
	11	0.11	-	-	-	-	-	0.71	0.84	1.40	1.38
	12	0.39	0.62	0.53	0.52	0.25	-	0.86	1.06	1.11	0.19
1982	1	0.16	0.57	0.53	0.67	0.20	0.14	0.58	0.65	0.10	0.10
	2	0.14	0.31	0.41	0.29	0.12	0.10	0.34	0.38	0.10	0.10
	3	0.11	0.25	0.19	0.26	0.14	0.10	0.23	0.36	0.10	0.10
	4	0.37	-	1.20	-	0.10	-	0.21	0.28	0.10	0.10
	5	0.12	-	-	-	0.12	-	0.39	0.72	0.10	0.10
	6	0.16	0.10	0.11	0.24	0.13	0.10	0.44	0.61	0.10	0.10
	7	0.16	0.10	0.10	0.10	-	-	0.13	0.22	0.10	0.12
	8	0.11	0.10	0.14	0.20	0.10	-	0.14	0.20	0.11	0.15
	9	0.44	0.91	0.85	0.89	0.54	0.54	0.56	0.50	0.52	0.69
	10	0.37	0.28	0.32	0.30	0.37	-	0.43	0.28	0.16	0.22
	11	0.40	0.35	0.31	0.34	0.34	0.34	0.40	0.43	0.13	0.10
	12	0.46	0.59	0.45	0.62	0.56	0.40	0.51	0.42	0.13	0.15
1983	1	0.35	0.39	0.46	0.36	0.50	0.45	0.42	0.55	0.12	0.19
Mean		0.24	0.36	0.41	0.40	0.26	0.25	0.55	0.60	0.32	0.26

* Had silt screen in field ditch

Table 6. Average concentration of NO_3-N in drainage waters.

		Sampling Sites									
		Field Ditches						Allen Canal		DeHoog Canal	
		Cleared				Vegetated					
Year	Mo.	201*	202*	203	204	205	206	301	302	601	602
		mg/l									
1982	3	0.47	0.43	0.44	0.55	1.00	-	1.10	0.52	0.80	0.80
	4	0.63	-	0.97	-	0.75	-	0.33	0.35	0.33	0.41
	5	0.34	-	-	-	-	-	0.36	0.39	0.37	0.50
	6	0.28	-	0.30	0.27	0.30	-	0.30	0.42	0.26	0.27
	7	0.36	0.35	0.35	0.36	-	-	0.46	0.75	0.30	0.37
	8	0.20	0.20	0.14	0.29	0.13	-	0.07	0.79	0.10	0.11
	9	0.13	0.13	0.32	0.16	0.26	0.52	0.18	0.89	0.24	0.18
	10	0.19	0.09	0.18	0.10	0.14	-	0.16	0.17	0.10	0.10
	11	0.18	0.03	0.06	0.04	0.09	0.15	0.04	0.08	0.02	0.01
	12	0.13	0.08	0.12	0.12	0.18	0.25	0.06	0.27	0.06	0.07
1983	1	0.10	0.01	0.07	0.02	0.12	0.08	0.12	0.18	0.07	0.02
Mean		0.27	0.17	0.29	0.21	0.33	0.25	0.29	0.44	0.24	0.25

* Had silt screen in field ditch

Table 7. Average Concentration of total Kjeldahl N in drainage waters.

		Sampling Sites									
		Field Ditches									
		Cleared				Vegetated		Allen Canal		DeHoog Canal	
Year	Mo.	201*	202*	203	204	205	206	301	302	601	602
		mg/l									
1981	9	1.24	1.05	0.34	0.70	-	-	-	2.01	-	-
	10	-	-	-	-	-	-	1.34	1.67	1.24	0.64
	11	3.58	-	-	-	-	-	1.38	1.23	2.39	2.16
	12	0.99	1.52	1.10	1.13	1.52	-	1.68	1.52	0.70	1.58
1982	1	1.83	1.93	1.96	2.36	2.40	1.67	1.88	2.14	1.37	1.55
	2	0.29	0.42	0.38	0.50	0.45	0.44	0.35	0.31	0.17	0.18
	3	0.65	0.96	1.01	1.22	1.14	0.99	0.67	0.55	0.23	0.18
	4	2.70	-	4.04	-	1.83	-	1.37	1.21	0.68	1.12
	5	4.29	-	-	-	1.96	-	1.86	2.21	1.23	1.46
	6	2.83	3.10	3.03	3.23	3.28	2.48	2.01	2.10	1.50	1.88
	7	2.62	2.70	3.76	2.68	-	-	1.64	1.60	0.96	1.31
	8	1.54	1.19	1.47	1.94	2.04	-	1.53	1.40	1.35	1.74
	9	-	-	-	-	-	-	1.77	1.64	1.84	2.33
	10	2.39	1.91	1.88	1.80	2.20	-	1.42	1.32	1.54	2.08
	11	3.48	1.71	1.35	1.84	2.47	2.14	2.08	2.07	1.72	2.14
	12	2.17	1.97	2.03	2.68	3.06	2.03	1.94	1.44	1.47	1.68
1983	1	1.53	1.25	1.48	1.44	2.13	1.72	1.56	1.30	1.02	1.20
Mean		2.14	1.64	1.83	1.79	2.04	1.64	1.53	1.51	1.21	1.45

* Had silt screen in field ditch

Duration of flow among the six field ditches varied somewhat so that the total number of water samples among all sites varied.

The average concentration of NH_4-N over the sampling period was slightly higher in water draining from the cleared fields than from the fields with natural vegetation. The NH_4-N concentration in Allen Canal was higher than in DeHoog Canal or the field ditches. It is not known if this difference was due solely to effects of the fire or also indicates a natural difference between the two canals. However, cation levels were much higher and pH somewhat higher in Allen Canal than in DeHoog Canal more than a year after the fire (Tables 10-13). Therefore, it is likely that the rate of biologic activity in soil and water was higher in the drainage area of Allen Canal than in that of DeHoog Canal throughout the study period. The concentration of NH_4-N was higher during the winter than during the summer at all sampling sites.

The mean concentration of NO_3-N was about the same for all sampling locations (Table 6). Comparisons over time are not possible since we believe that the ion electrode utilized for nitrate analyses during the period March-July 1982 was unreliable and the measured concentrations are probably somewhat higher than actual ones.

These data indicate that during the first 18 months following initiation of peat mining, there was no significant change in concentration of available nitrogen in drainage waters. Increased surface temperature and aeration resulting from the peat mining operation should increase the rate of organic matter decomposition and the release of available N from the thin surface layer. However, the surface layer is periodically removed and the water content of the peat probably remains high below the surface 3-4 cm due to the insulating effect of the milled surface and the lack of plant roots to withdraw water from greater depths. Therefore, there is likely no significant increase in decomposition rates. Also, the peat is already highly decomposed and the C/N ratio is very high, about 50/1, (Ingram and Otte, 1982). Little N is released on further decomposition. There was no difference in total Kjeldahl N concentration of water between mined and vegetated fields. The total Kjeldahl N concentration was slightly lower in the main canals than in the field ditches.

The average concentrations of all forms of N in drainage waters from both cleared and vegetated sites was considerably higher than those reported in other studies for similar sites (Kirby-Smith and Barber, 1979; Skaggs et al., 1980). The reasons for these differences are unknown. These data were not weighted for flow as were the flow-proportional data of Skaggs et al. (1980). Water samples were not taken when there was no flow over the weirs, but many samples were taken during low-flow conditions when near-stagnant conditions were present. Ashfall from the fire may have had a significant influence on water quality.

Additional research with more extensive sampling by flow proportional automatic samplers with concomitant sampling from similar but undisturbed sites and from a series of downstream locations should be conducted.

Phosphorus

The average concentration of total phosphorus was greater in water draining from Site 201 than from the other field ditches (Table 8). Most of the phosphorus present in the water was associated with the organic sediment, even though there was also an increase in orthophosphate in solution in the field where active mining had occurred in 1981 (Table 9). Orthophosphate and total phosphorus were both somewhat higher in the main canals than in the field ditches except for Site 201. That result may reflect the influence of drainage from the mineral layer beneath the peat. Mineral sediments are intersected by collector and main canals but not by field ditches.

Even though total phosphorus concentration in discharge from the field ditches was increased by mining, those concentrations were very low. Potential increases in phosphorus efflux can be controlled by controlling movement of suspended sediment from the mining site. The biological availability of phosphorus in the suspended sediment is also probably quite low. Such phosphorus is most likely present as a constituent of highly stable organic compounds in the peat. As with nitrogen the phosphorus concentrations were considerably higher than reported in other studies (Kirby-Smith and Barber, 1979; Skaggs et al., 1980).

Acidity

The acidity of field ditch outflow from the four cleared sites was slightly lower than from the two vegetated sites (Table 10). Mean monthly pH values consistently were in the range of 3.4 to 3.8 and there was no apparent seasonal variation. Transit time for rainfall to infiltrate and move through or over the soil to a ditch probably accounts for the difference in pH of field ditch outflow from cleared sites versus the vegetated sites. Average rainfall pH was about 4.5-5.0 during 1976-1979 (Skaggs, et al., 1980). Rain water has less transit time in the cleared fields and less time for reaction with the extremely acid organic soil. The highest pH values were measured in Allen Canal where leaching of cations from ash resulting from the wildfire was the likely cause. The relatively high pH values in DeHoog Canal during October-December 1981 are not characteristic of natural conditions. Flow conditions were artificially altered because of activities associated with development of the methanol conversion plant upstream of the sampling sites. The relatively high pH values occurred during a period of extremely low flow and stagnant conditions when a flashboard riser was installed. Depletion of CO_2 by biologic activity probably accounts for the elevated pH during that period. Windblown ash present in the drainage area of DeHoog Canal may have also been a factor.

Table 8. Average concentration of total phosphorus in drainage waters.

		Sampling Sites									
		Field Ditches: Cleared				Field Ditches: Vegetated		Allen Canal		DeHoog Canal	
Year	Mo.	201*	202*	203	204	205	206	301	302	601	602
		mg/l									
1981	9	0.28	0.14	0.04	0.11	-	-	-	0.41	-	-
	10	-	-	-	-	-	-	0.51	0.38	0.28	0.31
	11	0.28	-	-	-	-	-	0.38	0.33	0.42	0.34
	12	0.12	0.51	0.05	0.05	0.08	-	0.33	0.30	0.11	0.17
1982	1	0.08	0.08	0.09	0.08	0.02	0.02	0.62	0.40	0.31	0.13
	2	0.04	0.06	0.04	0.04	0.02	0.05	0.26	0.15	0.14	0.09
	3	0.46	0.52	0.37	0.50	0.50	0.09	1.05	0.71	0.54	0.52
	4	1.36	-	1.34	-	0.21	-	0.56	0.47	0.36	0.38
	5	0.72	-	-	-	0.08	-	0.59	0.67	0.40	0.28
	6	0.44	0.38	0.60	0.26	0.15	0.13	0.50	0.51	0.43	0.32
	7	0.59	0.10	0.17	0.05	-	-	0.28	0.43	0.25	0.27
	8	0.19	0.08	0.08	0.09	0.10	-	0.26	0.23	0.31	0.21
	9	-	-	-	-	-	-	0.21	0.17	0.19	0.19
	10	0.30	0.09	0.05	0.03	0.06	-	0.34	0.21	0.24	0.21
	11	0.27	0.11	0.10	0.06	0.06	0.08	0.66	0.52	0.32	0.26
	12	0.17	0.03	0.04	0.04	0.04	0.08	0.58	0.47	0.26	0.21
1983	1	0.20	0.01	0.01	0.01	0.01	0.02	0.58	0.45	0.23	0.19
Mean		0.37	0.18	0.23	0.11	0.11	0.07	0.48	0.40	0.30	0.26

* Had silt screen in field ditch

Table 9. Average concentration of orthophosphate in drainage water.

		Sampling Sites									
		Field Ditches									
		Cleared				Vegetated		Allen Canal		DeHoog Canal	
Year	Mo.	201*	202*	203	204	205	206	301	302	601	602
		mg/l									
1982	8	0.07	0.02	0.02	0.01	0.01	-	0.08	0.09	0.22	0.14
	9	-	-	-	-	-	-	0.15	0.08	0.19	0.18
	10	0.13	0.02	0.02	0.02	0.02	-	0.30	0.23	0.20	0.22
	11	0.23	0.04	0.03	0.02	0.02	0.01	0.32	0.30	0.21	0.16
	12	0.12	0.02	0.01	0.01	0.01	0.01	0.32	0.29	0.17	0.13
1983	1	0.12	0.03	0.02	0.02	0.02	0.01	0.32	0.27	0.15	0.11
Mean		0.13	0.03	0.02	0.02	0.02	0.01	0.25	0.21	0.19	0.16

* Had silt screen in field ditch

Potassium, Calcium, Magnesium, and Chloride

Mining of peat had no effect on the concentration of cations in discharge from the field ditches (Tables 11-13). The average concentration of K during the period August, 1982-January, 1983 ranged from 0.72-1.14 mg/l. Concentrations of Ca and Mg were much lower than K in drainage from field ditches. The concentration of Ca ranged from 0.16-0.33 mg/l and the concentration of Mg ranged from 0.37-0.60 mg/l.

The relative concentrations of Ca and Mg in discharge from the field ditches is rather unusual. The concentration of Ca in precipitation in North Carolina is usually higher than that of Mg (Skaggs, et al., 1980; Simmons and Heath, 1982) and the concentration of Ca is usually much higher than Mg in runoff from undisturbed, forested watersheds (Simmons and Heath, 1982). The K and Mg concentrations are similar but the Ca values are much lower than the concentrations reported by Skaggs, et al. (1980) for discharge from fields in the same area with deep organic soils and natural vegetation. The low concentrations of these important nutrient cations in comparison to waters draining mineral soils also illustrate the ombrotrophic nature of these deep organic soils (Skaggs, et al., 1980; Simmons and Heath, 1982).

Table 10. Average pH in drainage waters.

		Sampling Sites									
		Field Ditches						Allen Canal		DeHoog Canal	
		Cleared				Vegetated					
Year	Mo.	201*	202*	203	204	205	206	301	302	601	602
1981	8	3.92	3.74	3.63	3.56	3.47	3.46	-	-	-	-
	9	3.72	3.68	3.64	3.53	3.45	3.49	5.46	5.60	-	-
	10	-	-	-	-	-	-	6.27	6.62	5.18	5.74
	11	3.75	-	-	-	-	-	6.82	6.89	5.96	6.26
	12	3.62	3.62	3.59	3.51	3.42	-	4.77	5.51	6.19	4.22
1982	1	3.80	3.73	3.69	3.56	3.43	3.44	3.88	3.93	3.61	3.55
	2	3.82	3.71	3.71	3.60	3.45	3.47	4.82	4.81	3.69	3.65
	3	3.79	3.76	3.73	3.65	3.50	3.51	4.61	4.21	3.69	3.65
	4	3.54	-	3.47	-	3.42	-	5.74	6.07	4.00	3.82
	5	3.68	-	-	-	3.52	-	5.47	5.85	4.09	3.97
	6	3.79	3.86	3.84	3.76	3.60	3.68	4.37	4.09	3.97	4.01
	7	3.72	3.68	3.69	3.63	-	-	5.17	4.41	3.84	3.83
	8	3.67	3.60	3.63	3.56	3.42	-	4.43	4.92	3.92	3.80
	9	3.68	3.63	3.60	3.53	3.45	3.56	4.42	4.57	4.38	4.35
	10	3.68	3.61	3.59	3.56	3.48	-	4.42	4.40	4.02	4.07
	11	3.75	3.73	3.73	3.61	3.47	3.54	4.16	4.45	3.64	3.61
	12	3.75	3.75	3.72	3.60	3.45	3.53	4.22	4.16	3.67	3.57
1983	1	3.79	3.77	3.77	3.66	3.48	3.57	4.02	4.20	3.63	3.59
Mean		3.73	3.70	3.66	3.59	3.49	3.52	4.46	4.47	3.90	3.83

Note: Average pH`s were computed by taking negative log of mean H^+ concentrations.

Table 11. Average concentration of K in drainage waters.

		Sampling Sites									
		Field Ditches						Allen Canal		DeHoog Canal	
		Cleared				Vegetated					
Year	Mo.	201*	202*	203	204	205	206	301	302	601	602
		mg/l									
1982	8	0.50	1.34	1.20	0.62	2.46	-	2.65	2.60	2.11	2.70
	9	-	-	-	-	-	-	3.65	2.29	4.10	3.73
	10	1.20	1.01	0.36	1.71	1.22	-	4.57	2.81	2.64	2.04
	11	0.72	0.64	0.71	0.59	0.68	0.91	2.00	1.90	1.15	1.16
	12	0.65	0.55	0.65	0.66	0.43	0.47	0.65	1.64	1.01	0.58
1983	1	0.53	0.53	0.80	0.89	0.93	0.79	1.34	1.87	0.75	0.62
Mean		0.72	0.81	0.74	0.89	1.14	0.72	2.48	2.19	1.96	1.81

* Had silt screen in field ditch

Table 12. Average concentration of Ca in drainage waters.

		Sampling Sites									
		Field Ditches						Allen Canal		DeHoog Canal	
		Cleared				Vegetated					
Year	Mo.	201*	202*	203	204	205	206	301	302	601	602
		mg/l									
1982	8	0.26	0.17	0.29	0.26	0.24	-	6.49	14.03	2.12	2.40
	9	-	-	-	-	-	-	11.90	18.03	3.08	3.28
	10	0.21	0.35	0.27	0.25	0.36	-	8.38	13.33	2.42	2.71
	11	0.13	0.26	0.18	0.24	0.39	0.20	6.00	9.85	1.19	1.43
	12	0.10	0.16	0.13	0.16	0.34	0.16	6.34	8.37	1.20	1.07
1983	1	0.10	0.12	0.12	0.15	0.35	0.18	4.94	7.72	0.91	0.85
Mean		0.16	0.21	0.20	0.21	0.33	0.18	7.33	11.89	1.82	1.96

* Had silt screen in field ditch

Table 13. Average concentration of Mg in drainage waters.

		Sampling Sites									
		Field Ditches						Allen Canal		DeHoog Canal	
		Cleared				Vegetated					
Year	Mo.	201*	202*	203	204	205	206	301	302	601	602
		mg/l									
1982	8	0.55	0.60	0.44	0.64	0.64	-	2.30	3.09	1.49	1.55
	9	-	-	-	-	-	-	2.52	3.05	1.91	1.66
	10	0.67	0.63	0.58	0.63	0.71	-	2.51	2.99	1.58	1.68
	11	0.34	0.26	0.33	0.49	0.62	0.51	2.15	2.82	1.26	1.41
	12	0.19	0.28	0.24	0.37	0.52	0.42	2.05	2.59	1.24	1.27
1983	1	0.25	0.26	0.24	0.39	0.53	0.42	1.95	2.54	1.11	1.12
Mean		0.40	0.41	0.37	0.50	0.60	0.45	2.25	2.85	1.43	1.45

* Had silt screen in field ditch

The extremely high concentration of cations in Allen Canal were the result of the high concentration and solubility of these elements in the ash accumulation in the drainage area of Allen Canal. Outflow from mineral soil beneath the peat in the deeper collector canals and main canals would also contribute to higher concentrations of cations in the main canals. Soil disturbances associated with development of the methanol plant and the presence of wind-blown ash probably also influenced the relatively high concentrations of cations in DeHoog Canal.

The behavior of chloride ion is of interest because it has very low chemical and biologic reactivity. The presence of Cl in drainage waters is mainly a function of Cl sources and the hydrologic characteristics of the ecosystem. The concentration of Cl at all sampling sites (Table 14) was considerably higher than that previously recorded in rainfall but lower than concentrations in water draining from mineral soil (Skaggs, et al., 1980). Mining activities had no influence on Cl concentrations, but release of ash resulted in elevated levels in Allen Canal.

Dissolved Oxygen

Average concentration of dissolved oxygen did not vary significantly among all the sampling sites (Table 16). The lowest monthly average concentrations of 1-2 mg/l accurred in near stagnant pools behind weirs in the low flow conditions of late summer. Monthly average concentrations up to almost 10 mg/l occurred in the high flows of winter. These relatively low values are the result of high concentrations of organic carbon, high water temperature and low water velocity and are typical of dissolved oxygen values in lower Coastal Plain drainage basins (Kuenzler, et al., 1977; Kirby-Smith and Barber, 1979; Skaggs, et al., 1980).

Table 14. Average concentrations of Cl in drainage waters.

		Sampling Sites									
		Field Ditches						Allen Canal		DeHoog Canal	
		Cleared				Vegetated					
Year	Mo.	201*	202*	203	204	205	206	301	302	601	602
		mg/l									
1981	8	2.25	2.90	6.67	4.03	7.40	2.40	-	-	-	-
	9	6.70	7.42	7.12	8.18	6.88	5.87	10.65	10.60	-	-
	10	5.90	7.60	-	-	-	-	8.30	8.55	8.10	7.50
	11	5.02	-	-	-	-	-	7.17	7.58	7.40	7.78
	12	5.72	6.47	6.35	7.10	6.05	-	7.70	7.87	5.90	6.00
1982	1	4.03	4.72	4.90	5.48	5.15	4.13	9.17	8.40	5.43	5.25
	2	4.47	5.10	4.88	5.40	5.08	3.70	8.63	8.83	5.83	5.20
	3	4.17	4.37	4.00	4.48	3.92	3.40	8.50	8.10	4.83	4.80
	4	8.40	-	9.70	-	5.90	-	7.60	8.20	5.08	4.83
	5	5.65	-	-	-	4.55	-	7.08	7.35	5.45	4.78
	6	3.32	4.20	3.88	3.95	1.62	0.25	6.36	7.10	4.16	3.90
	7	5.22	4.70	5.53	4.85	-	-	6.20	5.52	3.32	3.82
	8	4.05	4.87	4.40	5.00	4.30	-	5.48	5.80	3.96	4.34
	9	3.00	3.90	3.20	3.30	0.30	2.60	9.45	6.85	3.62	4.42
	10	6.55	5.57	5.00	5.20	2.30	-	7.15	7.35	4.90	5.10
	11	3.60	4.62	4.20	3.98	3.80	1.95	6.52	7.18	4.90	4.55
	12	4.30	4.95	4.50	4.50	4.15	3.20	6.60	7.57	5.15	5.13
1983	1	3.90	4.13	3.72	4.40	3.72	2.10	6.10	6.22	4.32	4.08
Mean		4.79	5.03	5.20	4.99	4.34	2.96	7.57	7.59	5.14	5.09

* Had silt screen in field ditch

Table 15. Average concentration of dissolved oxygen in drainage waters.

		Sampling Sites									
		Field Ditches									
		Cleared				Vegetated		Allen Canal		DeHoog Canal	
Year	Mo.	201*	202*	203	204	205	206	301	302	601	602
		g/l									
1981	8	6.3	6.2	5.9	6.4	6.0	-	-	-	-	-
	9	-	-	-	-	-	-	-	-	-	-
	10	-	-	-	-	-	-	-	-	-	-
	11	-	-	-	-	-	-	-	-	-	-
	12	6.2	6.3	8.4	9.2	9.7	-	7.2	7.2	6.0	5.7
1982	1	5.0	4.9	5.4	4.8	7.3	7.0	7.0	8.0	6.9	5.9
	2	8.1	7.4	7.3	8.4	8.4	4.8	9.2	9.6	7.5	6.0
	3	7.3	6.5	7.2	6.9	8.2	8.3	9.5	9.5	7.8	7.2
	4	1.5	-	4.8	-	7.0	-	7.0	7.0	5.9	4.4
	5	1.2	-	-	-	6.3	-	4.4	4.1	4.5	3.1
	6	2.8	4.4	4.3	4.2	5.7	3.8	4.0	4.1	3.6	1.1
	7	2.4	2.8	2.0	3.4	-	-	3.7	3.9	3.2	1.2
	8	2.5	3.6	4.0	5.8	4.9	-	3.7	3.6	2.8	.8
	9	3.2	4.2	4.4	4.2	5.4	1.8	3.1	4.7	3.0	3.0
	10	4.6	4.9	3.9	5.9	6.2	-	3.5	4.3	3.8	4.0
	11	5.4	5.5	5.6	7.1	7.1	5.6	4.5	4.7	5.7	4.0
	12	7.8	8.0	7.6	7.3	8.3	7.8	5.2	6.1	5.5	4.8
1983	1	5.8	6.7	7.1	8.4	9.1	7.2	5.8	6.7	7.2	6.0
Mean		4.7	5.5	5.6	6.3	7.1	5.8	5.6	6.0	5.2	4.1

*Had silt screen in field ditch.

SUMMARY AND CONCLUSIONS

Field studies on the hydrologic and water quality impacts of peat mining were conducted at the pilot peat mine of First Colony Farms, Creswell, North Carolina (Washington and Hyde Counties). The 6,000 ha area that has been permitted by the state is located on a pocosin with an organic surface that ranges from 1.3 m to more that 2.5 m in depth.

Runoff was measured at the ends of six field ditches that drain 4 ha sites from August, 1981, to January, 1983. Two of the sites remained in natural vegetation and four were cleared and prepared for peat mining. Mining was conducted on two of the four cleared sites in the fall of 1981. Silt screens were installed in ditches draining two of the four cleared sites.

Water quality parameters were determined weekly for the six field ditches described above, at two locations in the main canal draining the area containing the study fields, and at two locations on a main canal draining an undisturbed area. Dissolved oxygen was measured in situ. Grab samples were analyzed in the laboratory for suspended sediment, turbidity, NH_4, NO_3, total N, pH, P, K, Ca, Mg, and Cl.

Conclusions of these studies are:

(1) The relative volume, duration, and peak flow of runoff from the field ditches were greatly increased by site preparation and further slightly increased by mining. The hydrologic changes result from reduction in ET, reduction in surface detention storage, and increase in surface gradient to the field ditches. Some reduction in infiltration capacity and hydraulic conductivity of the surface layer probably occurs after the surface layer (0-15 cm) of peat is removed.

(2) The relative increase in peak flow from the field ditches caused by mining probably diminishes downstream in the drainage system because of inadequate discharge capacity. Whether peat mining will cause substantial increases in peak flow in the main canals will depend on the proportion of the drainage area that is being mined at particular times, reclamation procedures, and water management practices utilized.

(3) The suspended sediment concentration and turbidity of field ditch outflow was increased by mining. Removal of protective vegetation; grinding, milling and turning the peat at the surface; and increased overland flow resulted in increased surface erosion. Organic sediment was also contributed to field ditches by wind deposition and by sloughing of ditch banks. Silt screens in the field ditches were ineffective in reducing sediment yields. Weir stilling

ponds (created by flashboard risers) were highly effective traps for sediment and wood. Suspended sediment concentrations were lower downstream on the main canals than at field edge due to dilution and sedimentation.

(4) The concentration of NH_4-N in field ditch outflow was slightly increased by peat mining. There was no change in concentration of NO_3-N or total Kjeldahl N.

(5) There was a substantial increase in the total P concentration of the field ditch outflow due to peat mining. However, the P concentration was very low and most of the P was present in an unreactive form in the organic sediment and can be controlled by controlling sediment movement.

(6) Peat mining resulted in a slight increase in pH of field ditch outflow.

(7) The concentrations of K, Ca, Mg, and Cl in field ditch outflow were not influenced by peat mining.

(8) Peat mining had no effect on the concentration of dissolved oxygen in field ditch outflow.

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IV. COMPUTER MODELS FOR ANALYSIS OF HYDROLOGIC EFFECTS OF PEAT MINING

R. G. Broadhead and R. W. Skaggs

INTRODUCTION

The hydrologic impacts of peat mining are unknown because such operations have never been conducted on the proposed scales in this area. Field monitoring can show up effects as they occur but some method of predicting and analyzing these effects before mining takes place must be developed if wise planning and permitting for future conditions is desired. The objectives of this study were to develop and test hydrologic models to accurately predict the storage and movement of water in the First Colony Farms' permitted peat mine area and then use the models to evaluate various mining and reclamation strategies in terms of their hydrologic effects. A field scale water management model (DRAINMOD) was used to analyze the in-field effects of mining and reclamation. DRAINMOD was linked to a numerical flood routing model to analyze the effects of mining and reclamation on canal flows.

It is proposed (from NPDES Permit Application for Phelps Peat Mine by PMA - May 31, 1983) that mining will begin in the center of the permitted area adjacent to the methanol plant. Drainage water from the initial mined blocks will be pumped to temporary settling and evaporation ponds. A lake will be formed surrounding the plant comprised of five 130 hectare blocks. Upon completion of the lake (in the tenth year of operation) all subsequent drainage water from all mining operations will be routed through the lake which has an anticipated depth of between one and three meters. Discharge from the lake will be through a control structure on Boerema Road canal (canal B, Figure 20). Average canal B discharge will be considerably increased while other main canal outlets from the area will be blocked off and serve only as collection areas for pumping stations. This report does not consider the use of the lake, but assumes that gravity drainage to the existing canal outlets will continue during and after mining. This represents a possible worst case scenario. The results presented herein may also be used to size pumps and predict failures for the system proposed by PMA.

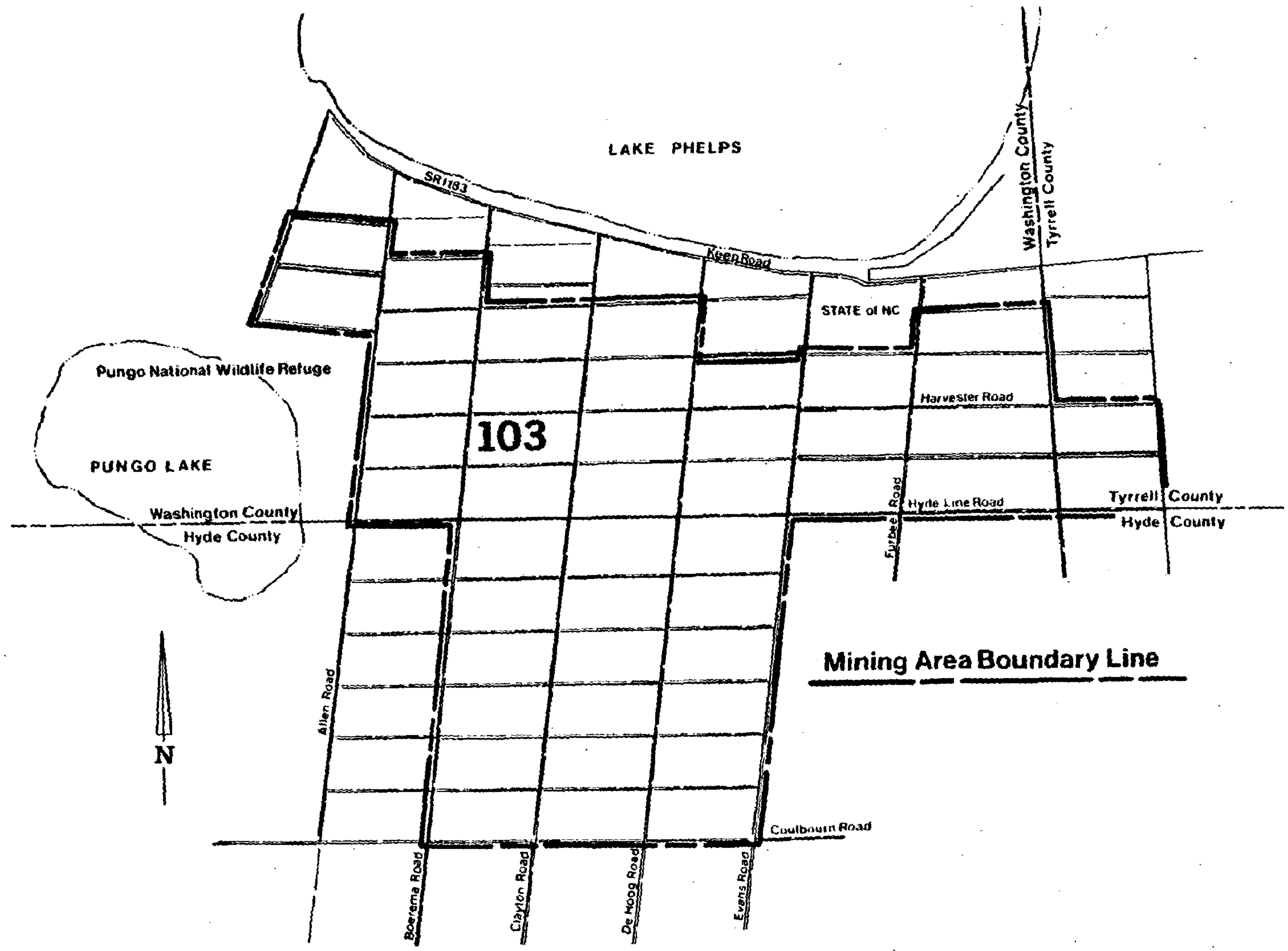

Figure 20. Permitted peat mining area on First Colony Farms.

MODEL DEVELOPMENT

Field Hydrology Model

The computer model, DRAINMOD, has been developed and validated for shallow water table mineral soils as a tool in the design of water management systems (Skaggs, 1978). It was developed for field scale applications and has the capability of simulating on a day-to-day, hour-by-hour basis the water table position, the soil water content, drainage, evapotranspiration and surface runoff in terms of climatological data, soil properties and the water management system.

The basis of the model is a soil water balance for a column of soil of unit surface area which extends from the impermeable layer to the surface and is located at the mid-point between adjacent field drains. The water balance for a time increment Δt may be expressed as;

$$\Delta Va = D + ET + DS - F \qquad (1)$$

where ΔVa is the change in the air volume, D is the drainage from the column, ET is the actual evapotranspiration, DS is the deep seepage and F is the infiltration entering the section in Δt. All values have units of cm^3/cm^2 or cm. The surface runoff and surface storage are computed from a similar water balance at the surface and may be written as;

$$P = F + \Delta S + RO \qquad (2)$$

where P is the precipitation, F is the infiltration, ΔS is the change in surface storage and RO is the surface runoff during time Δt. All values are in cm. The basic time increment used in DRAINMOD is one hour. For periods when rainfall exceeds infiltration capacity the time increments are reduced to three minutes and conversly when rainfall does not occur and the drainage rates are low, the time increment is increased first to two hours and then to one day.

Model Components

DRAINMOD is composed of a number of separate components, incorporated as subroutines, which evaluate the various mechanisms of water movement and storage in the soil profile. In order to keep computation time to a minimum and to keep data input requirements within reach of an average user, approximate methods were used for many of the components. Each approximate method has been evaluated as being satisfactory although, because of the component system, improved methods could be easily substituted. The component descriptions given below focus on the modifications developed for this project. A complete description of DRAINMOD as developed for field scale applications on mineral soils is given by Skaggs (1978). Details of field tests and applications of the model are given by Skaggs (1982), Skaggs *et al.* (1981), Skaggs and Gilliam (1981), Gayle (1982) and Chang *et al.* (1983).

Precipitation: Hourly rainfall data are used as input to the model. These are available for many North Carolina stations in the computer based HISARS (Wiser, 1972, 1975) for long periods of record. Hourly rainfall data are also available for the First Colony Farms permitted mining area from this study and a previous one in this area (Skaggs *et al.*, 1980).

Infiltration: Infiltration is determined from the simplified form of the Green and Ampt (1911) equation. For a given soil the equation may be written as;

$$f = A/F + B \tag{3}$$

where f is the infiltration rate, F is the accumulated infiltration and A and B are parameters that depend on the soil properties, initial water content and distribution and surface conditions. The model requires inputs for infiltration in the form of a table of A and B values versus water table depth.

Surface Runoff: Surface runoff occurs once the average surface depressional storage has been satisfied. It is assumed that the depressional storage is evenly distributed over the field and is estimated from site observation.

Subsurface Drainage: Subsurface drainage as it effects the mid-point water table depth is determined from the Hooghoudt equation for steady state drainage which may be written as;

$$q = \frac{8\ Kd_e m\ +\ 4\ Km^2}{L^2} \tag{4}$$

where q is the drainage flux in cm/hour, K is the effective saturated lateral hydraulic conductivity, m is the mid-point water table height above the drain bottom, L is the distance between drains and d_e is the equivalent depth from the drain to the impermeable layer. Flow line convergence close to the drains is in violation of the Dupuit-Forcheimer assumptions used in deriving equation (4). However, the use of d_e in equation (4) corrects for this.

Bank Storage: The Hooghoudt equation described above assumes a developed (approximately elliptical) water table profile at all times. In a typical agricultural field on mineral soil with normal conductivities this assumption appears to hold. However, with the very poorly drained and highly saturated peats this assumption falters. Water table profiles in this area become essentially horizontal from the mid-point to within a few meters from a drainage ditch after a rainfall. In this situation the water lost to the ditches (both the field ditches and the collector canals) from the banks adjacent to the ditches, in the development of the elliptical profile, becomes significant. Not only is the actual volume of water stored in the banks greater than for mineral soils (in relation to their relative drainable porosities) but also the flow from the banks to the ditches is a far more significant proportion of the total water lost from the soil profile. This is because of the high gradients developed close to the ditches and the low lateral drainage due to the wide ditch spacings and low hydraulic conductivities.

Numerical solutions to the Boussinesq equation were found for the various situations simulated in this project. The Boussinesq equation may be written as;

$$\frac{\partial h}{\partial t} = \frac{\partial}{\partial x} \left[-K \frac{\partial h}{\partial x}\right] + R \qquad (5)$$

where h is the height of the water table from the impermeable layer at position x from the drainage ditch or pipe, K is the saturated lateral hydraulic conductivity and R is the net rainfall less evapotranspiration. Solution to the Boussinesq equation results in a water table profile from drain to drain as a function of time. The drainage flux from the banks into the ditches was calculated from Darcy's Law knowing the slope of the water table at the drain. It was observed for typical ditch spacings of about 100 m on peat soils, that drainage from the banks to the drains did not affect the mid-point water table depth even over the longest period of dry weather on record for First Colony Farms. This meant that bank drainage determined by the Boussinesq equation could be added to the drainage flux calculation without fear that the drainage flux was being double counted. The Boussinesq bank flow losses were determined for the field ditches and collector canals as a function of water table depth. An exponential equation relating the bank drainage fluxes to the water table depth was generated by non-linear least squares regression for a typical 800 m *100 m (0.5 mi * 330 ft) field on the First Colony Farms (FCF) mining site, for conditions prior to mining. This equation was included in the subsurface drainage subroutine. As a result the water table depth predicted is not the true mid-point water table depth but an average water table depth for the field. It is somewhat greater than the true mid-point water table depth. While the bank storage conponent is essential to accurate predictions of water flow in the ditches, it was removed when testing the model against observed mid-point water table depths. The mid-point water table depth on pre-mined FCF sites is controlled primarily by ET and infiltration.

The empirical equation for bank storage flow was determined for a period when the water table levels in the ditches were known. The flow from bank storage was limited immediately after a rainfall when the water level in the ditches was high and the bank outflow was thus determined more realistically than if a constant ditch water level was assumed. Despite such precautions the mechanisms for storage and release of water from the banks under the many varied boundary conditions that can occur remains a problem and more work needs to be done in this area. The current technique is most prone to error immediately following a rainfall which does not bring the water table to the surface. In this case a brief sudden increase in flow would be expected but a smaller increase is predicted by the model as a result of the bank outflow being dependent on the water table depth alone. Ideally, bank outflow should be tied not only to the water table depth but to the volume of previous infiltrations and to the time since the last rainfall.

Evapotranspiration: The determination of actual evapotranspiration (ET) within the model is a two step process. First the daily potential evapotranspiration (PET) is calculated from climatological data and distributed as an hourly value over 12 daytime hours. The second step is to check the ability of the soil water to supply PET and determine if this is limiting. If soil water is considered unable to supply enough water to satisfy the PET then the ET is set equal to the upward flux value. The upward flux value is the amount of water the soil is capable of supplying at the soil surface over the time interval. If soil water is not limiting then ET is set equal to the PET. ET is set equal to zero for any hour in which rainfall occurs.

Prior to this project the PET was calculated from daily maximum and minimum temperatures by the methods of Thornthwaite (1948) and Thornthwaite and Mather (1957). This is an empirical method which has proved to be surprisingly accurate over the growing season (Mohammad, 1978) and was selected for DRAINMOD because of the small and available (in HISARS) input data requirement. However, this project deviates from the original function of DRAINMOD in that we are interested in the system hydrology for the whole 12 months rather than the ground preparation and growing season alone. It became clear that the Thornthwaite method severely underpredicted PET for the winter months. Whenever the average of the maximum and minimum temperature dropped close to freezing the predicted PET became zero. In an area with cold nights and warm days this is clearly in error.

Two methods were considered to remedy this. The first was to use a more accurate prediction of PET using a wider base of atmospheric data. The Christianson R_s method was selected as being the most suitable following comparisons with Thornthwaite, Penman, Jenson-Haize and pan methods. This method requires daily maximum and minimum temperatures, daily average relative humidity, average daily windspeed and incoming solar radiation in Langleys/day (which can be determined from daily percent sunshine and diurnal solar radiation at the top of the atmosphere). These data were available for First Colony Farms for the 18 month period from 6/1/77 to 12/31/78. The daily Christianson PET values for this period were calculated and read into DRAINMOD directly.

The second remedy to the underprediction problem of Thornthwaite was to apply a correction factor to the Thornthwaite values. The long term average monthly PET values for Chapel Hill, NC were available from the NOAA Evaporation Atlas for the 48 Contiguous United States (1982). The long term average monthly Thornthwaite values were calculated using temperature data for Raleigh, NC and Elizabeth City, NC. The monthly correction factors were found by dividing the NOAA values by the Thornthwaite values. There was no significant difference between the correction factors using Raleigh data and from using Elizabeth City data so applying the derived correction factors to simulations for FCF would appear reasonable.

The final solution was thus to use the Christianson PET for the model testing and to use temperature data from Elizabeth City with the correction factors for the long term model simulations and predictions. Although it is believed that the correction factors will give an accurate account of PET over a long period of time they cannot be expected to give accurate predictions for an individual day. In model testing, when it is necessary to know precisely what the water balance was for a given day (sometimes for a given hour), it was necessary to use data for that particular day. While use of the correction factors may cause ET predictions to be over or under estimated for a given day, the monthly totals should be correct. This means that the monthly water balances should be correct and antecedent conditions for storms closely approximated.

<u>Hourly Flow Rates:</u> DRAINMOD determines the water lost from the soil profile by subsurface and surface drainage on an hourly, two hourly and daily basis, depending on rainfall and drainage rates, and prints the results as a daily water loss in cm. In order to use the DRAINMOD output as lateral flow input to the flood routing model it was necessary to convert this unit area water loss to a flow rate. In the unmined condition, FCF consists of a large number of 1600*800 m^2 (1*0.5 mi^2) blocks. Each has 16 parallel field ditches spaced 100 m apart which empty into a deep 1600 m long collector canal (Figure 21). For the purpose of the

analyses each block was divided into 16 fields, each field being 800 m long and 100 m wide with a field ditch running through the middle. All water loss values from DRAINMOD were converted into hourly depths and then into volumes by multiplying by 100 * 800 m^2 and finally into flow rates in liters/sec for output into a DISK file as a string of hourly values for later access by the flood routing model.

The calculation of the soil water distribution is as given in Skaggs (1978) and was left unchanged for this application.

Flood Routing Model

The permitted mining area on First Colony Farms (FCF) is highly channelized, which lends itself handily to flow rate determination by flood routing. By routing the water lost from the individual fields down through the network of canals, continually accounting for new input along the way, an accurate assessment can be made of how flow rate (and stage, if desired) varies with time throughout the network and, most importantly, at the outfall of a watershed. Because the FCF area is very flat (surface slopes < 0.02 %) and because of the intensive canal network, each watershed is clearly defined.

The basic flood routing model selected and used in this study was described by Amein (1968). The model is based on a numerical solution of the equations of unsteady flow in open channels. These are the equation of continuity;

$$\frac{\partial A}{\partial t} + \frac{\partial Q}{\partial x} - q = 0 \qquad (6)$$

and the equation for the conservation of momentum;

$$\frac{1}{A}\frac{\partial Q}{\partial t} + \frac{1}{A^2}\left[\frac{\partial}{\partial x}\left(\frac{Q^2}{2}\right)\right] + \frac{\partial}{\partial x}\left[\frac{Q^2}{2A^2}\right] = g(S_o - S_f) - g\frac{\partial y}{\partial x} \qquad (7)$$

where: Q is the flow rate,
A is the cross sectional area of flow,
q is the lateral inflow,
S_o is the slope of the channel bottom,
S_f is the friction slope,

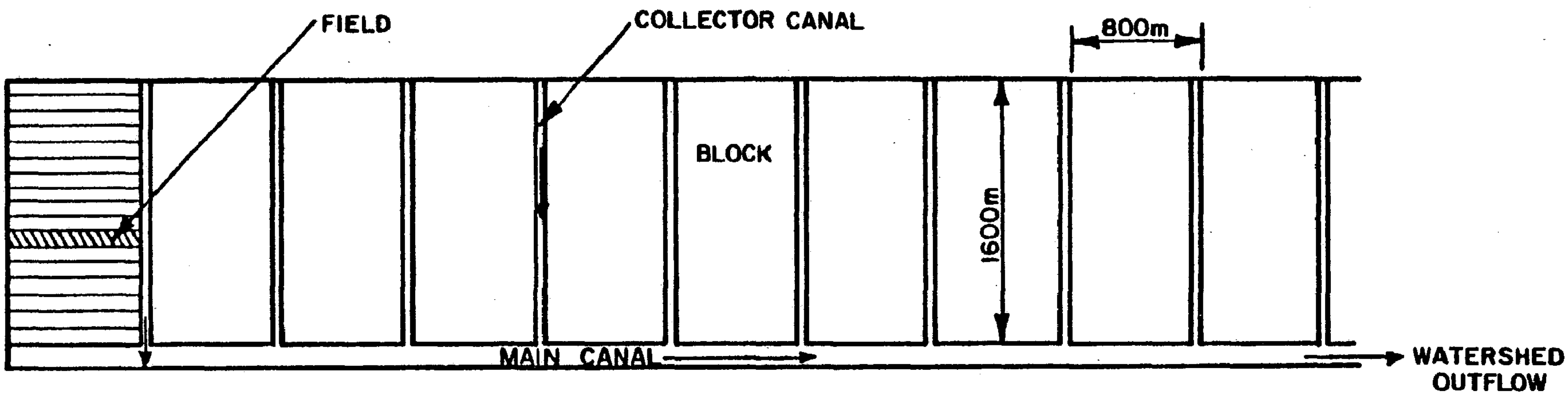

Figure 21. Layout of a typical stretch of main canal on First Colony Farm's permitted mining area.

y is the water depth,
g is the acceleration due to gravity,
x is the distance along the channel, and
t is time.

The above equations are written in an implicit finite difference form using a centered difference scheme for distance and a forward difference scheme for time. These equations (known as the Saint-Venant Equations) are based on the following assumptions; a) flow is one-dimensional,

b) pressure is assumed to be hydrostatic (i.e. vertical acceleration is neglected),

c) water is incompressible and homogeneous in density,

d) the effects of boundary friction for steady uniform flow are considered applicable, and an empirical equation (e.g. Manning's Equation) can be used to describe the friction effects, and

e) the beds of the channels are fixed (i.e. no scour or deposition is assumed to occur).

The finite difference equations are non-linear and are solved by the generalization of Newton's iteration method. This procedure requires the solution of a very large system of simultaneous equations. However, the coefficient matrix is very sparce and banded about the major diagonal with a maximum of four non-zero elements per row. By taking advantage of this knowledge, a rapidly convergent method for obtaining the solution was developed (Amein, 1972).

This model was selected primarily for its proven ability to handle rapidly varying lateral inflows. The flood routing model has been tested for a wide range of applications in natural and artificial channels, reservoirs and estuaries (Amein and Fang, 1969, Amein, 1972, and Wardak, 1977). Observed and computed flows have been in excellent agreement for all cases.

Several modifications were made to the basic flood routing program described above for use in this project. Much of the program's generality was removed to reduce the data input requirements and to reduce computational time. Despite the efficiency of solution it was still an expensive model to execute in terms of computer time.

Channel Description

Because of the regular man-made nature of the canals on FCF, average cross-sectional areas were used for the collector and main canals instead of specifying a different cross-section at every station. Also the stations along the canals were spaced at regular intervals corresponding to the field ditches on the collector canals and the collector canal outfalls on the main canals.

Overbank Flow

There was no provision in Amein's original model to deal with flooding. The channel depth was allowed to incease indefinitely during storms and remove all water resulting in very high and unrealistic flow rates. This was overcome by specifying a maximum channel depth and switching to a composite channel section once the water level had breached the banks (see Figure 22). The frictional coefficient for the overbanks was set much higher than for the channel to reflect actual conditions. This change is an approximation only. The overbank width was set equal to half the distance to the next canal but assumed that the water continued to flow in the same direction as the water in the canal. In reality, once the canal levels began to drop, the flow direction would be back towards the canal. However this could not be represented in the model. While this modification is a definite improvement over the original version, it is still an approximation to what is really a two-dimensional problem.

Convergence Criteria

In this project the flood routing model was subjected to extremely varied lateral inflow. At the instant that surface runoff is initiated the flow rate from the field to the collector canal over an hour could easily increase by a factor of 100. Occasionally such rapid changes, especially if the canal banks are breached, would result in non-convergence within the model's solution procedure. Convergence was determined by the flow rates and depths in subsequent iterations being within a certain range of each other. In order to keep the low flows accurate this range was initially made very small, resulting in excessive iterations for high flows. A convergence criteria which was a function of the predicted flow rate or depth was installed with much improved results.

Boundary Conditions

The original purpose of the model was to route flow from its source to its outlet through a channel reach. The lateral inflow component was primarily used to cope with seepage losses from the channel (hence the negative sign in equation (6)). In the application for this project all of the channel flow originates as lateral inflow. The channels all dead end at their farthest upstream points. For solution, at least one dependent variable (Q or y) must be known at the upstream and downstream boundaries at all times. The model does not allow for zero upstream discharge. To overcome this, the farthest upstream lateral inflow station was taken as the upstream discharge. Ordinarily, any lateral inflow is considered to occur at right angles to the direction of channel flow and thus not contribute to the momentum equation (no q term in equation (7)). By altering the upstream

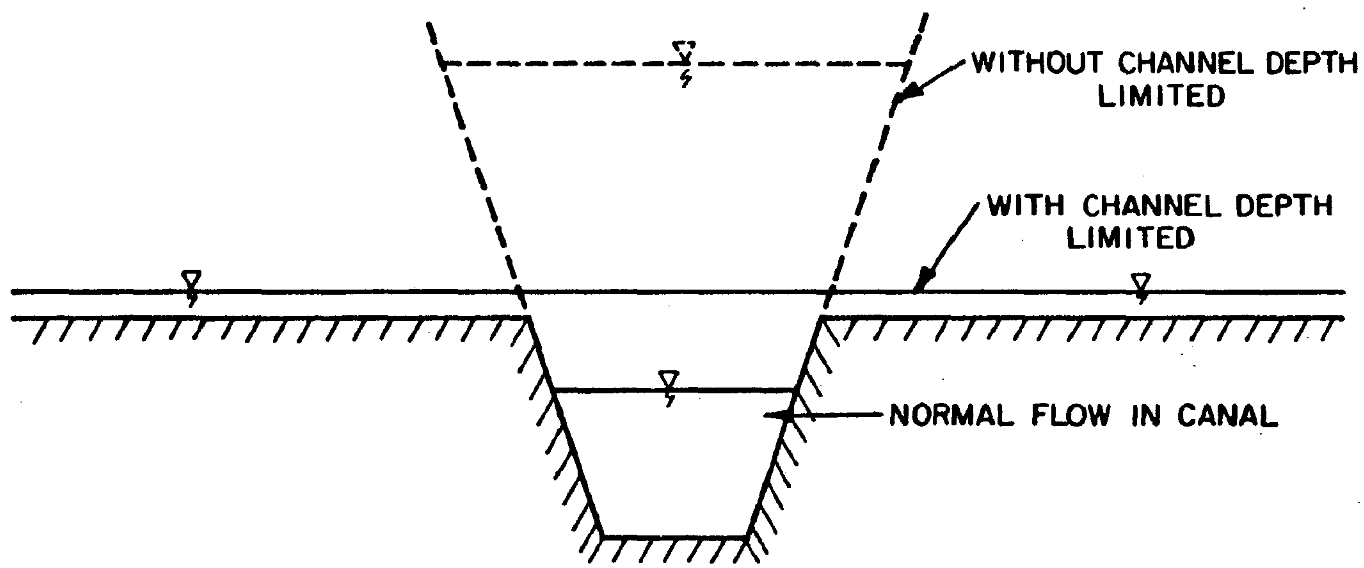

Figure 22. Effect of limiting channel depth.

inflows in this manner an error is introduced, but it was found to be negligible in trial simulations.

The downstream boundary condition is fixed by assuming normal flow and relating the flow (Q) to the depth (y) by Manning's equation;

$$Q = \frac{C}{n} R^{2/3} A S_f^{1/2} \tag{8}$$

where R is the hydraulic radius, n is the Manning's roughness coefficient, C is a constant depending on the units used and other terms are as previously described. Thus for any y, Q can be calculated and the downstream boundary condition is met.

MODEL TESTING

Introduction

Rainfall, water table depth and runoff measurements were made on FCF and adjacent areas for approximately three years from 1976 to 1979. This data formed part of the basis for a report entitled 'Effect of Agricultural Land Development on Drainage Waters in the North Carolina Tidewater Region' by Skaggs et al. (1980). The model testing presented here concentrates on one field site, Site 103, in the above report (see Figure 20). The site is centrally located in the permitted area and is a typical example of the area.

The model testing consisted of three phases. The DRAINMOD water table depth predictions were compared with the observed for the period of historical record for Site 103. The monthly total outflow volumes per unit area were compared to the observed and finally the collector canal outflows as determined from the DRAINMOD - flood routing model combination were compared to the observed on a storm by storm basis.

Input Data

A summary of soil property and other input parameters for DRAINMOD are given in Table 16. The hydraulic conductivities for the soil layers other than the surface layer were taken from studies by Badr (1978), Purisinsit (1982) and Foutz (1983). The surface layer has a relatively high measured conductivity (about 15 cm/hr). This is attributed to the irreversible drying and curing of the surface peat layer as well as the porous root mat close to the surface. However, field observations and water table depth measurements indicate a much higher surface hydraulic conductivity than measured. Many of the sites on the farm, which were developed for agricultural use, had their surfaces land formed and sloped towards the field ditches. Thus the gradient when water is in the surface layer would be higher than expected from assuming a level surface. Also, the higher conductivity layer overlays an extremely low conductivity (0.02 cm/hr), high water content peat. This would presume an abrupt change in infiltration capacity of the layers and result in water infiltrating the surface layer and then moving horizontally with the slope towards the field ditch after encountering the second layer. Thus a high (70 cm/hr) conductivity was used for the surface layer to compensate for these phenomena.

The ditch dimensions used were as surveyed when the original data collection was made. Fire over part of the permitted area in 1981 altered the shape and depth of the channels as a result of

Table 16. Summary of soil property, drainage system and crop parameter inputs to DRAINMOD for organic soil (site 103) prior to mining.

INPUT	VALUE
1. Soil properties.	
Depth to restricting layer	234 cm
Saturated hydraulic conductivity, K	K = 70 cm/hr for depth < 25 cm K = 0.02 cm/hr 25 < depth < 140 cm K = 0.44 cm/hr 140 < depth < 234 cm
Water content at lower limit available to plants	0.45 cm^3/cm^3
Saturated water content in root zone	0.74 cm^3/cm^3
Green-Ampt infiltration parameters;	

Water table depth (cm)	A (cm^2/hr)	B (cm/hr)
0	0	0
25	0.85	4.12
50	1.64	2.63
75	1.74	2.00
100	1.40	1.60
150	1.80	1.20
500	0.50	1.00

INPUT	VALUE
2. Drainage system parameters (open ditches).	
Drain depth	140 cm
Effective drain radius	19 cm
Surface depressional storage	1.25 cm
Drain spacing	100 m
3. Crop parameters (grass)	
Effective root depth	12 cm
Length of growing season	365 days

burning away of the banks and ash and sediment accumulation in the ditches. Collector canal specifications used as input to the flood routing model are given in Table 17.

Hourly rainfall data used in this section were collected on Site 103 for the period of concern. It was stored in the HISARS form and was available as direct input to DRAINMOD.

Evapotranspiration data were calculated daily by the Christianson R_s method and read into DRAINMOD in card image form for the period available. The corrected Thornthwaite method was used for other winter months and the uncorrected Thornthwaite method for other summer months.

Other soils data required by DRAINMOD were taken from Purisinsit (1982) and are summarized in Table 16. The upward flux - water table depth relationship is given in Figure 23 and the volume drained - water table depth relationship is given in Figure 24 (before mining curve).

Table 17. Collector canal specifications for input to flood routing model.

INPUT	VALUE
Channel dimensions	
Depth	230 cm
Bottom width	76 cm
Side slope (horizontal:vertical)	1:1
Channel slope	0.02%
Manning's roughness coefficients	
Channel	0.20
Overbanks	1.50

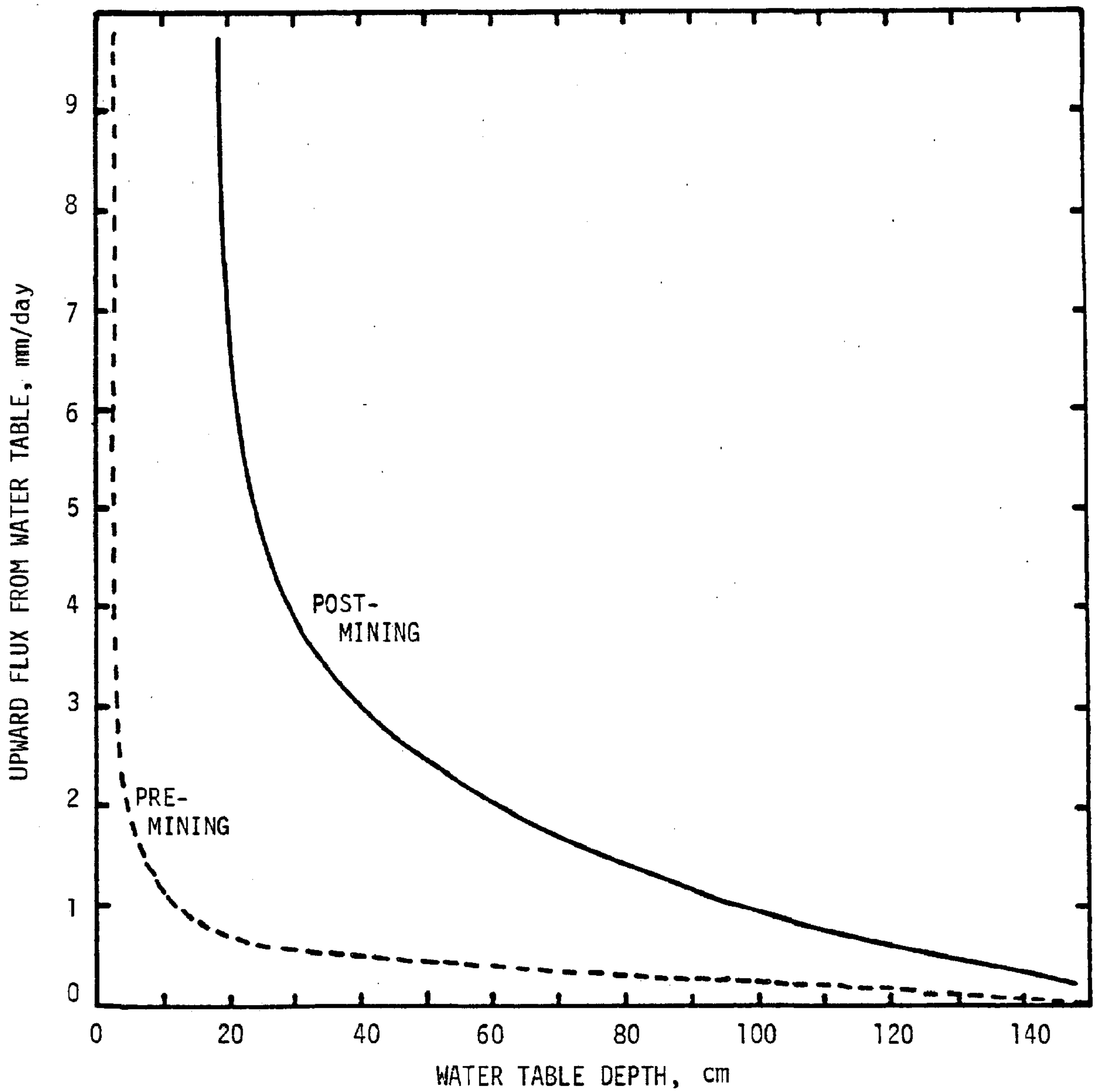

Figure 23. Steady upward flux as a function of the water table depth for the pre- and post-mining soils.

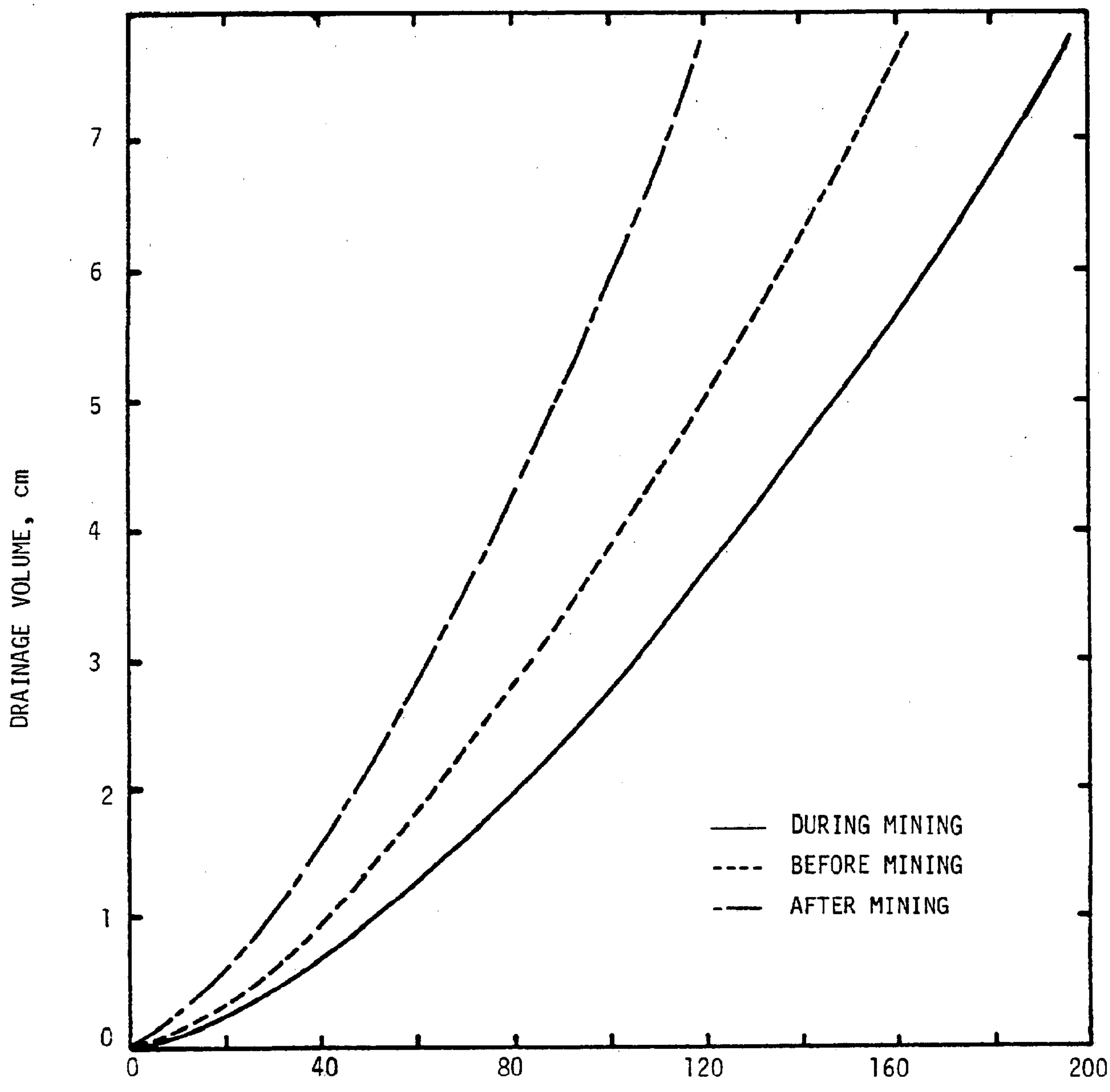

Figure 24. Drainage volume as a function of the water table depth for the before, during and after mining soils.

Results and Discussion

Water Table Depth Comparisons

The water table depth comparisons for the three years of observed data are given in Figures 25, 26 & 27. Only the simulated periods for which unbiased, independently obtained evaporation data were used are shown. These are periods for which Christianson PET were available and for summer months when Thornthwaite was considered reliable (a total of 25 months between May, 1977 and Sept., 1979). For the other periods, ET factors were selected to "fit" the simulated to the observed water table depths. There is good agreement between the observed and simulated water table depth measurements. An average daily deviation between observed and simulated water table depths of only 10.2 cm was found for the periods considered.

Discrepancies do occur close to the surface. The observed water table depths rarely rise completely to the surface. This is believed to be a combination of errors in the well calibration and of the surface conductivity effects discussed earlier. The water table at the mid-point is essentially controlled by ET and infiltration and any other differences between observed and simulated water table depths are primarily a result of differences in the actual and predicted values of these parameters. These differences, however, appear minor as evidenced by the high degree of correlation exhibited in Figures 25, 26 & 27 and the low average deviation.

Monthly Flow Volumes

The observed and simulated monthly unit-area flow volumes are given in Table 18. The observed flow data were collected using a weir and continously recording chart. The measured flows were stored as four hourly weir depths. The weir was subjected to repeated periods of submergence. This occurs when the water level downstream of the weir rises to above the bottom of the 'v' in the weir. In this condition high stages over the weir may be recorded but the weir calibration becomes invalid and flows in excess of reality are recorded. These periods were adjusted in a previous study (Purisinsit, 1982) so that approximately the correct volume of water was recorded for a particular storm. Actual hydrograph shape and magnitude during the flow peak, however, could not be recreated. For the analysis, only the periods 5/77 to 12/78 and 5/79 to 9/79 were considered because of the unavailable data for calculating ET during other periods as discussed for the water table depth comparisons.

On average the observed flows are greater than the simulated flows by 0.35 cm/month or 13.6%. This could be partly due to

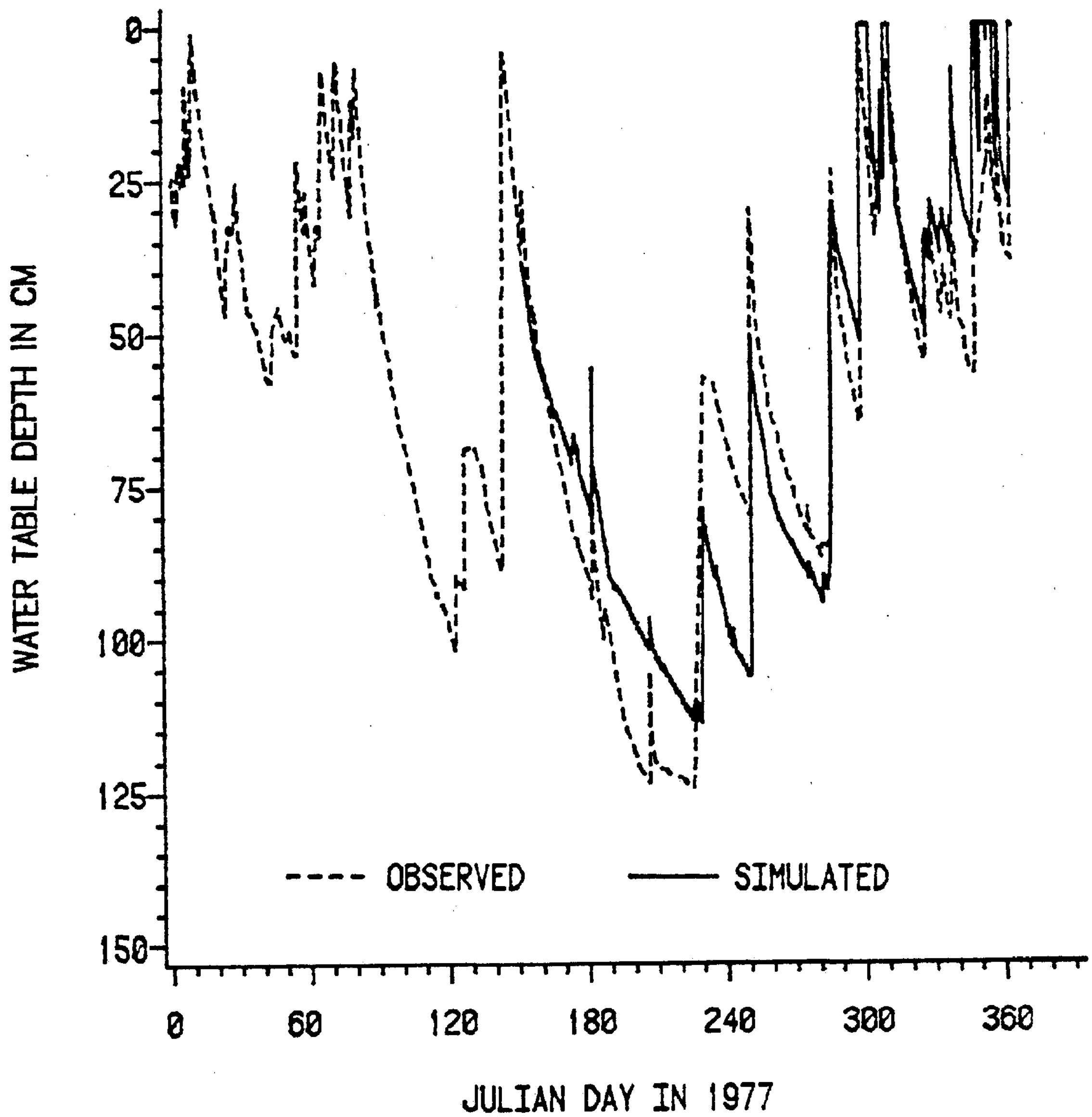

Figure 25. Observed and simulated water table depth comparison from site 103 for 1977.

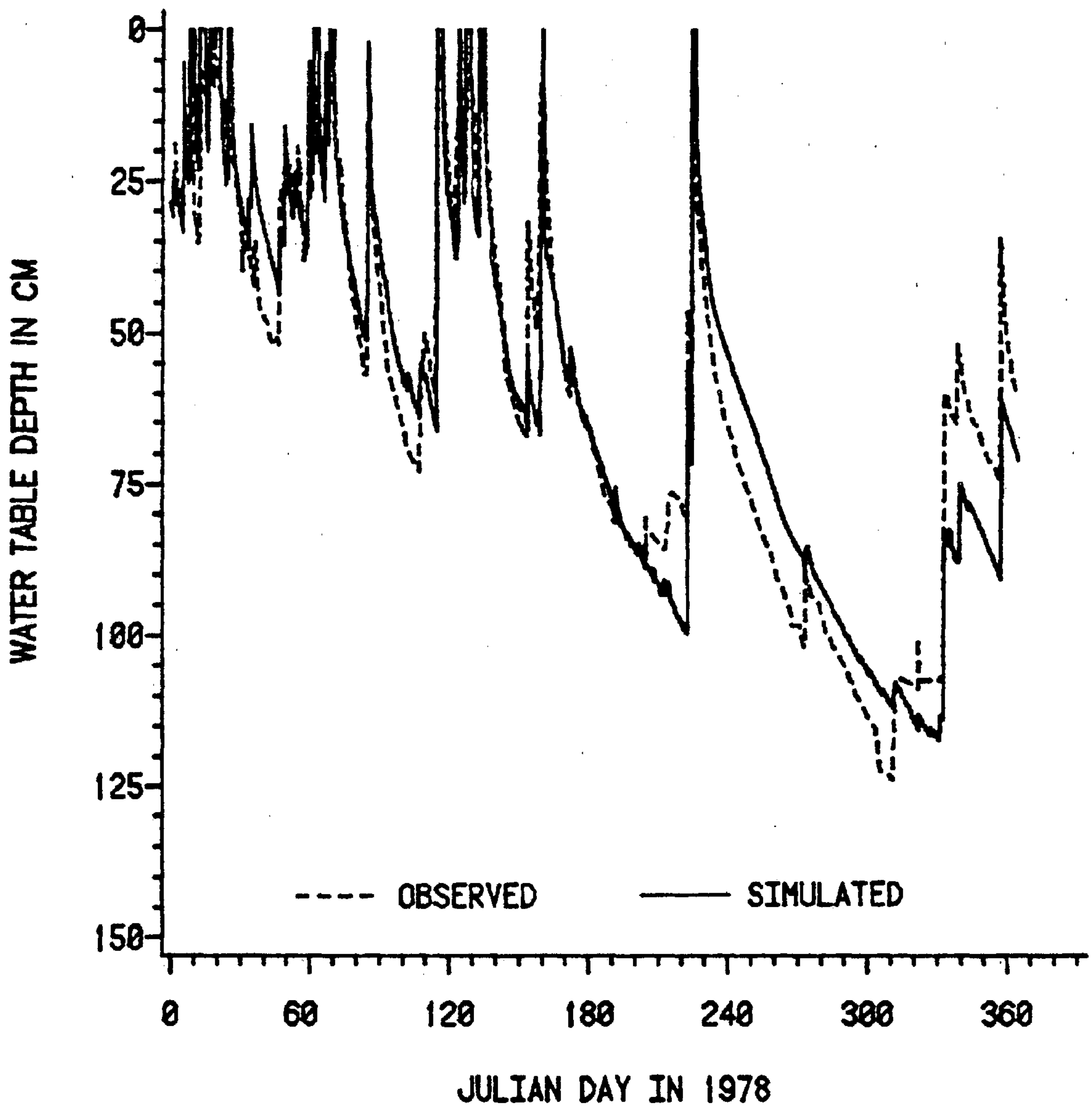

Figure 26. Observed and simulated water table depth comparison from site 103 for 1978.

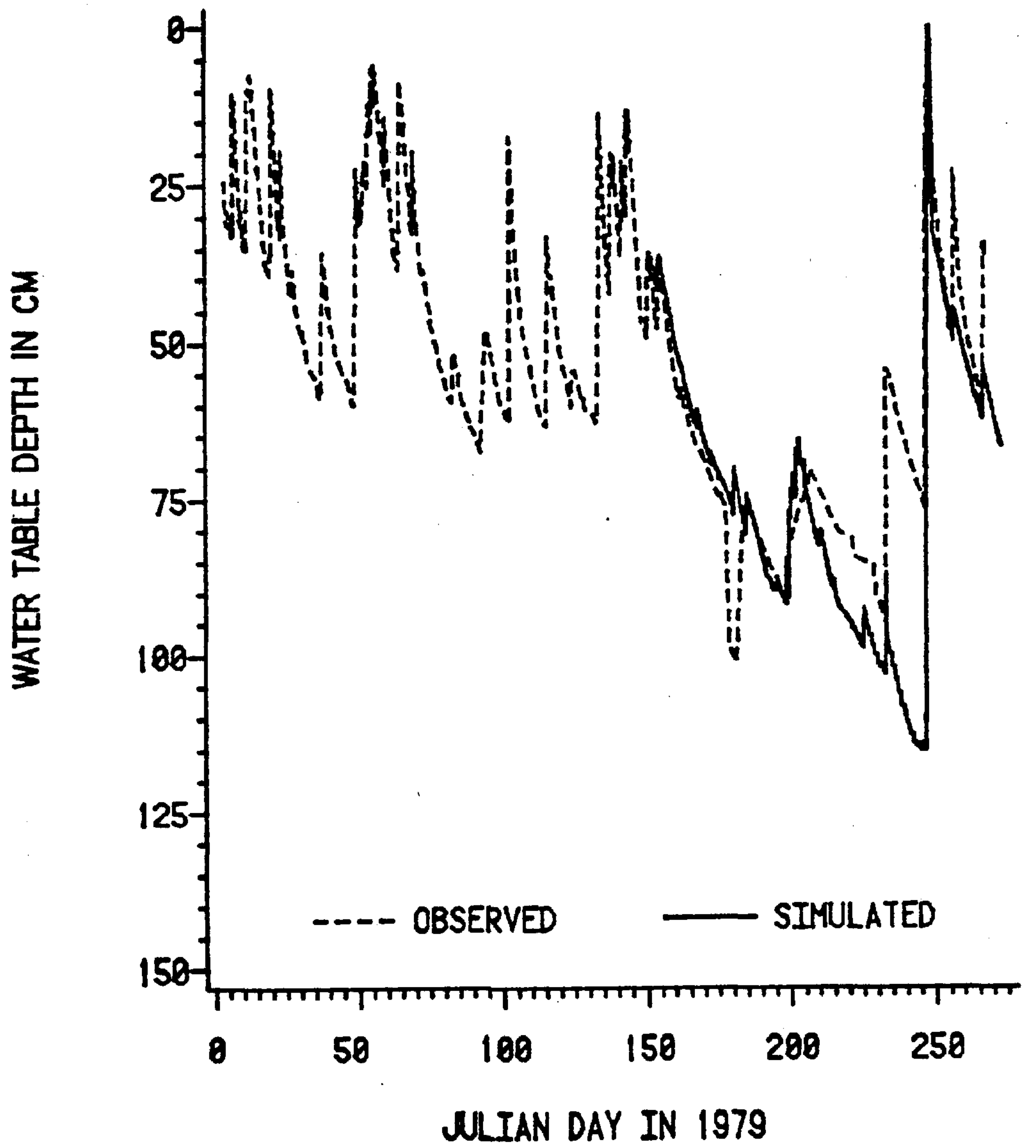

Figure 27. Observed and simulated water table depth comparison from site 103 for 1979.

Table 18. Monthly totals of observed and simulated flow volumes from site 103.

	1977		1978		1979	
Month	Obs.	Sim.	Obs.	Sim.	Obs.	Sim.
1	7.16	7.31*	10.15	12.68	10.54	9.15*
2	0.84	0.83*	2.6	1.05	5.57	5.63*
3	2.57	2.59*	4.5	3.40	3.72	2.85*
4	0.46	0.43*	5.0	3.70	2.68	2.77*
5	6.73	6.61	5.8	4.21	3.17	5.42
6	1.97	0.53	1.7	0.84	1.72	0.61
7	0.23	0.48	0.43	0.29	1.24	0.48
8	0.15	0.39	3.6	2.20	1.36	1.79
9	1.38	0.58	0.24	0.23	6.54	3.31
10	4.95	6.31	0.20	0.14		
11	12.04**	7.16	0.36	0.22		
12	3.90	6.09	1.90	0.31		

* values obtained using ET correction factors to fit the observed and simulated water table depths. These results were not used in the comparative analysis.

** an obvious data error - only 10 cm of rainfall was recorded for this month. 11/77 is excluded from further analysis.

inaccurate corrections for the periods of submergence. The observed and simulated monthly totals for the validation period are plotted in Figure 28 to show the range of scatter about the line of equal value. A Chi-Squared test performed on these values results in a 95% probability that the simulated data is from the same distribution as the observed data. On the average the monthly totals are in good agreement considering the number of potential problems. If the periods when ET values corrected to fit the curves are included, then the monthly total agreement is considerably improved.

Hydrograph Comparison

The hourly water loss predicted by DRAINMOD was routed down the collector canal to the main canal for a number of storms during the period of observed record. The results were encouraging but clearly indicated areas for future work. The flows in liters/sec at the collector canal outlet are compared with observed flows in Figures 29, 30, 31, & 32.

Figure 29 illustrates one of the best agreements obtained. The slight displacement would appear to be caused by chart timing error as the initial observed flow rate increase begins several hours before rainfall is recorded. In this comparison the observed and simulated water table depths match very closely, rising to just below the soil surface at each peak. No surface runoff is simulated and most likely none occurred in reality. Thus with water tables being accurately predicted the DRAINMOD - flood routing combination appears to do a very good job in predicting collector canal outflows when the water table is between the surface and about 40 cm below the surface. This would indicate that the bank storage flow and surface layer conductivity modifications are justified and, at least cumulatively, accurate.

The models tend to over-predict the flow rate once surface runoff occurs. This is illustrated in Figure 30. Unfortunately the flood routing model assumes an unlimited outlet for all water. If for some reason the downstream outlet is restricted, water backs up the canals and reduces the outflow rate. While the flood routing model has the capability to model such occurrences, there was no way of knowing exactly when they occurred for the experimental site. It is suspected that the channel banks were actually breached more often than predicted because of water backup. Water backup would also cause weir submergence and thus recorded data error as well.

The simulation models accurately account for all water movement within the system. Not only does the unit area, soil water budget balance, but the water balance for the whole block is maintained. During peak flows with overland runoff occurring it is likely that water is being lost directly from the fields into the main canal, and possibly in other directions, bypassing

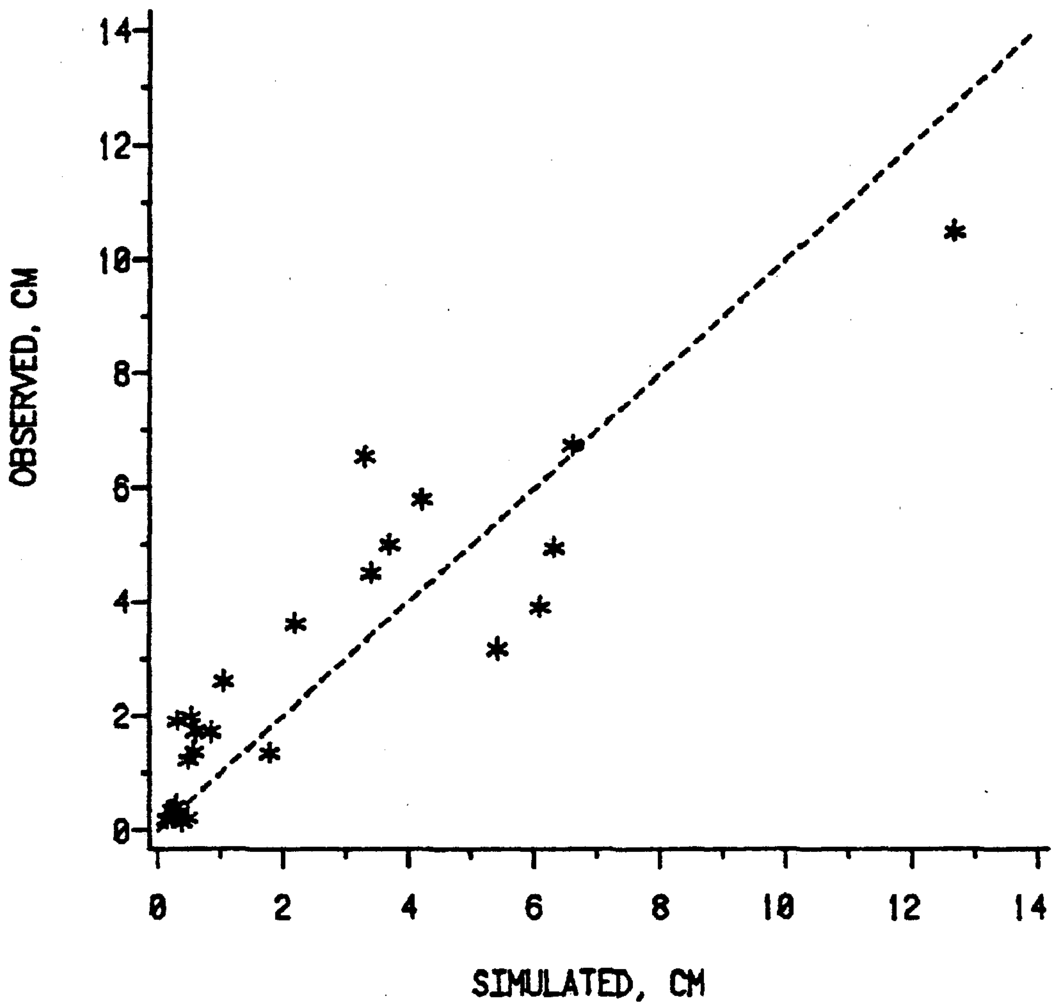

Figure 28. Observed versus simulated monthly flow volumes per unit area for 24 months.

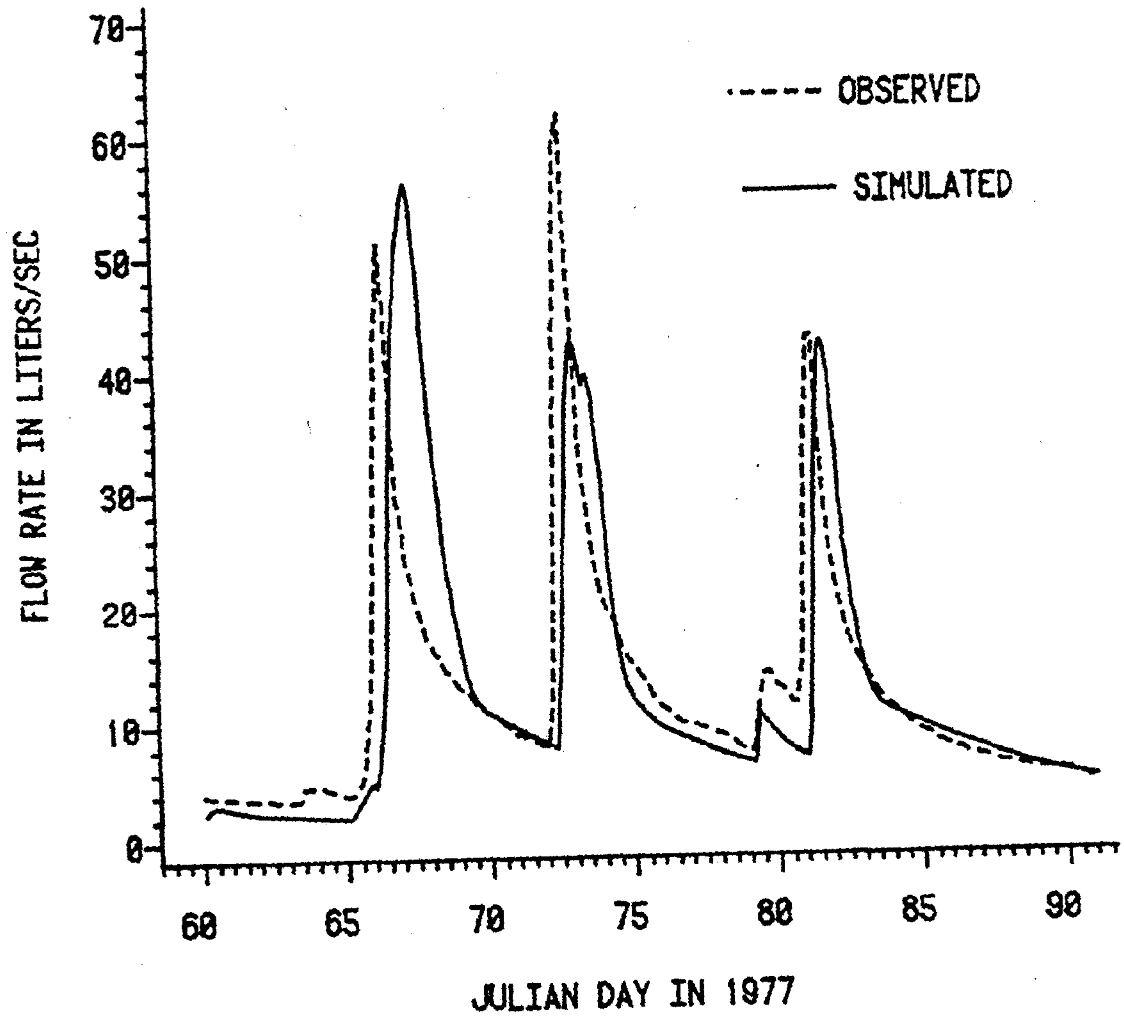

Figure 29. Observed and simulated hydrograph comparison from site 103 for model validation.

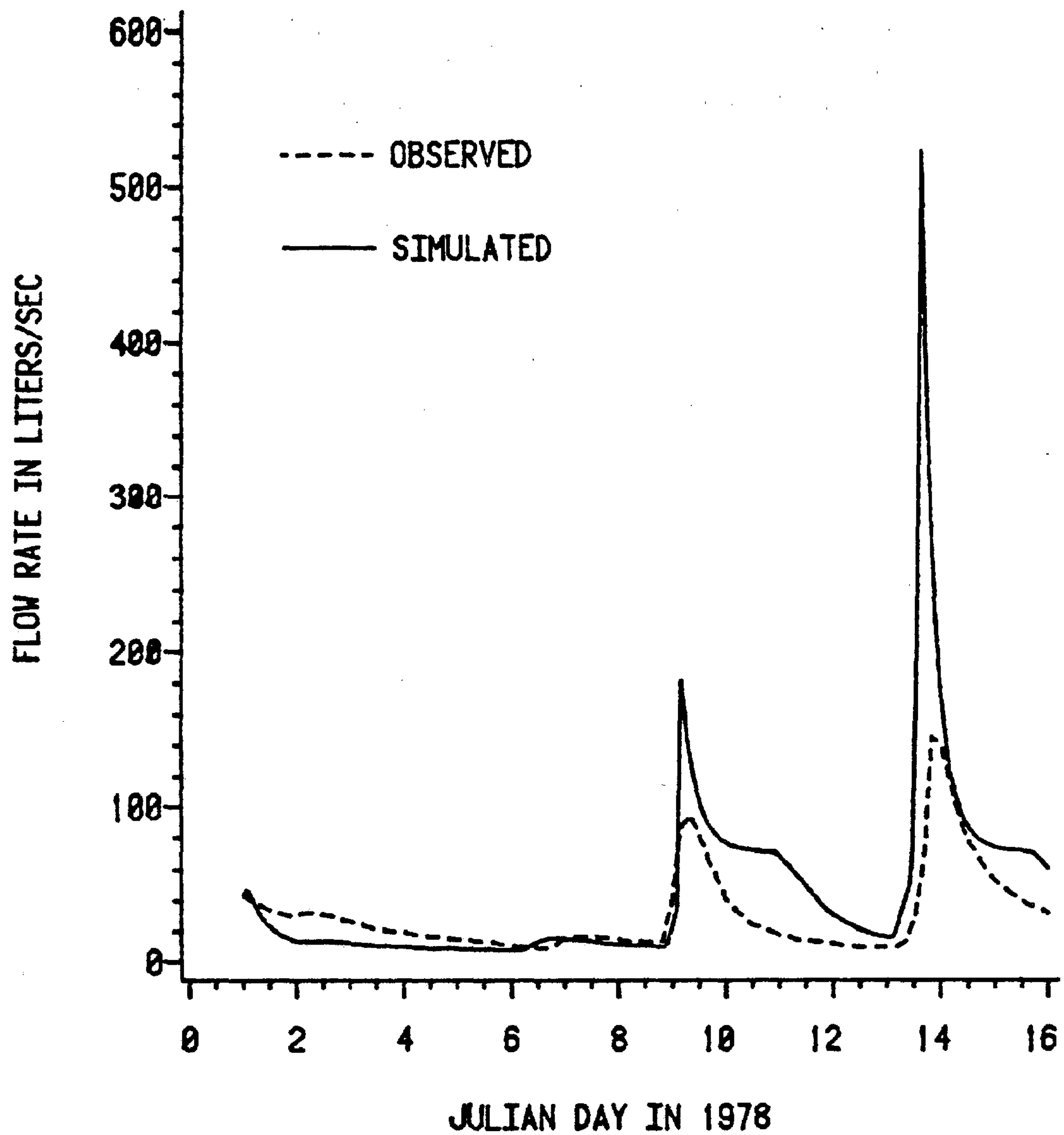

Figure 30. Observed and simulated hydrograph comparison from site 103 for model validation.

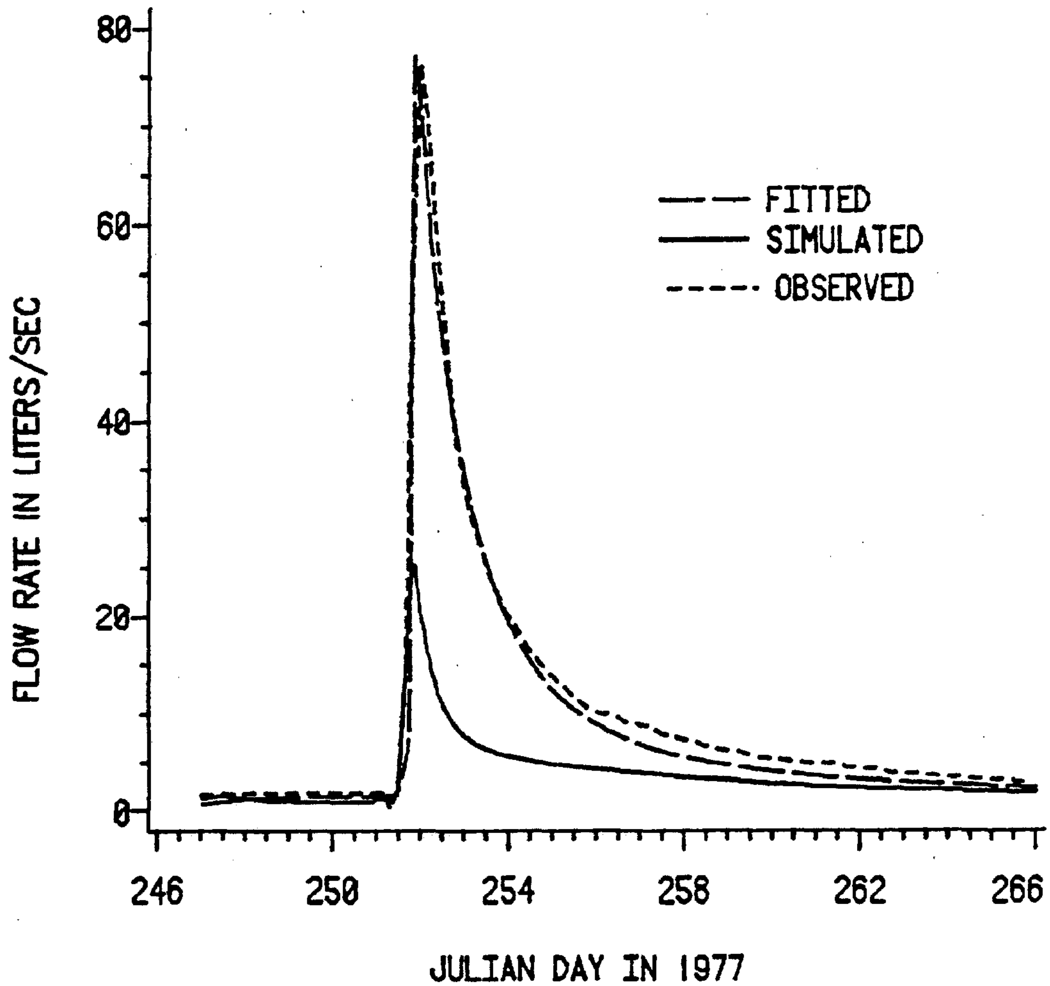

Figure 31. Observed and simulated hydrograph comparison from site 103 for model validation. Also shown is the simulated hydrograph forced to fit the observed by using model calibration.

the collector canal weir. Thus the rigorous accounting procedure of the models would be expected to over-predict the observed flows under these conditions.

When the water table is deep (>50 cm below the surface) and a moderate rainfall occurs, the observed runoff would often exceed the simulated by a significant amount. Figure 31 is a example of this. The observed flow, while higher than the simulated, is unlikely to be from overland runoff because of the low rainfall intensity. It seems more likely that the rainfall infiltrated the surface layer of dry, cured peat but then only a portion infiltrated the moist peat layer underneath leaving the remainder to move quickly, with the ground slope gradient, to the field ditches through the surface layer. The high conductivity surface layer has no effect within DRAINMOD unless the actual water table rises up into that layer.

The observed hydrograph in Figure 31 and other such events could be fitted by simply manipulating parameters controlling infiltration or antecedent conditions. This is demonstrated in Figure 31 by the plot labeled "fitted" which was fitted to the observed hydrograph. That is, the model parameters and antecedent conditions were adjusted so that the predicted hydrograph was in close agreement with the observed. Hydrologic models of a less deterministic nature than the one used here are often "calibrated" in this manner. However, unless long periods (years) of data are available for such calibration, results such as given in Figure 31 may be extremely misleading. Antecedent conditions used in the calibration of a storm of a certain magnitude might rarely be duplicated when used for projection. Also, the fitted hydrographs usually make the model appear more precise than it really is. Altering parameters to fit the hydrographs, while fixing one simulation, would adversely affect others and the results could be misleading. When calibration is used, it is extremely important that predictions of the "calibrated model" be compared with observed data which have not been "fitted" to test the validity of the result. In our case, model parameters were determined independently and calibration was not used.

Results indicate that changes in surface properties affecting infiltration capacity of the peat occurs with wetting and drying in a manner that is not completely described by methods presently used in the model. More work is needed in this area if complete understanding and modeling of peatland hydrology is to be accomplished.

Figure 32 illustrates a low flow condition. The models (0.018 cfs). The hydrograph peaks in Figure 32 are a result of rainfall falling directly into the ditches and canals. The percentage of channel surface area for a block was found to be approximately 2% of the total block area. Under most situations the water falling directly into a channel will find its way to the weir at the collector canal outlet. Thus 2% of the rain falling on a field was considered to be direct runoff in the

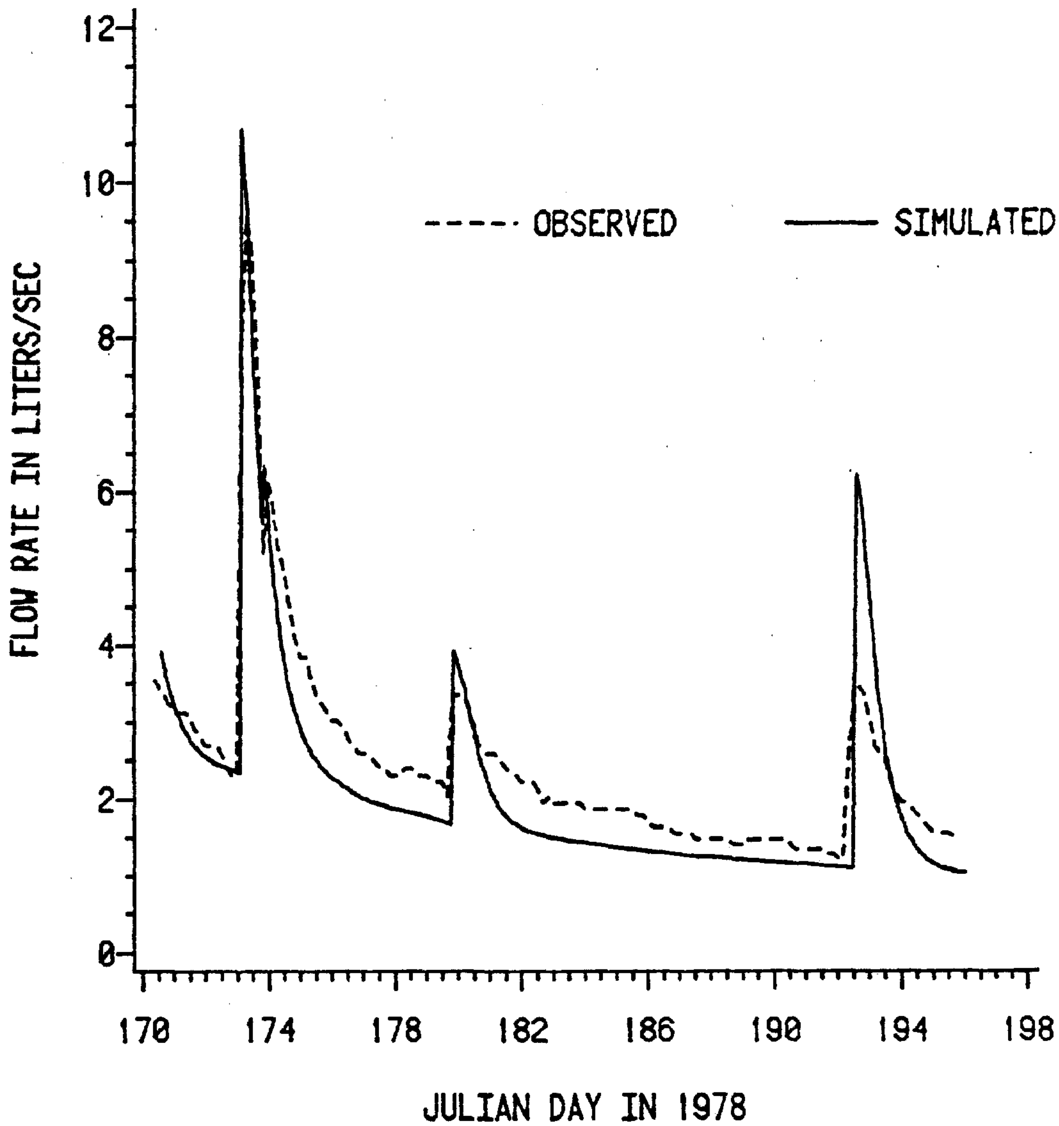

Figure 32. Observed and simulated hydrograph comparison from site 103 for model validation.

models. Because of the very low drainage rates from the peat soils this 2% rainfall is very significant during low flow situations as evidenced by Figure 32. The 2% figure calculated appears to be realistic as very good observed and simulated comparisons are produced for these events.

For the period 1/1/77 to 9/31/79, nineteen periods containing rainfall events were simulated to produce collector canal outflow. The average length of these periods was 28 days. This was a little over half the total period. Data collected for rainfall events during the remainder of the period were not used because of weir submergence, missing periods, incorrect reading of time scale, or simply incorrect flow readings. There were also periods of no rainfall or runoff which were excluded from the analysis. The nineteen periods simulated included all periods containing rainfall events which had corresponding runoff data. As well as looking at individual hydrograph comparisons, the flow rates for these periods were collected together and compared on a flow duration curve (Figure 33). This curve shows the probability of occurrence of a flow of any given magnitude. The simulated and observed flow duration curves are in good agreement illustrating the models capability to accurately represent the general hydrology of the site.

Conclusions

While problems still exist, on the average the models are able to give a realistic picture of the hydrology of the First Colony Farms peat mining area. Although the models do not always accurately predict the response to individual rainfall events the predicted water table depths are in good agreement, the monthly flow volumes are in close agreement as are the magnitudes and distribution of the collector canal outflows (Figure 33). Thus the solutions produced are considered to be representative of the area and can be confidently used for predictive and comparative simulations of present and future conditions.

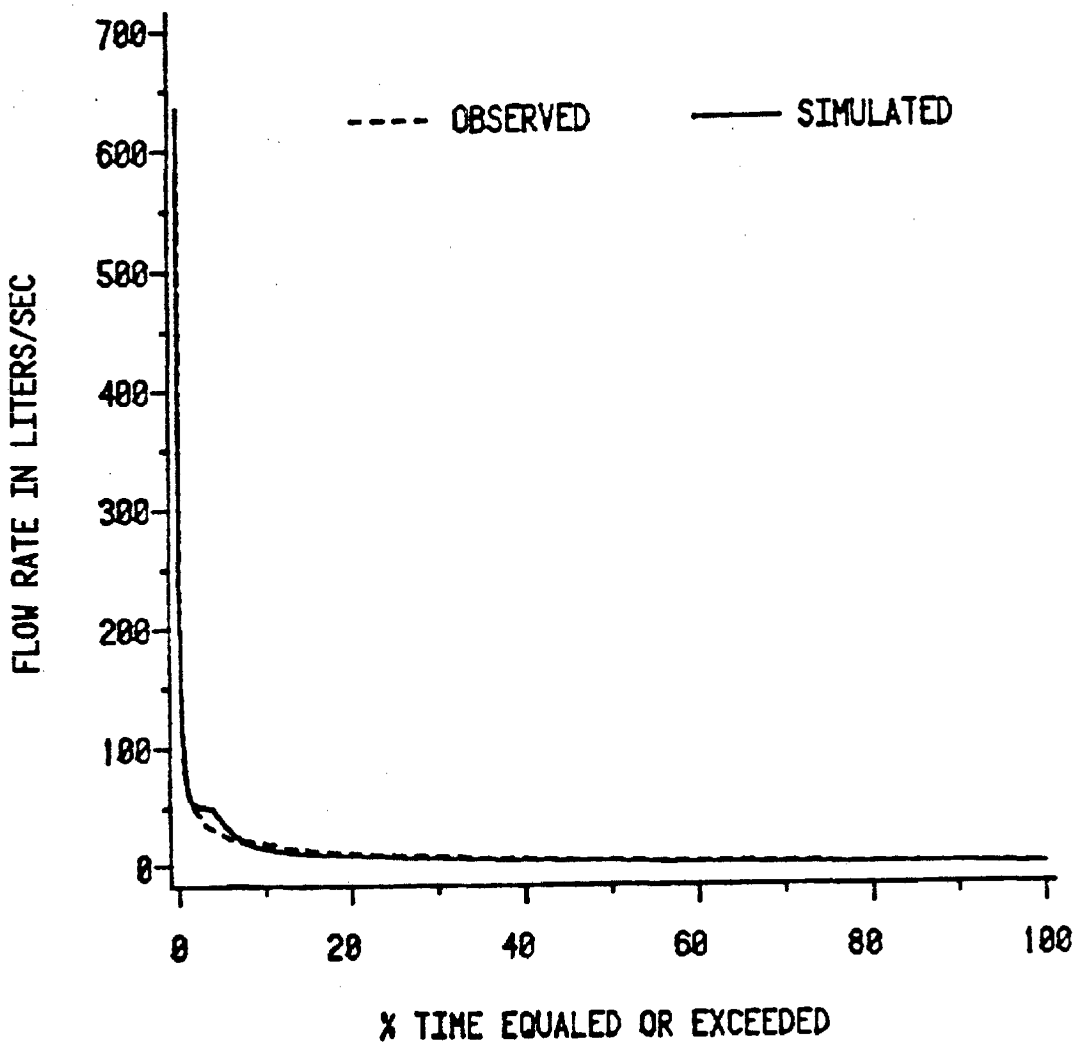

Figure 33. Observed and simulated flow duration curves of collector canal outflows from site 103 for model validation.

ANALYSIS OF HYDROLOGIC EFFECTS OF PEAT MINING

Comparison of Pre-Mining and Post-Mining Hydrology

The following simulations assume that the First Colony Farms (FCF) lands, once mined, will be drained and used for agricultural production. Corn-winter wheat rotation was used for this analysis. The effects of different crop rotations is studied in a following section. This section compares the current hydrologic situation with the expected hydrologic situation at the completion of mining and reclamation.

Summaries of the soil profile properties used for these simulations are given for pre- and post-mining conditions in Tables 16 and 19 respectively. The post-mining properties are similar to properties determined for Site 101, a developed mineral soil site in an adjacent county (Skaggs et al., 1980, and Purisinsit, 1982). The average depth of harvestable peat over the 6,000 ha site is approximately 190 cm and it is assumed that the harvesting operation will remove as much as it can until the ash content begins to incease. Because the present land surface is greater than 5 m above sea level, there is no danger of mining peat to below sea level on the FCF permitted area. The transition layer and the underlying mineral soil will become the new surface soil profile. With cultivation, incorporation of plant material, oxidation of remaining peat, and fertilization the soil properties of the surface layers are expected to resemble those of surrounding, agriculturally developed mineral soils. Mixing and tillage of the layers immediately underlying the peat should decrease their bulk density and thus increase the hydraulic conductivity of those layers. The drainage system initially assumed was open ditches spaced 100 m apart, a customary drainage system for this region. A subsurface drainage system to increase crop yields and alter the drainage water distribution was also considered as an alternative option in the following section.

The canal system after mining was assumed to remain similar, relative to the ground level, to the before mining situation. This assumption anticipated canal deepening as the peat was removed to maintain gravity flow in the canals.

A year of approximately average rainfall was selected for a mid-point water table depth comparison. A total of 127.4 cm of rainfall was recorded in 1960. A comparison of predicted water table depths is plotted in Figure 34. On average the post-mining water table was 23.2 cm deeper than the pre-mining water table. In the post-mining case, as with the pre-mining situation, the wide spaced drainage ditches, existing for both cases, have little effect on the mid-point water table depth. The water

Table 19. Summary of soil property and drainage system and crop parameter inputs to DRAINMOD for post mining analysis.

INPUT	VALUE
1. Soil properties.	
Depth to restricting layer	160 cm
Saturated hydraulic conductivity, K	K = 15 cm/hr for depth < 25 cm K = 2.0 cm/hr 25 < depth < 70 cm K = 0.44 cm/hr 70 < depth < 160 cm
Water content at lower limit available to plants	0.31 cm^3/cm^3
Saturated water content in root zone	0.60 cm^3/cm^3
Green-Ampt infiltration parameters;	

Water table depth (cm)	A (cm^2/hr)	B (cm/hr)
0	0	0
50	1.20	1.0
100	3.30	1.0
150	6.00	1.0
200	9.20	1.0
500	25.00	1.0

INPUT	VALUE
2a. Drainage system parameters (open ditches - poor subsurface drainage)	
Drain depth	100 cm
Effective drain radius	19 cm
Surface depressional storage	1.50 cm
Drain spacing	100 m

Table 19 cont.

2b. Drainage system parameters (drain tubes - good subsurface drainage).

Drain depth	100 cm
Drain diameter	10 cm
Effective drain radius	0.51 cm
Drain coefficient (as limited by hydraulics)	2.50 cm/day
Drain spacing	20 m

3. Crop parameters (corn - winter wheat rotation)

Desired planting dates	April 1 for corn October 1 for wheat
Length of growing season	167 days (for corn)

Rooting depths

Julian Day	Root Depth (cm)
1	3.0
70	3.0
110	6.0
120	15.0
150	25.0
173	30.0
245	30.0
255	10.0
270	3.0
280	3.0
300	10.0
330	15.0
366	15.0

table rises and falls as a function of infiltration and ET. The differences observed in Figure 34 are a result of different soil properties, profile depths and crop rooting depths. Deeper water tables during the summer months for the post-mining situation may be attributed to increased ET over the growing season because of increased crop rooting depths. This is illustrated in Figure 35.

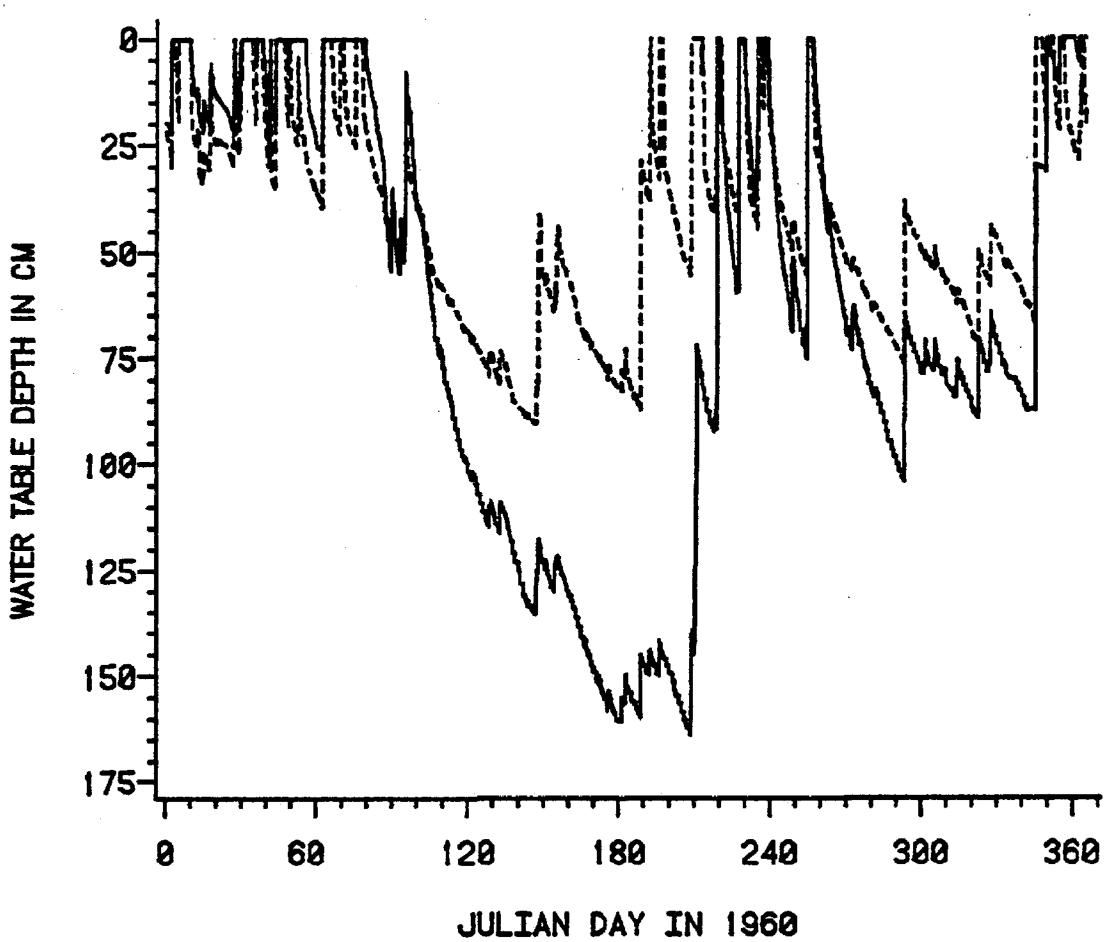

Figure 34. Before and after mining water table depth comparison for a typical First Colony Farm field for a year of average rainfall. The subsurface drainage for both cases was considered to be poor. Before mining vegetation was grass and after mining was a corn-winter wheat rotation.

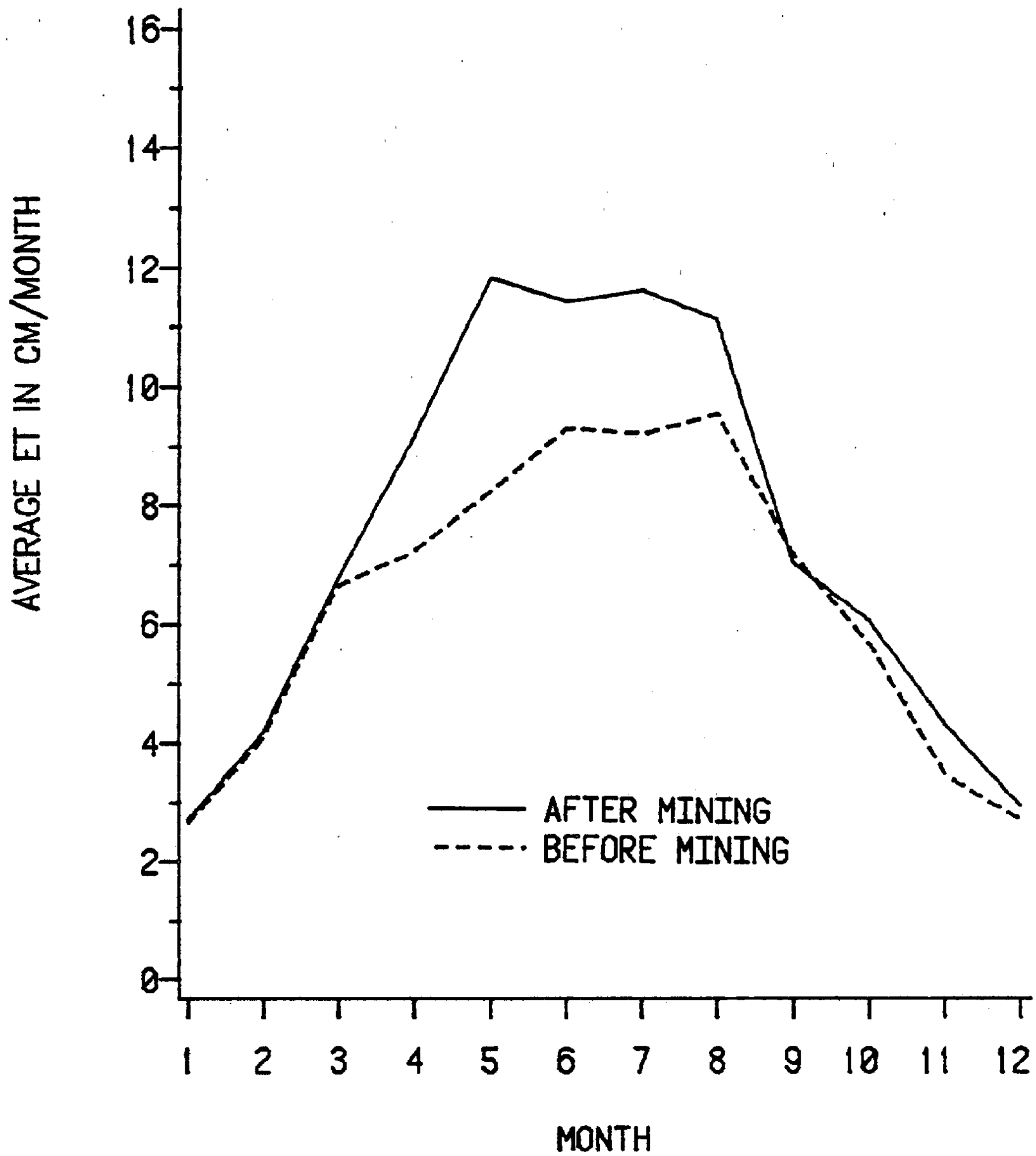

Figure 35. Comparison of actual evapotranspiration for before and after mining. Values are monthly averages from 20 years of simulation. Before mining vegetation was grass and after mining was a corn-winter wheat rotation.

Conversely, in winter and early spring months the post-mining winter wheat (maximum root depth of 15 cm) and bare soil (in spring) would be expected to exhibit slightly reduced ET in comparison to the pre-mining situation (constant 12 cm root depth). However, because of low PET (in winter and early spring) the water tables remain close to the surface resulting in similar monthly ETs for both cases for these periods (see Figure 35).

The differences in summertime ET between pre- and post-mining water balances are further illustrated in Figure 36. Figure 36 is a plot of the average, unit area, monthly total water loss, as predicted by DRAINMOD, compared for the before and after mining cases. The pre-mining situation exhibits lower average monthly flows when the post-mining case is in a fallow condition (January, February and March) but the monthly averages are much higher during the peak growing season (June,July and August) when ET is limited more often by soil water conditions in the pre-mining case because of the shallower root depths.

While the monthly averages give a good indication of trends, important effects may be obscured and it may be more informative to examine changes in the distribution of individual storms. The flow duration curve is a useful method for comparing the effects of various treatments and situations. However, to route enough flow periods to get a representative data base is very time consuming and expensive. Unit area, hourly water losses from the fields do not accurately represent canal flow rates as shown in Figure 37. However daily, unit area water losses converted to average flow rates are very close to the hourly routed flow rates as shown in Figure 38. Figure 38 was derived from the collector canal routed flows for the nineteen periods used in the model testing and the average daily flow rates from DRAINMOD for the exact same periods. The two curves match almost exactly. Because of the relative ease of producing very long periods of average daily flows it was decided to use these as being representative of expected flow rates in the canals.

Note: The rainfall data used to calculate long term monthly averages was taken from the Elizabeth City weather station. Elizabeth City is the closest rainfall station to the FCF mining site with hourly rainfall records and similar rainfall patterns. Unfortunately, a number of missing data records (including several whole months) existed during the period considered. Many of the shorter periods of missing data were "patched" using daily rainfall records from Elizabeth City. The daily values were distributed over four hours between 4 a.m. and 8 a.m. ET is normally distributed between 6 a.m. and 6 p.m. By using the above rainfall distribution the effect of rainfall on ET is restricted to 50% of the time. A total of 24 years between 1955 and 1979 were simulated for each condition desired. The four years with the worst rainfall data records were subsequently deleted from all analyses (1968, 1970, 1971 and 1973). The annual average rainfall for the remaining 20 years was 123 cm (48.3 in.) which compares closely with the estimated average at First Colony Farms of 127 cm (Heath, 1975).

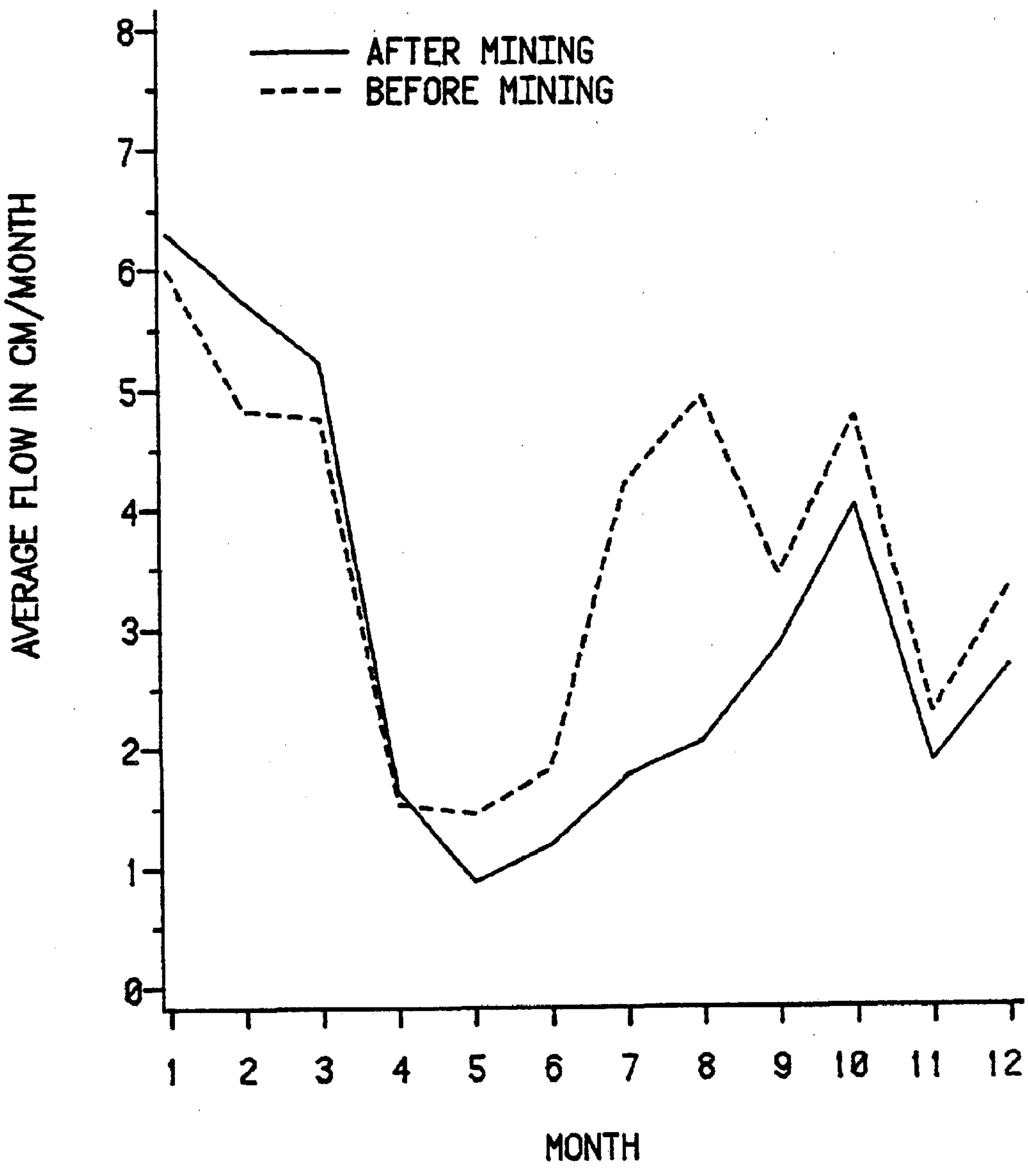

Figure 36. Comparison of flow volumes per unit area for before and after mining. Values are monthly averages from 20 years of simulation. Before mining vegetation was grass and after mining was a corn-winter wheat rotation.

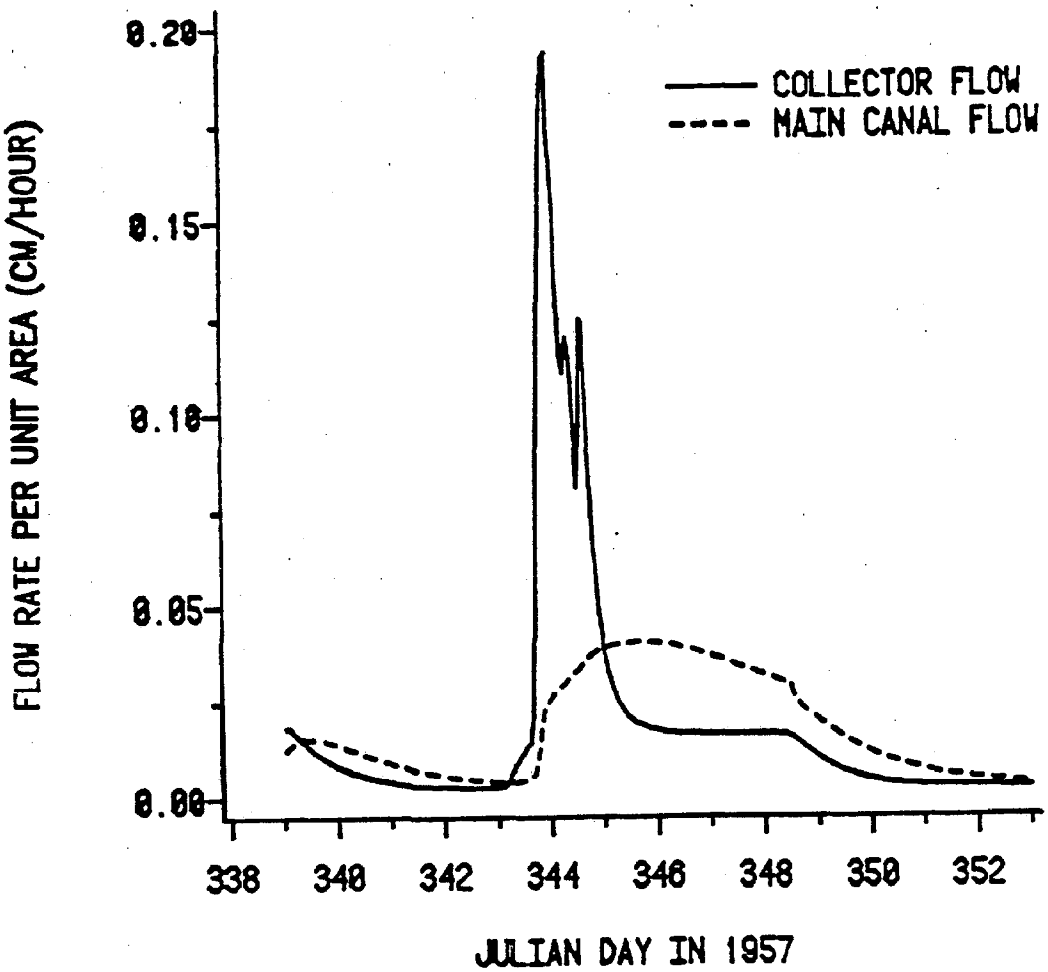

Figure 37. Comparison of collector canal outflow with main canal outflow on a per unit area basis for a 200 day return period peak flow. The simulations were for the pre-mining condition and illustrate the moderating effects of canal routing.

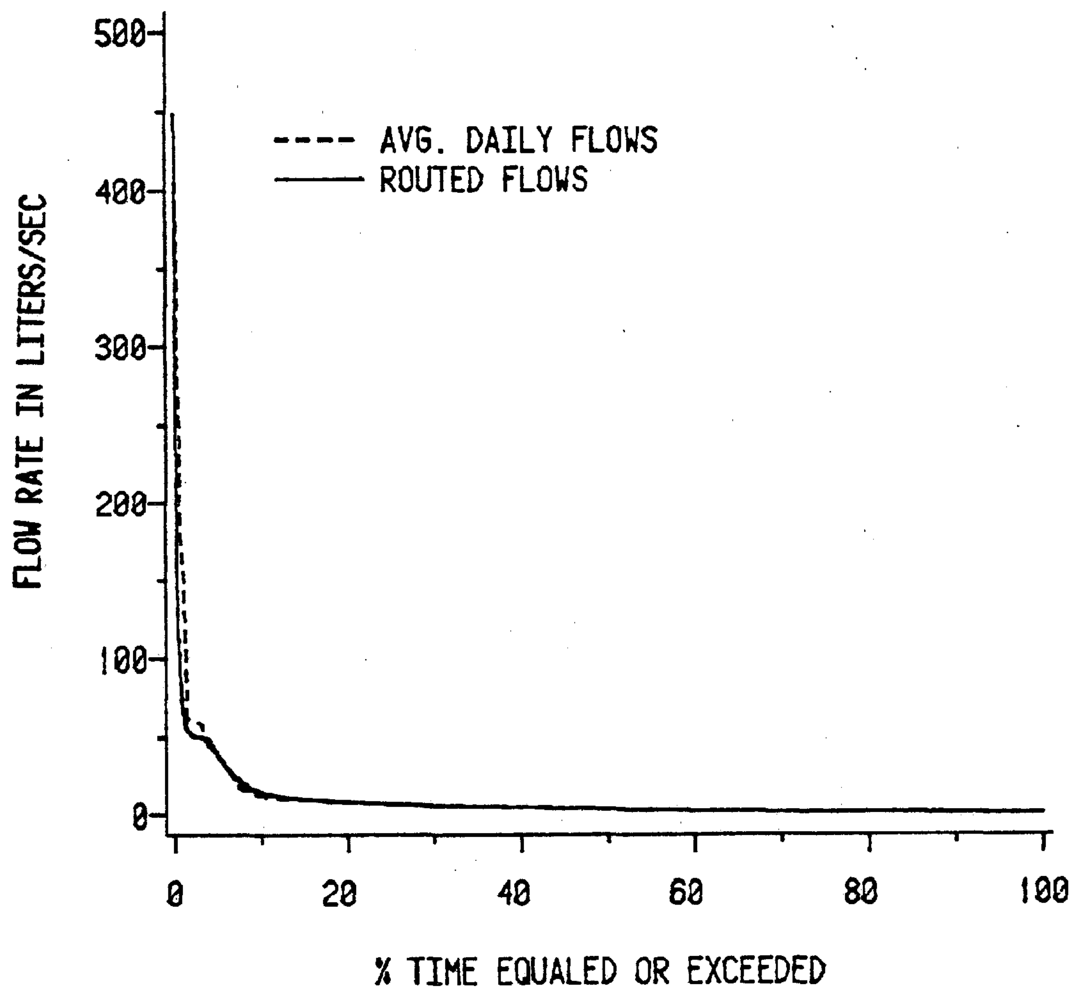

Figure 38. Comparison of hourly, routed collector canal outflows with average daily flows from the field edge as determined by DRAINMOD, converted to equivalent units for the before mining condition.

A flow duration curve using average daily flow rates from a 20 year simulation period is given in Figure 39 for conditions before and after mining. The flow duration curves for the before mining condition exhibit a plateau around the 5 to 10% mark. The flow rate at this plateau corresponds to water loss with the water table at or near the soil surface. The differences at very high flows between pre- and post-mining are small. Both soil profiles have a maximum daily average flow for the 20 year period of about 1800 liters/second (l/s). The slightly higher flows from the before mining case at flow rates less than about 10 l/s are caused by the greater bank storage flow from the peat soils. The higher conductivity soil in the after mining situation would not allow as complete filling of the soil profile close to the ditches after a rainfall. Thus not as much water was available in the banks for subsequent release to the ditches. However, because of the wide ditch spacings, the flow rates from the collector canals are less than 10 l/s (.35 cfs) for over 80% of the time.

The flow rates of major concern for flooding, nutrient loss and erosion are usually the very high flows. The effects of mining and reclamation were looked at more closely for individual storm hydrographs. DeHoog canal was selected as a typical outlet for the permitted area and storm hydrographs in DeHoog canal at the permitted area boundary were simulated. Several storms were selected for hydrograph comparison. The routed outflow from the collector canals was determined as described under model testing. Outflow from the collector canals was taken as new input to the flood routing model and the hourly flows were routed down DeHoog canal to the outlet.

An average cross section of DeHoog canal was used as input to the second level of flood routing. The canal specifications are given in Table 20. Eleven blocks contribute to the flow in DeHoog canal before it leaves the permitted mining area. The two farthest upstream blocks, however, are not in the permitted area. When simulating the after mining situation these two blocks were left in the before mining condition.

The 50, 75, 100 and 200 day return period daily flow rates were selected for the analysis. These return period flows only cover a very narrow range on the flow duration curve, but cover the range of likely problem flows. The return period flows were determined by ranking the pre-mining, average daily flow rates. The rank of a particular return period was found from;

$$\text{rank} = \frac{n + 1}{T} \qquad (9)$$

where n is the total number of daily flow periods and T is the desired return period. Because an individual storm may have several days of high flow this method is only approximate. The selection of storms by the above method is justified because of

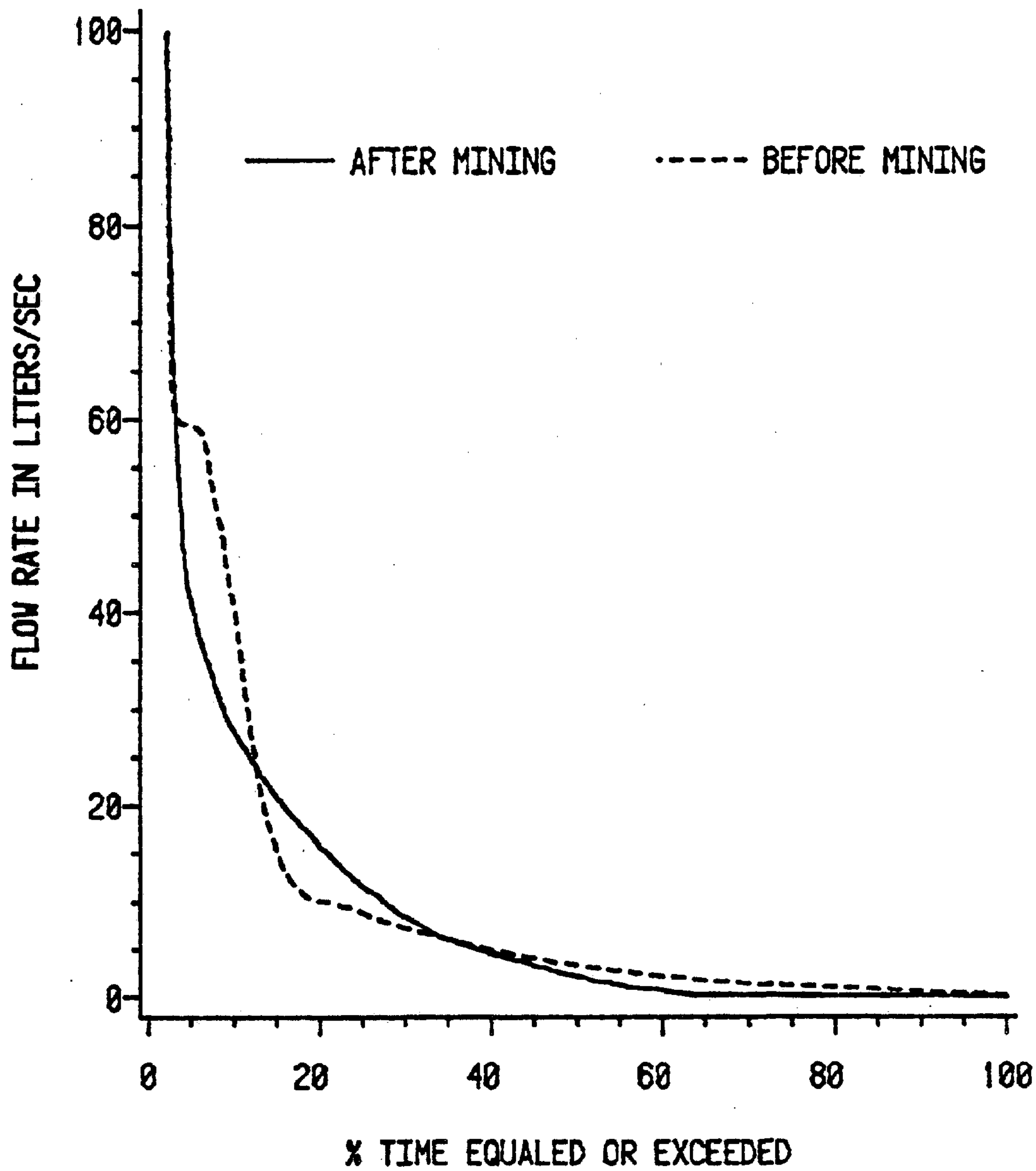

Figure 39. Comparison of before and after mining flow duration curve. Values are average daily flows, as determined by DRAINMOD, converted to collector canal outflows. Twenty years of daily values are represented for each condition. Note that the vertical scale is truncated at 100 ℓ/s. The curves are coincident above this point.

Table 20. Main canal (DeHoog canal) specifications for input to flood routing model.

INPUT	VALUE
Channel dimensions	
Depth	245 cm
Bottom width	122 cm
Side slope (horizontal:vertical)	1.8:1
Manning's roughness coefficients	
Channel	0.20
Overbanks	1.50

the close association between daily flow rates from DRAINMOD and hourly, routed collector canal outflows (see Figure 38). It is very important to look at canal flows in this analysis. Often very dramatic field edge differences will be significantly reduced by the frictional and mixing effects of a canal reach. This is clearly shown in Figure 37 in which the routed hydrographs at the collector canal and main canal outlets are compared on a per unit area basis.

Figures 40 to 43 illustrate the before and after mining DeHoog canal hydrographs for the 200, 100, 75, and 50 day return period flow rates. The after mining peaks are all greater than the before mining peaks by an average of 16% for the 200, 100 and 75 day return period storms and about 107% for the 50 day return period storm.

The abrupt changes in the hydrographs are because of the necessity of delineating different conditions within the models by discrete values. The abrupt change on the rising portion of the hydrographs at about 1000 l/s corresponds to the onset of overbank flow. The bulge on the falling portion of the hydrograph from the before mining case corresponds to the switch from surface runoff to flow primarily through the surface cured layer of peat.

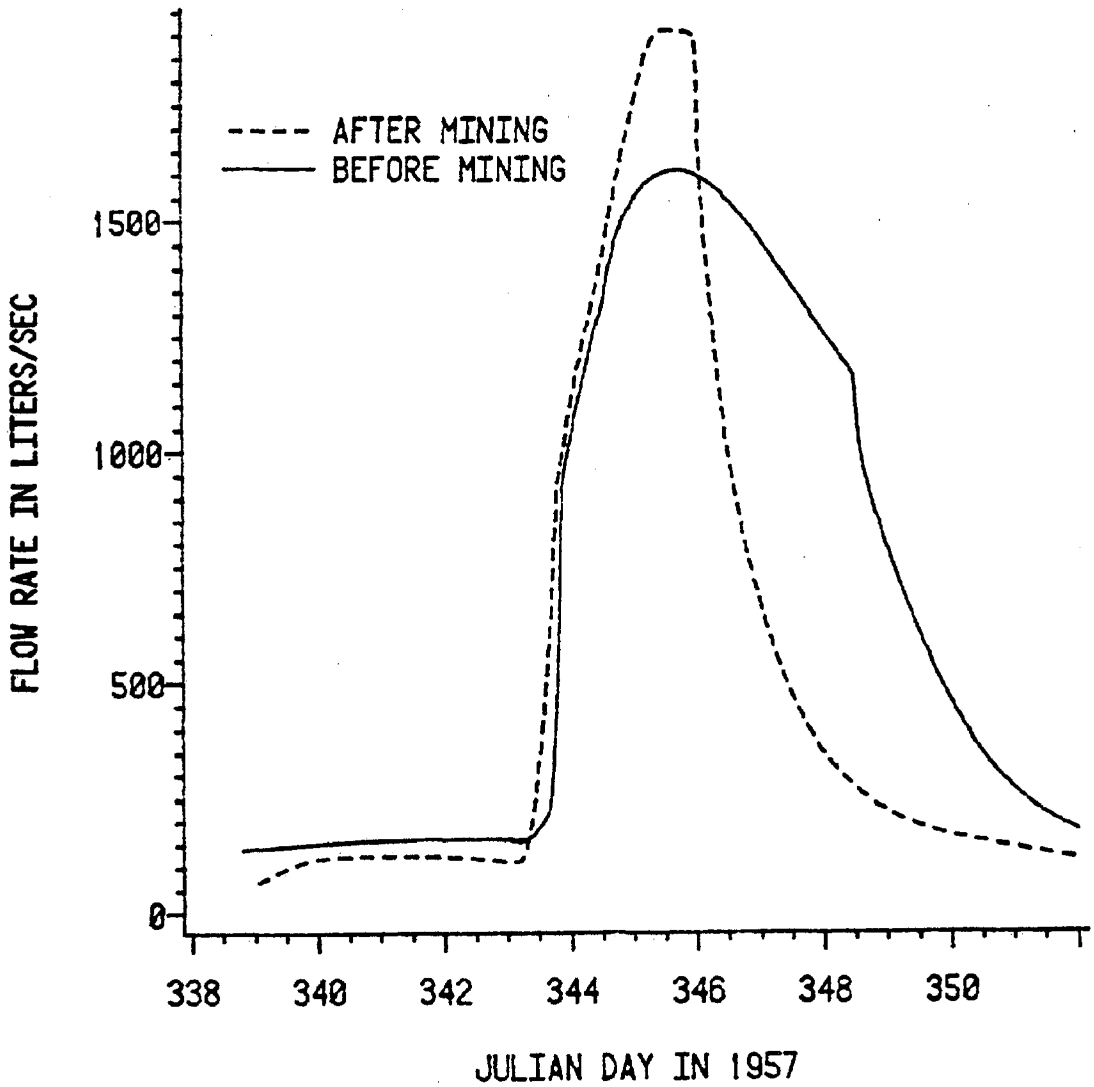

Figure 40. Comparison of before and after mining hydrographs for a 200 day return period peak flow. Drainage for both conditions was by conventional open ditches which provides poor subsurface drainage.

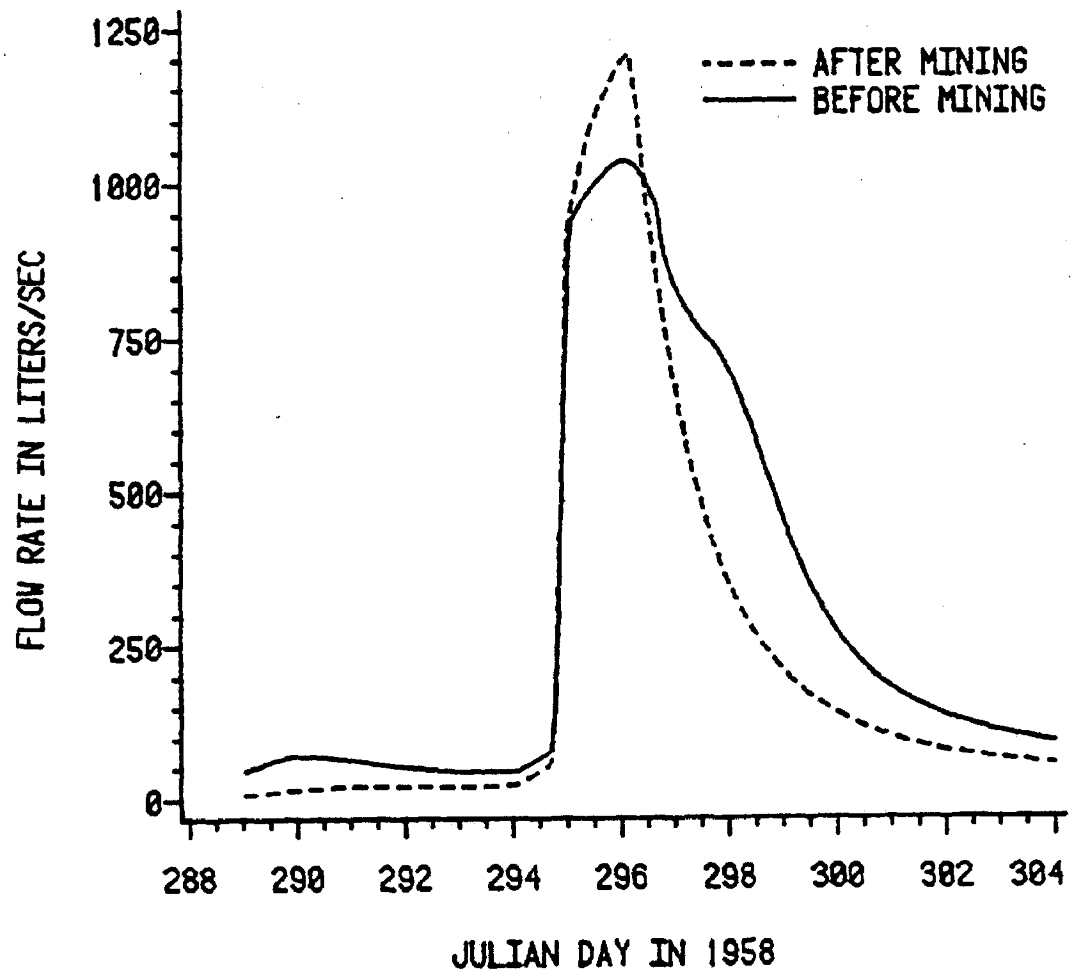

Figure 41. Comparison of before and after mining hydrographs for a 100 day return period peak flow. Drainage for both conditions was by conventional open ditches 100 m apart which provides poor subsurface drainage.

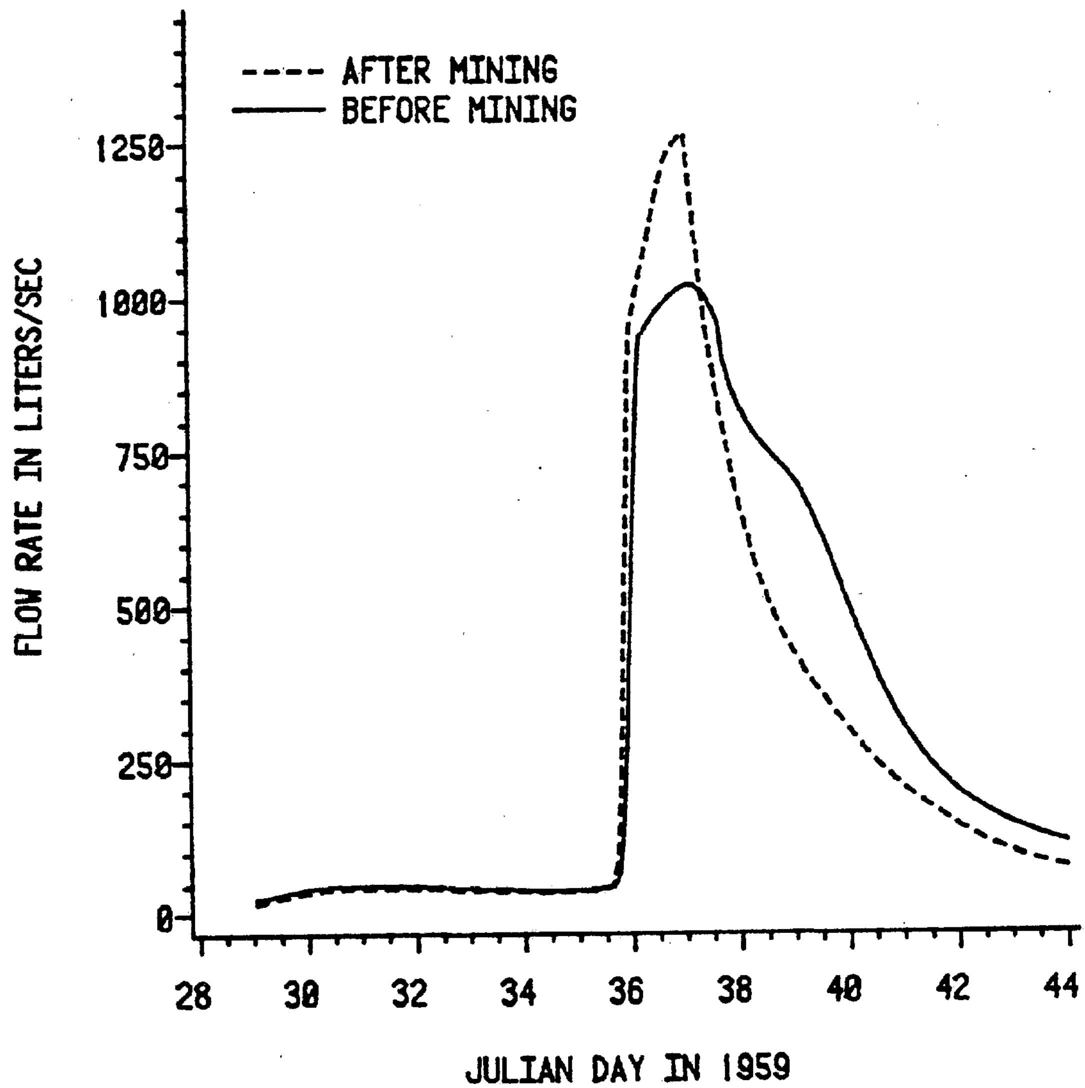

Figure 42. Comparison of before and after mining hydrographs for a 75 day return period peak flow. Drainage for both conditions was by conventional open ditches 100 m apart which provides poor subsurface drainage.

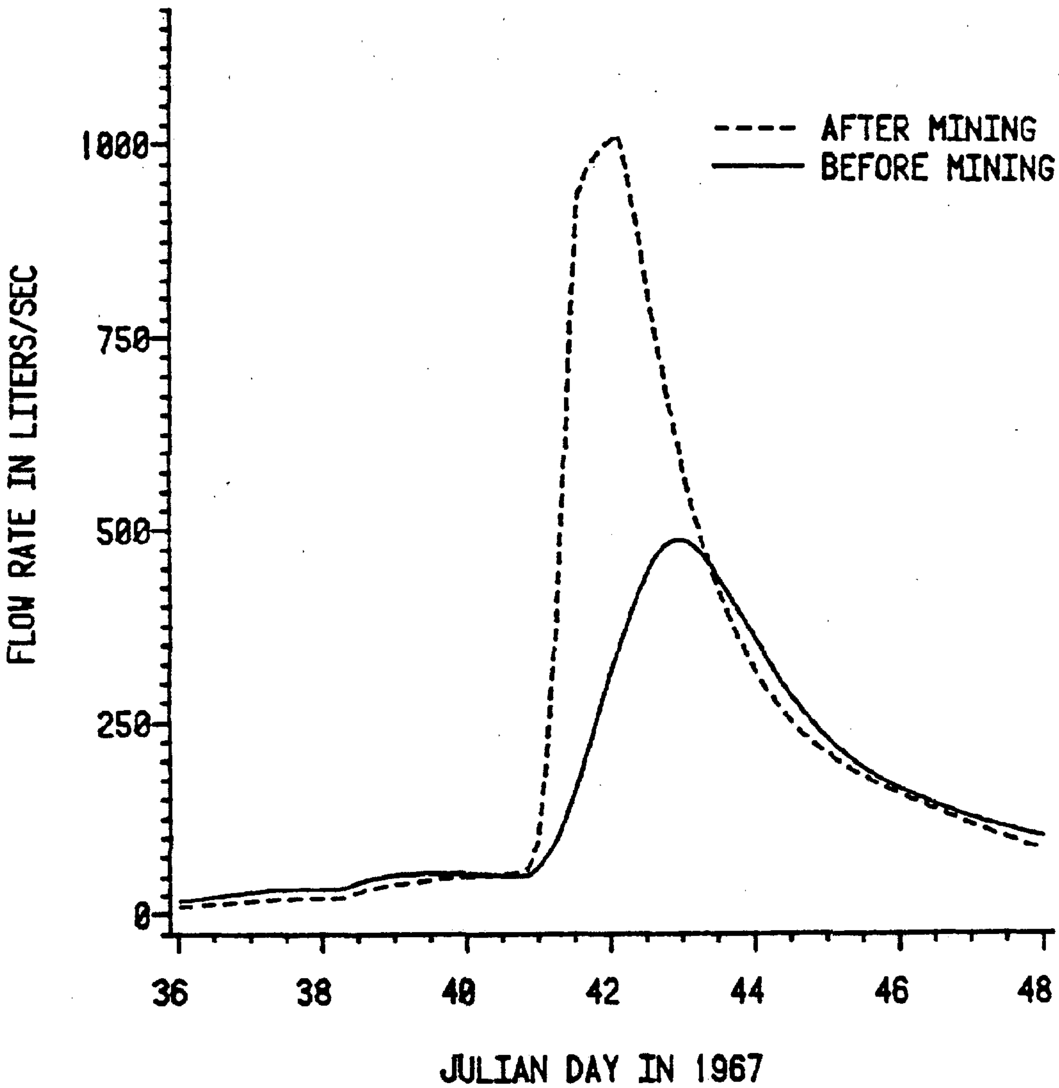

Figure 43. Comparison of before and after mining hydrographs for a 50 day return period peak flow. Drainage for both conditions was by conventional open ditches 100 m apart which provides poor subsurface drainage.

For flows in excess of the 100 day return period storm all canal banks are breached to the extent that the hydrographs become very similar for the two conditions. At flow rates in excess of 1200 l/s in Figure 40 a substantial portion of the flow would be overland flow which may be held up by road berms or other obstacles and not be true canal flow as portrayed.

The differences in shape and magnitude of the hydrographs are primarily dependent on antecendent soil water conditions, the assumed soil surface conditions and the hydraulic conductivity of the soil surface layers. A drain spacing of about 20 m would be required to provide good subsurface drainage. Therefore the subsurface drainage was poor (drain spacing of 100 m) for all cases considered in this section. For all four comparisons the water table prior to rainfall was deeper for the before mining simulation. Thus more storage was available in the soil profile and the after mining peak is higher. The total volume of flow (the area under the hydrographs) is greater for the before mining case for the 200, 100 and 75 day return period flows because of the continued high flow rate through the surface layer of peat.

In Figure 43 (50 day return period comparison) the water table depth for the before mining case is substantially deeper than for the after mining case prior to the rainfall event. The water table does not reach the surface in the before mining case and no surface runoff is predicted. In contrast, surface runoff is predicted for the after mining case and a much higher peak flow is recorded.

All of the above storms occurred during October to February, covering periods of fallow conditions and shallow rooting depths in the after mining condition. In this period the water table would be expected to be deeper in the before mining situation and thus peak flow rates less. To show the effects of a deeper water table for the after mining case during the growing season a simulation was run using data for the entire month of July 1959. The winter months were originally selected because of the discrete nature of the storms. The summer months usually contain many convective storms often occurring on several consecutive days. This makes it difficult to select individual events for analysis. Figure 44 contains the simulated flow rates for the DeHoog canal outlet for the month of July 1959. A total of 25.35 cm of rain fell during this month with rain falling on 18 of the 31 days. The initial water table depth on July 1st was deeper for the after mining case by 4 cm but, more importantly, the depth of soil which was dried to the wilting point at the soil surface was calculated to be 30 cm for the after mining case but only 12 cm for the before mining case. These depths correspond to the assumed effective rooting depth for the crops. The very dry preceding month had dried the root zone out completely. The shallow before mining rooting depth is because of the frequently high water tables and the acidic subsoil restricting root growth. Thus for July 1959 the soil profile filled more quickly in the before mining case producing surface runoff on Julian day 192, 5

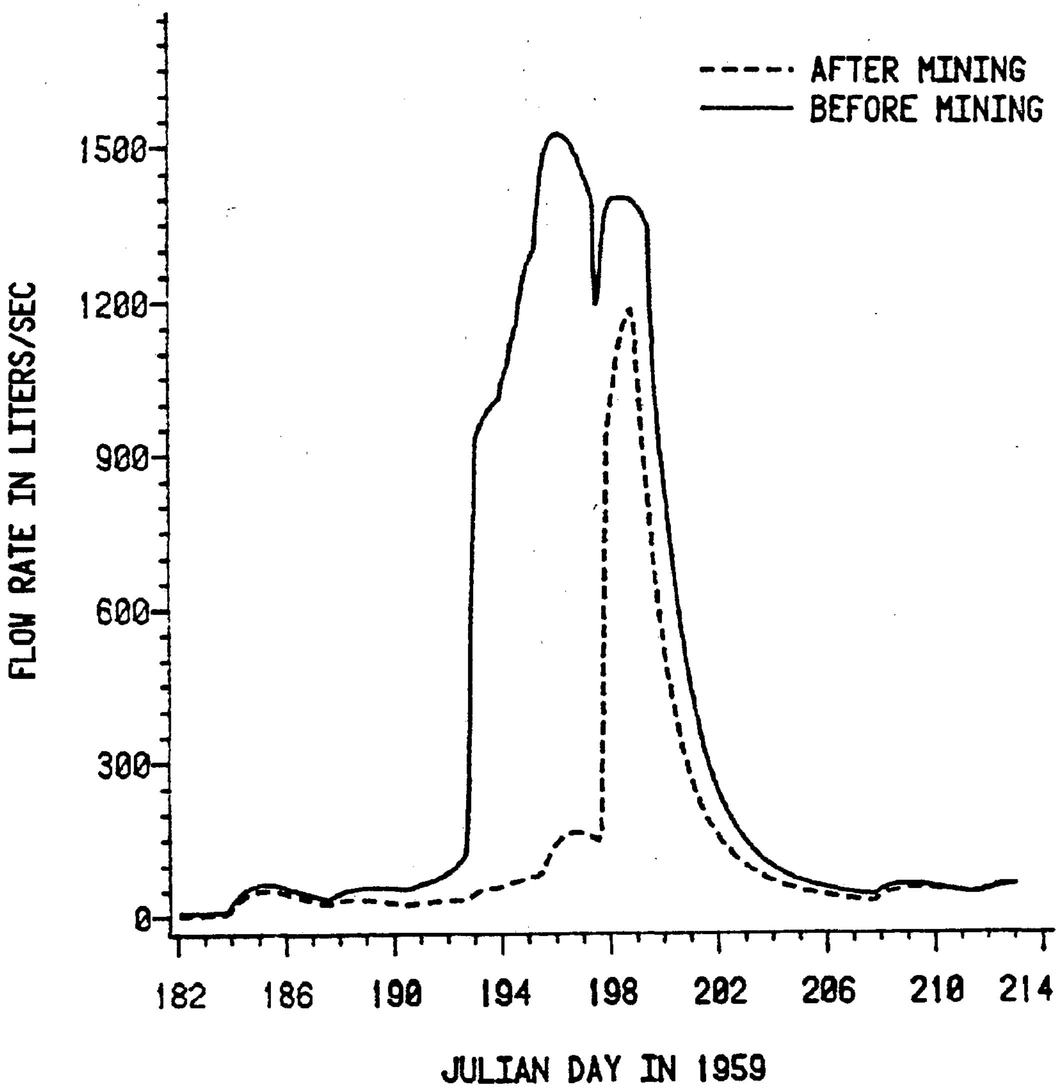

Figure 44. Comparison of before and after mining hydrographs for a mid-summer storm. Drainage for both conditions was by conventional open ditches which provides poor subsurface drainage.

days before the after mining situation and as a result considerably more total water loss over the month.

Similarly, Figure 45 is for a heavy (5.6 cm) rainfall late in the growing season. A deeper initial water table and dry root zone in the after mining situation results in a lower hydrograph peak.

These results are dependent on the surface cover and root depths before and after mining. A number of blocks, primarily in the north-west corner of the permitted area, untouched by the 1981 fire, are covered by a dense growth of natural pocosin vegetation. The hydrology of these areas is dominated by a layer of highly porous leaf and root material overlying the peat. This layer essentially prevents any true overland flow as the conductivity of this layer is sufficient to transport all water to the canals. This layer also inhibits evaporation keeping the water table high compared to the grassed areas. Although the total annual water loss from the areas in natural vegetation is similar to grassed areas, the hydrograph peaks are depressed by the increased time of travel through the root mat rather than the rapid overland flow. More work needs to be done, however to quantify these effects.

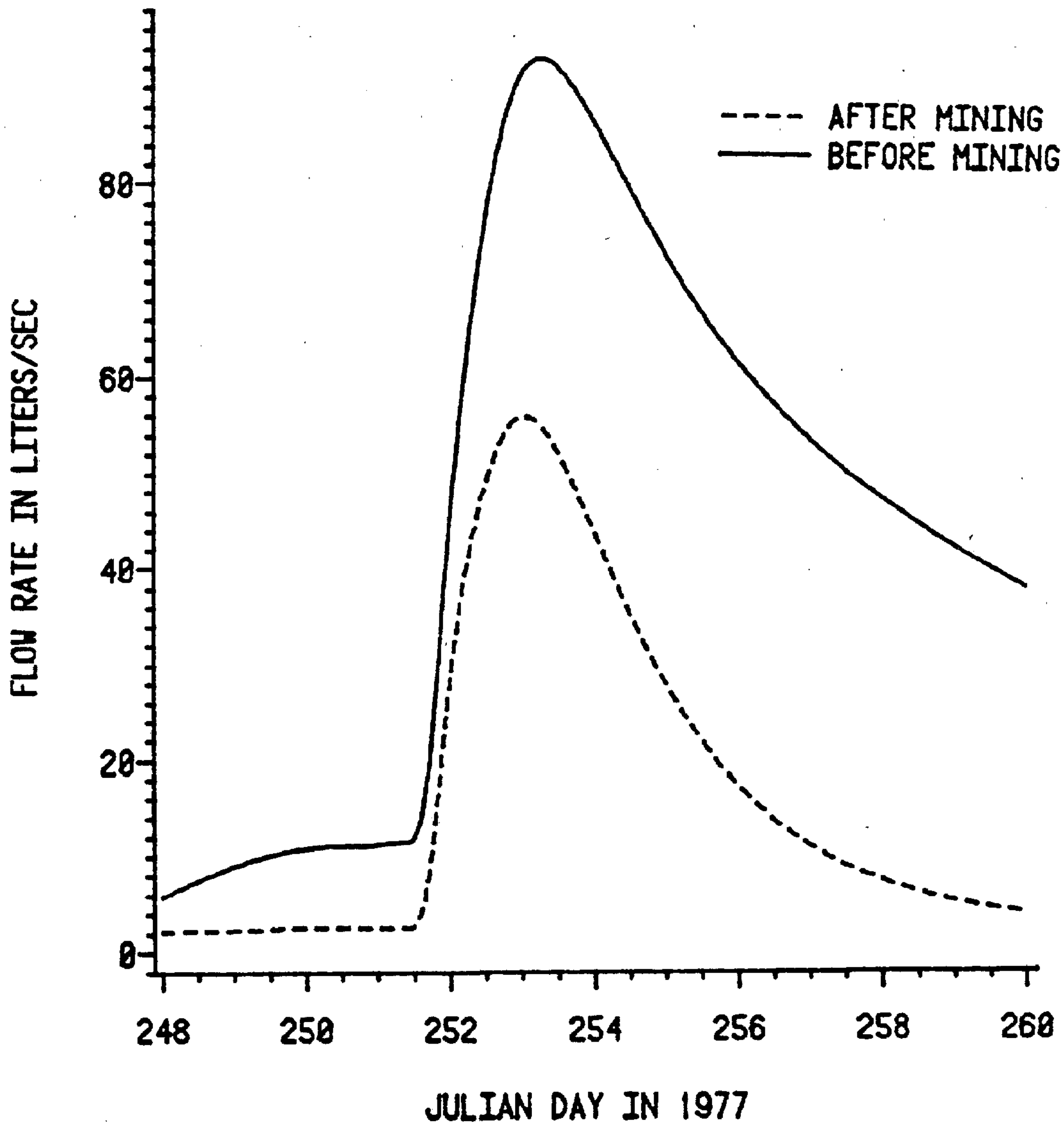

Figure 45. Comparison of before and after mining hydrographs for a growing season storm producing no surface runoff. Drainage for both conditions was by conventional open ditches which provides poor subsurface drainage.

Hydrologic Effects of Alternative Post-Mining Drainage Systems

Conventionally, in the Coastal Plain of North Carolina, agricultural drainage systems are designed to remove surface water. Open ditches are dug approximately 100 m apart and the land surface is sloped towards these ditches in "turtle back" fashion to encourage overland flow to the ditches. Disadvantages of these systems include soil and nutrient loss by excessive sheet runoff, land lost from productivity at and near the ditches and insufficient water table drawdown during the planting and harvesting seasons to allow machinery on the fields. Also, while removing the surface water efficiently, water table drawdown during the growing season is almost solely dependent on ET. Frequent summer rains often maintain high water tables resulting in shallow root growth and poor crop development and yields.

An alternative drainage system design involves the placement of subsurface drain tubing at close intervals. Drain tubes placed at 100 cm deep and 20 m apart were selected as an optimum design for the reclaimed mining area. The design was selected using techniques described in Hardjoamidjojo and Skaggs (1982) to maximize the long term average corn yield by minimizing crop stress due to excess water and planting date delays due to lack of spring trafficability.

Model inputs for the poor subsurface drainage treatment in this analysis were identical to the after mining case in the previous section. The drain spacing and affected drainage parameters were altered for the good subsurface drainage treatment and are given in Table 19. Again the crop rotation used was corn-winter wheat.

Water table depths for a year of average rainfall (1960) are compared for the two drainage treatments in Figure 46. The average water table depth for the good subsurface drainage treatment (86.8 cm) is 21.9 cm greater than for the poor subsurface drainage case. The average monthly flow totals from a unit field area are given in Figure 47. The flow volume loss for the field with good subsurface drainage is consistently greater by an average of 10% or 3.5 cm/year.

A flow duration curve using average daily flow rates for these two drainage conditions is given in Figure 48. The good subsurface drainage appears to cause an increase in the average daily flow rates for low flows. However, good subsurface drainage reduced flows in the 60 l/s to 1000 l/s range. As emphasized in the previous section, it is these higher flows which may be more important as a design consideration. Figure 49 is an expanded scale look at the flow duration curve of Figure 48. Flow rates from the well drained field are consistently lower at these higher flow rates. For example, the daily flow

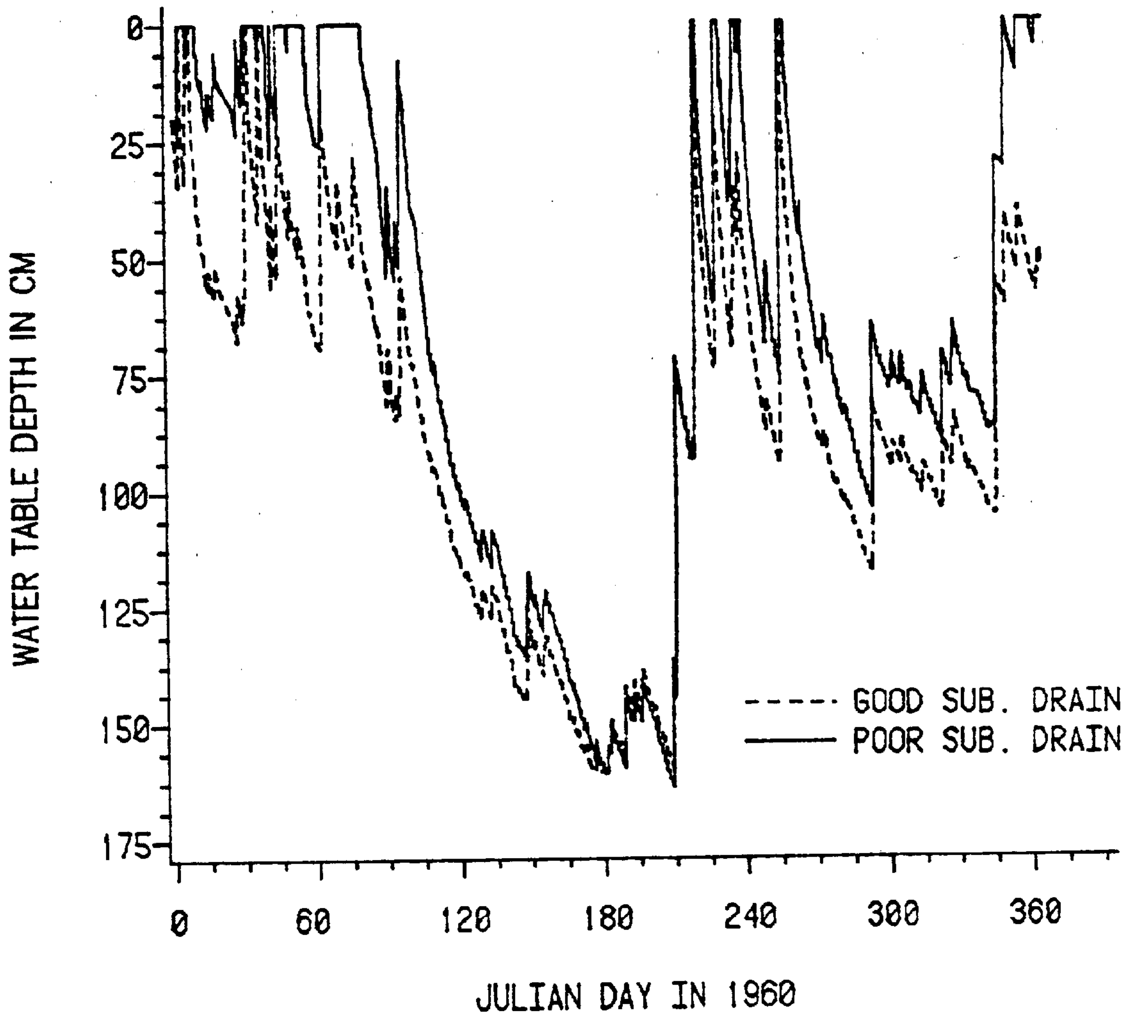

Figure 46. After mining water table depth comparison for a typical First Colony Farm mining site for a year of average rainfall. Comparison is between good and poor subsurface drainage. A corn-winter wheat rotation was used for both cases.

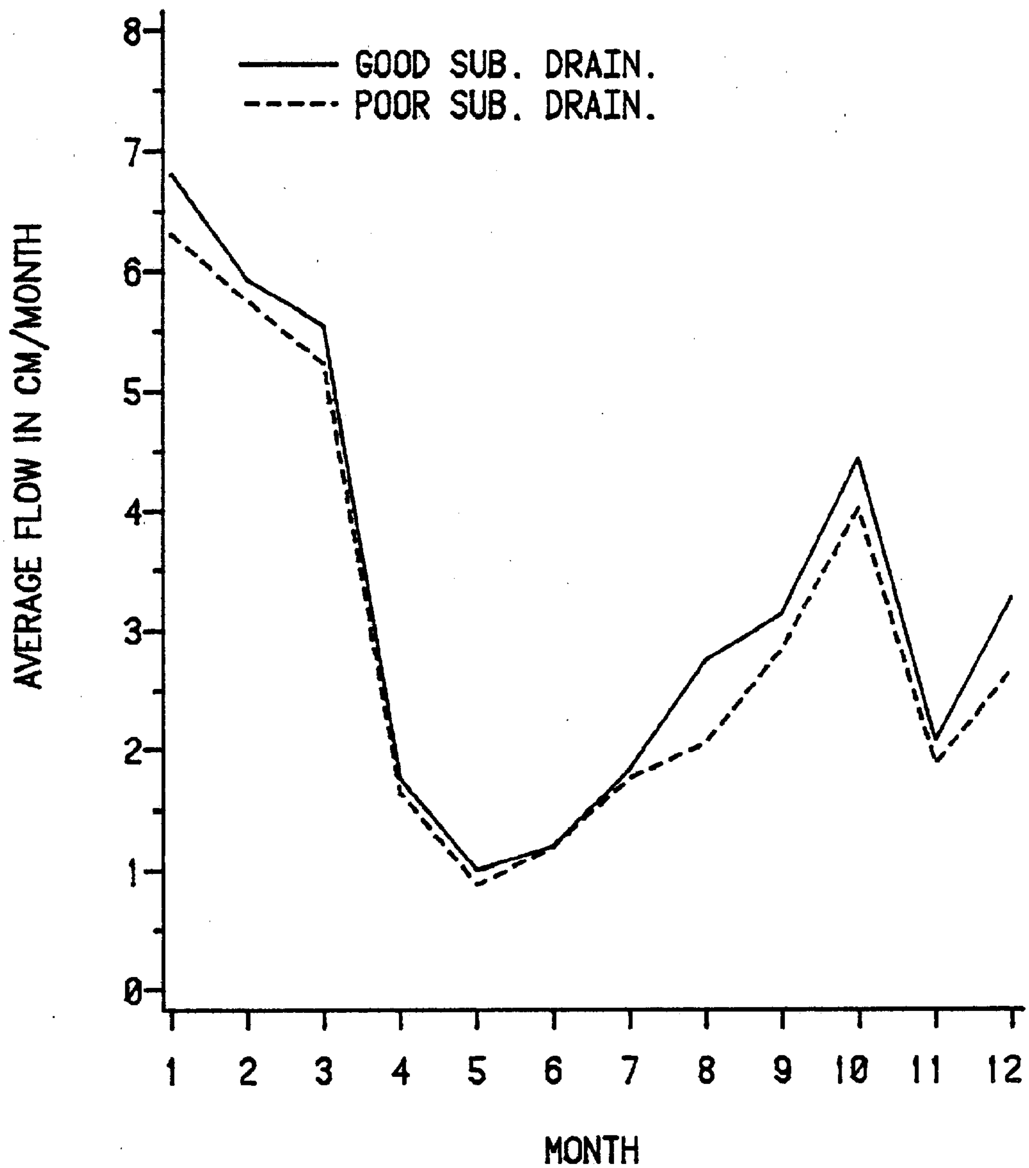

Figure 47. Comparison of monthly flow volumes per unit area between good and poor subsurface drainage conditions. Values are monthly averages from 20 years of simulation. A corn-winter wheat rotation was used for both cases.

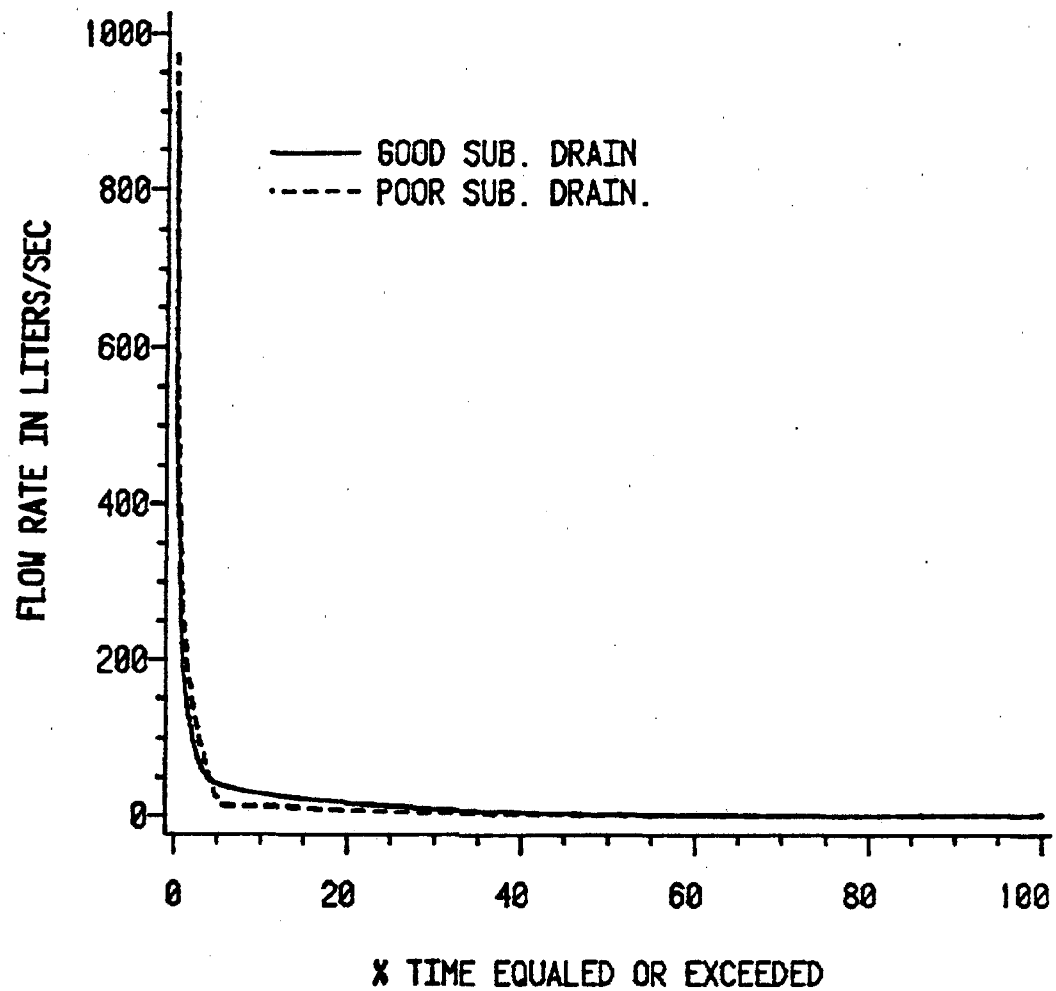

Figure 48. Flow duration curve comparison between good and poor subsurface drainage treatments. Values are average daily flows simulated by DRAINMOD. Twenty years of daily values are represented for each condition.

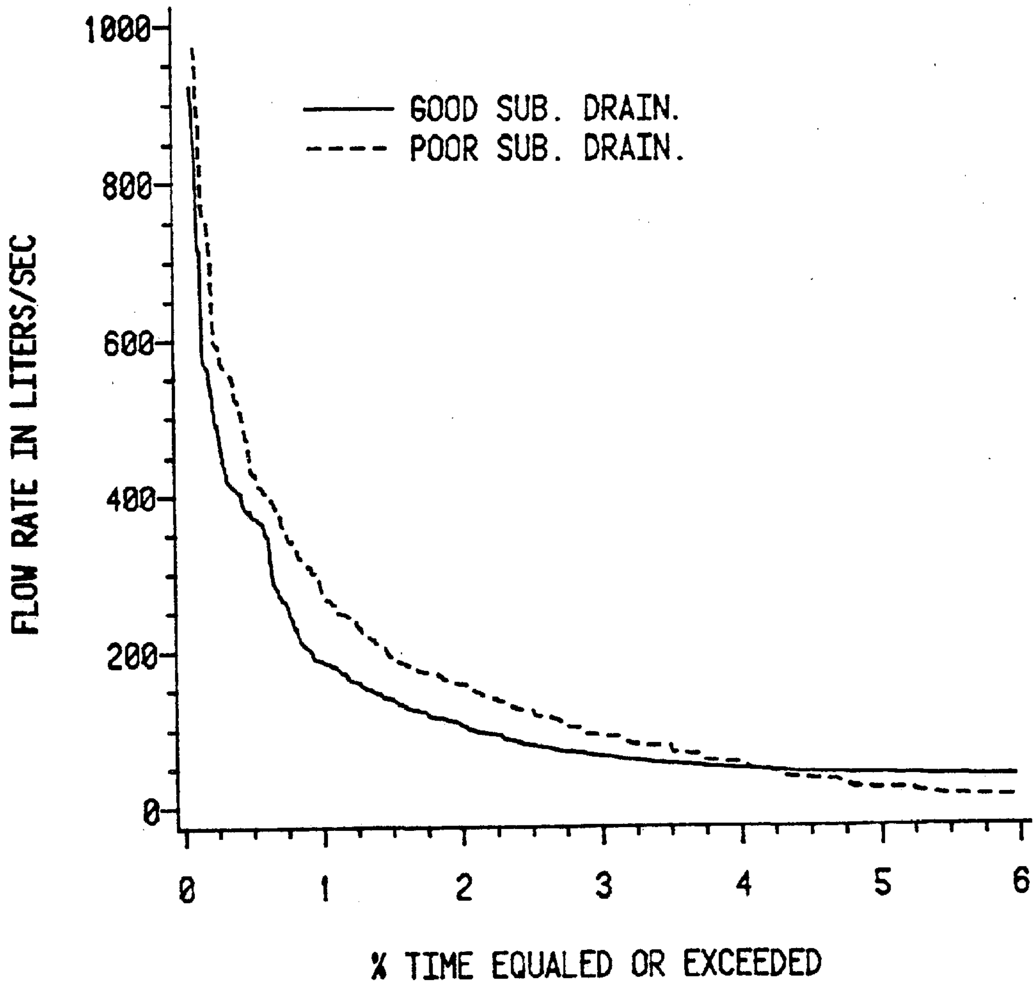

Figure 49. Flow duration curve comparison between good and poor subsurface drainage treatments. Note that the horizontal scale is limited to less than 6% time equaled or exceeded.

rate which is expected to be equaled or exceeded 1% of the time equals 285 l/s for the poor subsurface drainage case but equals 180 l/s for the field with good subsurface drainage (a decrease of 37%).

Good subsurface drainage maintains a steady, constant outflow from the soil profile. This outflow continues when the drainage from a similar but poorly drained field has ceased. This lowers the water tables as shown in Figure 46 and increases the average outflow as shown in Figure 47. However, because of the deeper water tables, the soil profile of the well drained field has a greater storage capacity for subsequent rainfall. This is the reason for the reduced flow rates illustrated in Figure 49. In a field with good subsurface drainage the proportion of subsurface flow and surface runoff will be significantly altered from that which will occur for poor subsurface drainage. Figure 50 gives the average monthly subsurface flow and surface runoff for the case with poor subsurface drainage. Except for the high rainfall months, the average monthly totals of subsurface drainage and surface runoff are of similar magnitudes. Conversely, for the field with good subsurface drainage (Figure 51), the average monthly surface runoff is consistently less than 50% of the subsurface drainage and very close to zero for some months. Thus the well drained field releases its water more evenly during the year through the drain tubes thereby reducing high peak outflows and flash floods. The total surface runoff from the area with good subsurface drainage can be expected, on average, to be about 5.3 cm (57%) less per year than for the conventional, wide spaced ditch drainage for this area.

The effects of improving the subsurface drainage on individual hydrographs is illustrated in Figures 52 to 56 for peak flows of varying magnitudes. As can be seen in Figure 48 as well as in Figures 52 and 53 there is little difference between drainage treatments for the extremely high flows. If the rainfall is sufficient to bring both water tables to the surface, and wholesale flooding occurs (i.e. the canal banks are breached), then the predicted hydrographs for both cases will be similar. Major differences occur for the flows which fall between 60 and 1000 l/s. These effects are clearly evident in varying proportions in Figures 54 to 56. The buffering capacity of the soil profile was sufficient in Figure 55 to prevent any surface runoff in the soil with good subsurface drainage thus keeping the peak outflow significantly less than for the poor subsurface drainage case. For the storm illustrated in Figure 56 no surface runoff is predicted for either case but the rapid lowering of the water table between each rainfall event by the subsurface drains keeps the peak outflow rates about 25% less than for the case with poor subsurface drainage. Note, however, that the flow rates between, after and often before rainfall events are higher for the hydrographs with good subsurface drainage than for the hydrographs with poor subsurface drainage.

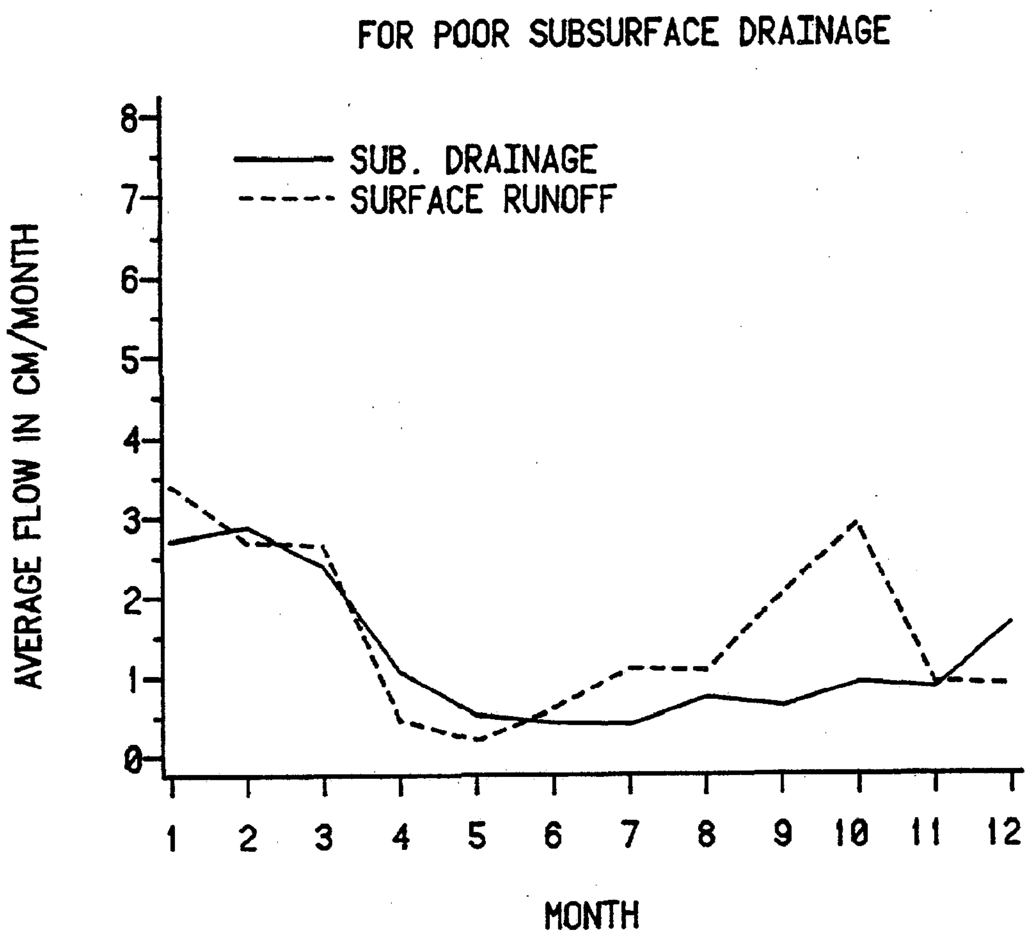

Figure 50. Average monthly flow volume from subsurface drainage and surface runoff for the post-mining condition with poor subsurface drainage. Values are monthly averages for 20 years of simulation.

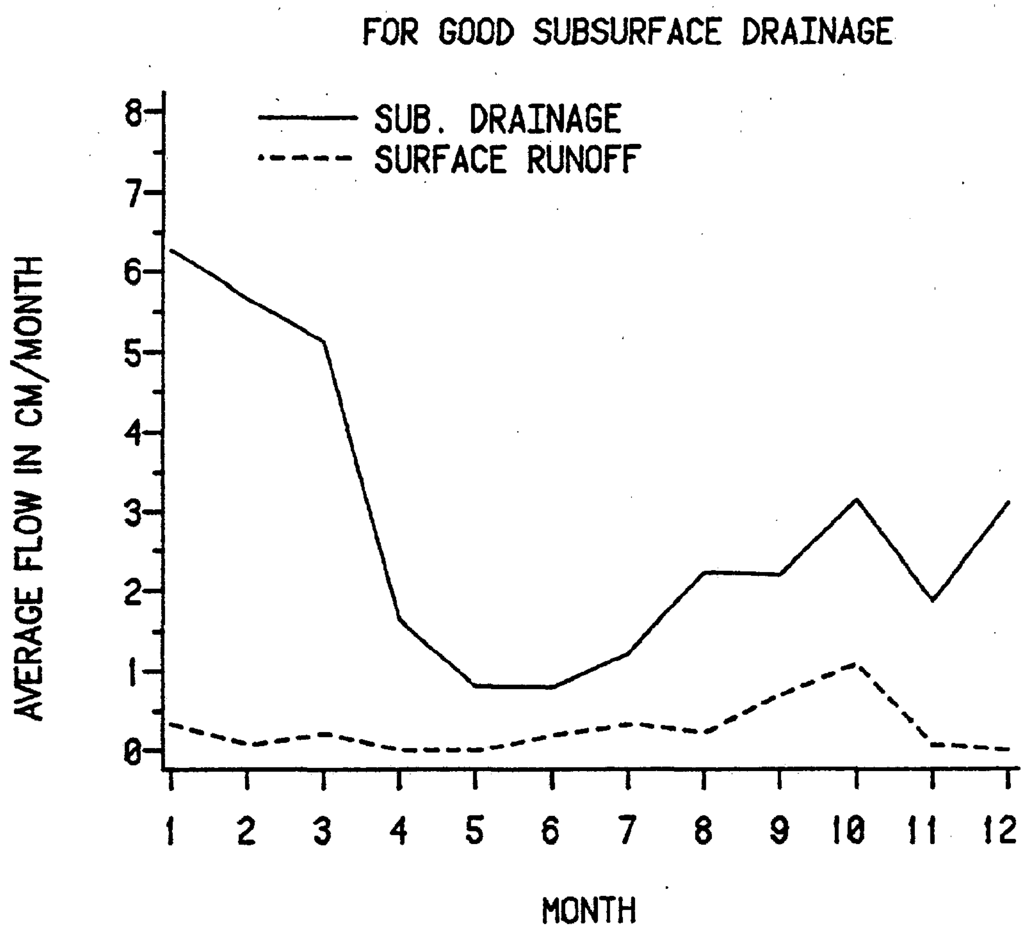

Figure 51. Average monthly flow volume from subsurface drainage and surface runoff for the post-mining condition with good subsurface drainage. Values are monthly averages for 20 years of simulation.

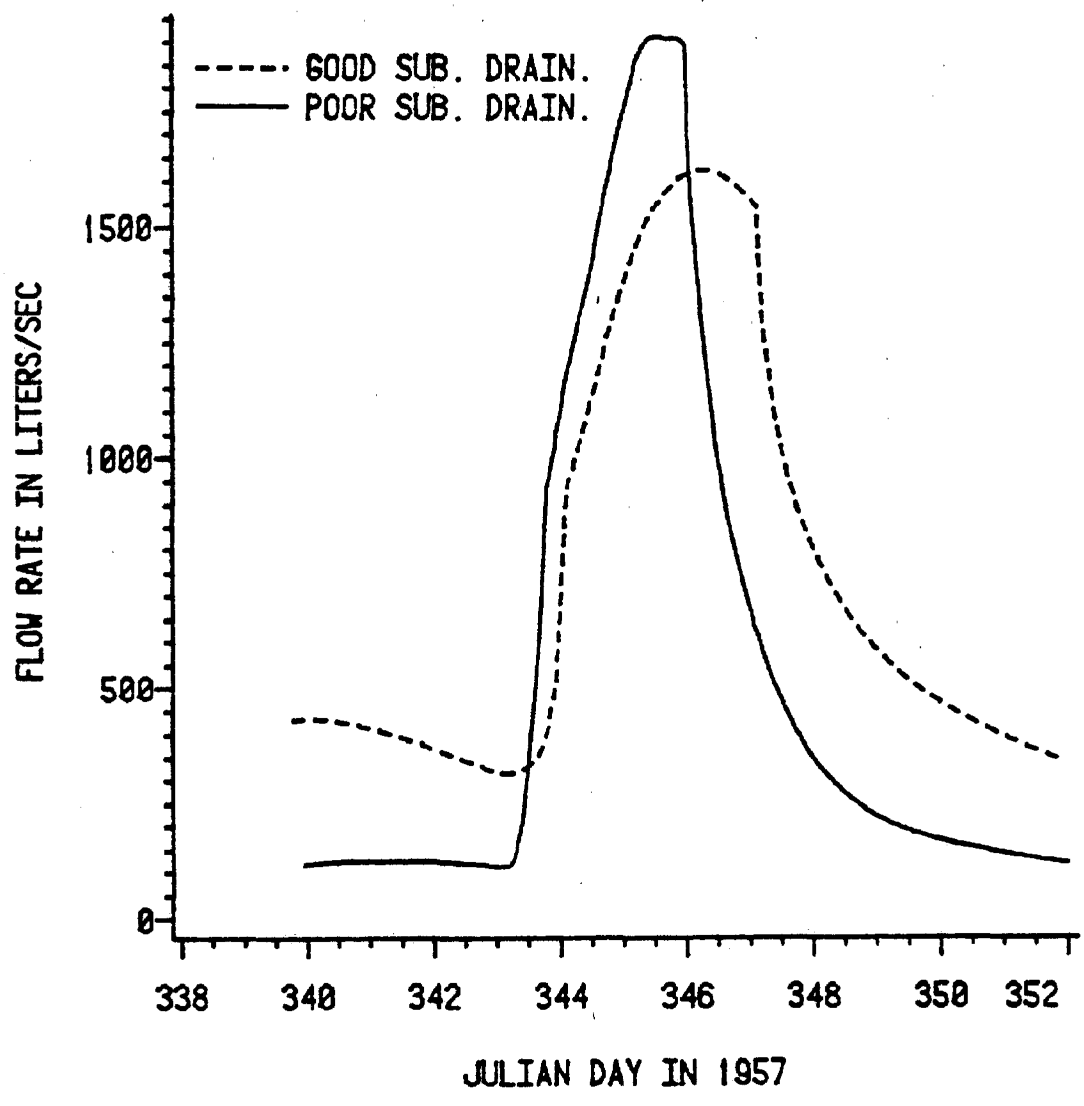

Figure 52. Good and poor subsurface drainage hydrograph comparison for post-mining for a 200 day return period flow. Eleven blocks (1430 ha) contribute flow to each hydrograph.

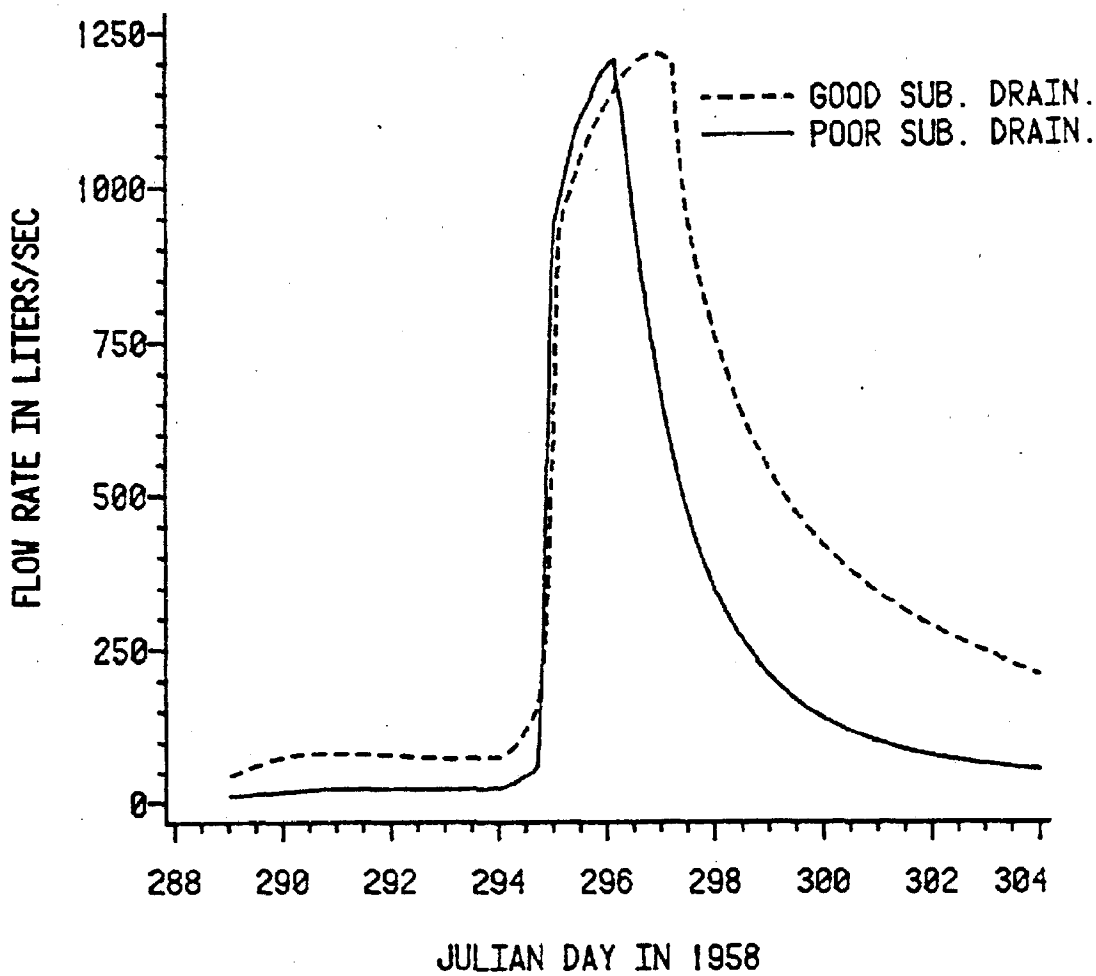

Figure 53. Good and poor subsurface drainage hydrograph comparison for post-mining for a 100 day return period flow. Eleven blocks (1430 ha) contribute flow to each hydrograph.

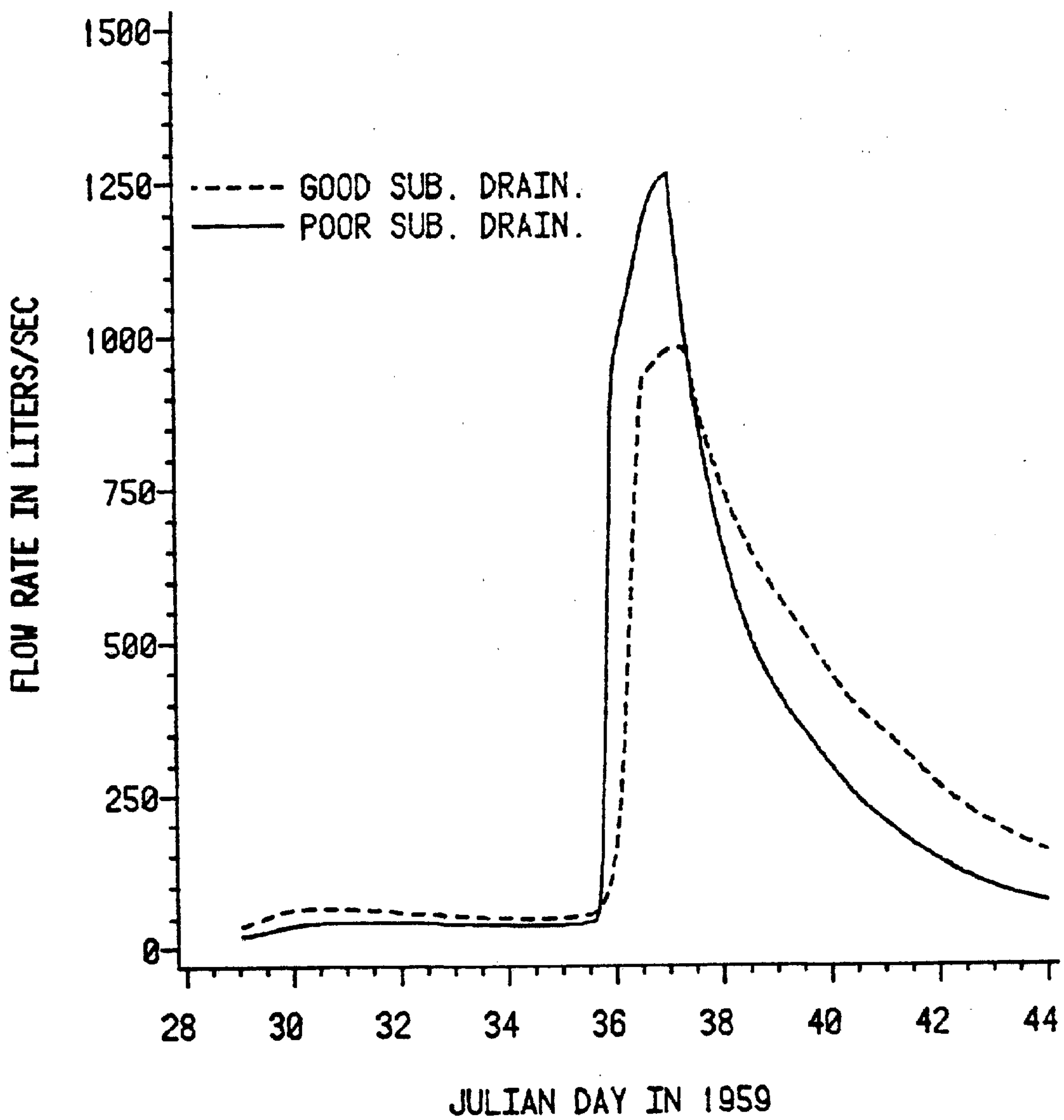

Figure 54. Good and poor subsurface drainage hydrograph comparison for post-mining for a 75 day return period flow. Eleven blocks (1430 ha) contribute flow to each hydrograph.

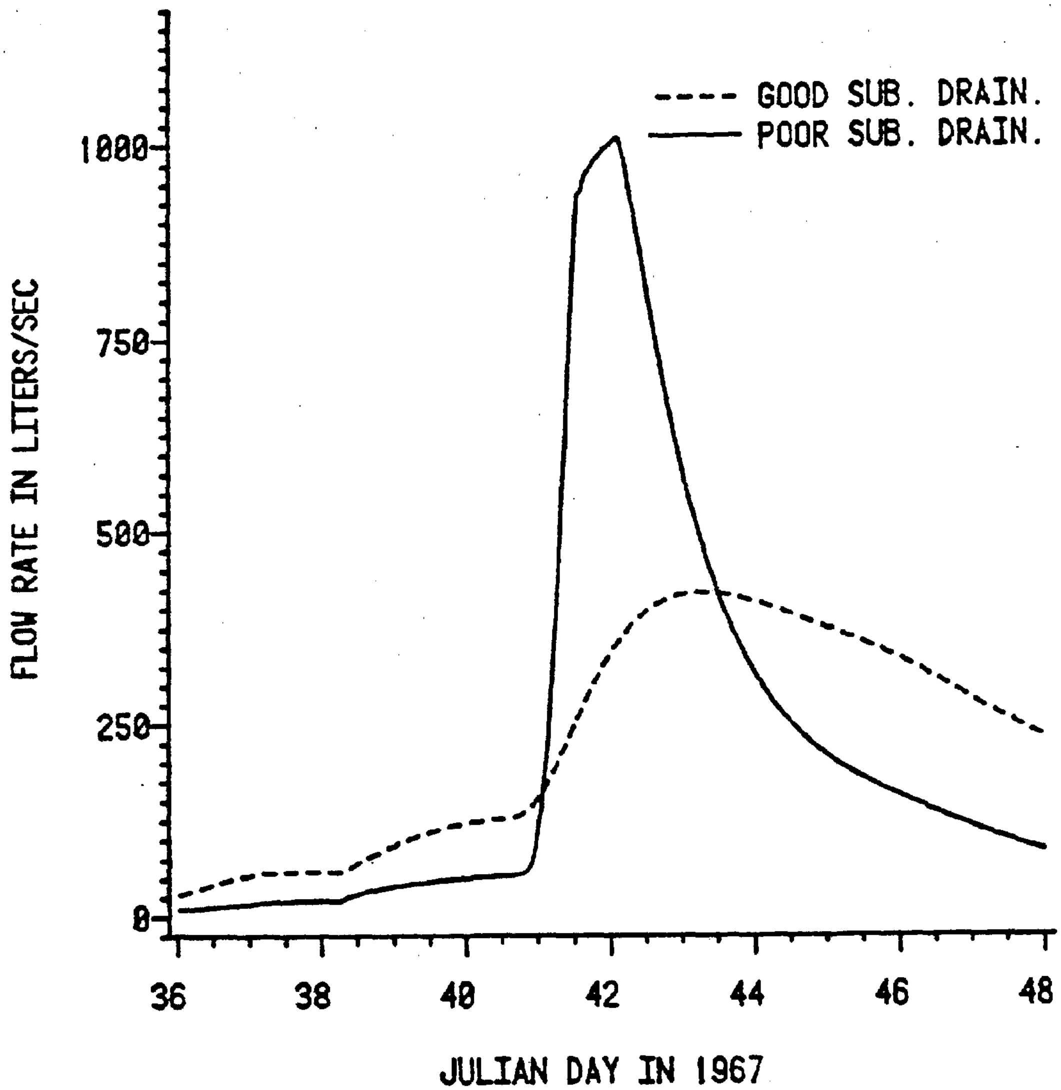

Figure 55. Good and poor subsurface drainage hydrograph comparison for post-mining for a 50 day return period flow. Eleven blocks (1430 ha) contribute flow to each hydrograph.

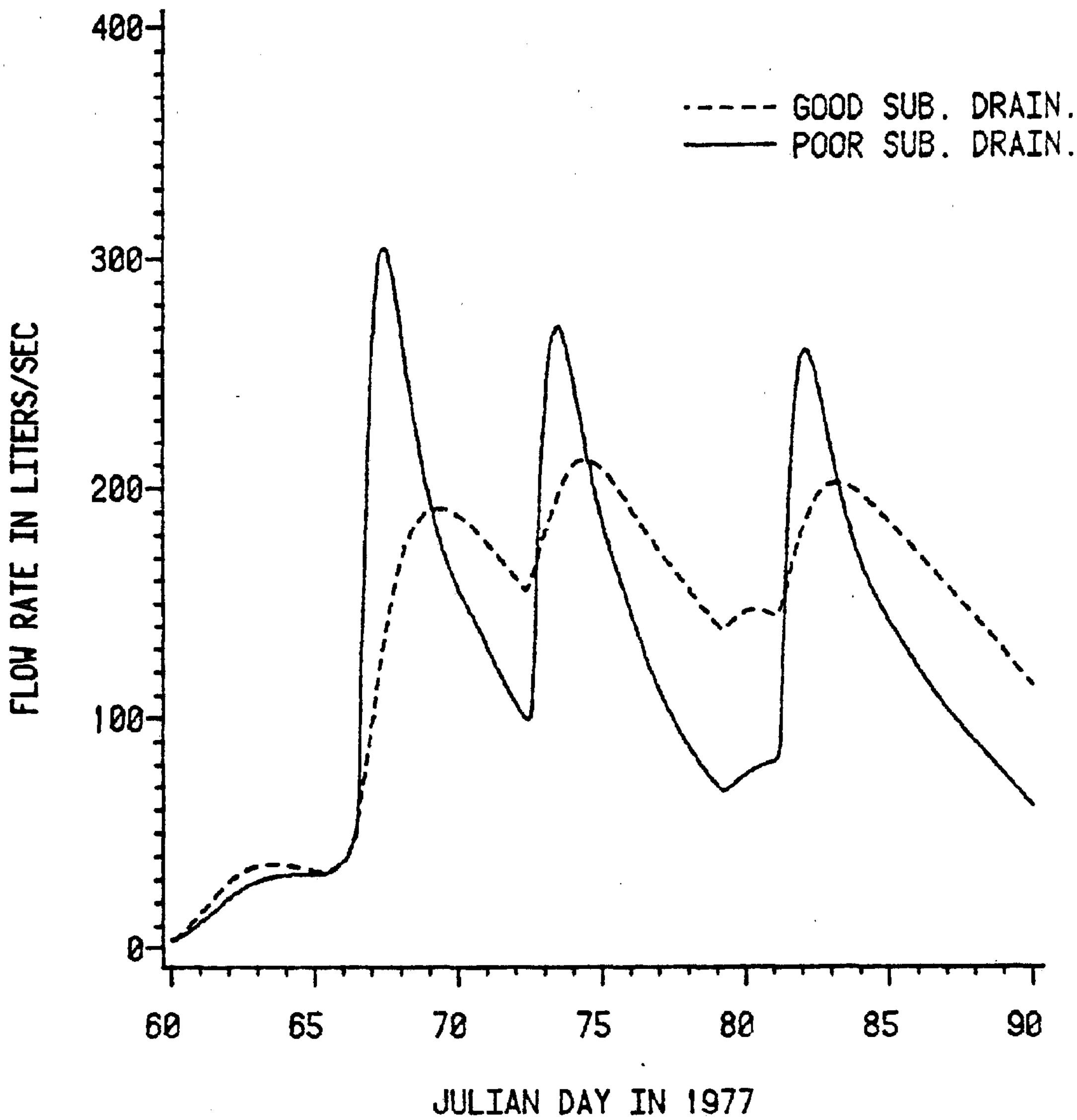

Figure 56. Good and poor subsurface drainage hydrograph comparison for the post-mining condition. Eleven blocks (1430 ha) contribute flow to each hydrograph.

Hydrologic Effects of Alternative Post-Mining Crop Rotations

Accurate, predictive analysis of the post-mining hydrology is dependent upon many assumptions. These assumptions include the soil layering and properties as well as the cropping strategy used. Previous sections have assumed a corn-winter wheat rotation. This section analyzes the potential effects of various cropping practices on the soil water balance and hence total runoff. Five different, commonly employed, crop rotations were selected. The assumed rooting depths throughout the year are tabulated in Table 21 for the five rotations. Rooting depths for days between those given in the table are determined in the model by linear interpolation. The effects of different crops are limited to the effects of their different rooting depths and thus the effects on evapotranspiration.

For the purposes of this analysis a particular annual rotation is assumed to occur on a continuous basis and then monthly averages are compared. In reality, a rotation may consist of several different crops extending over two or three years. For example a corn-wheat-soybean rotation would start out with a fallow condition from Jan. 1 to April 15 when corn is planted. Corn would be harvested by Sept. 15 and winter wheat planted Oct. 1. Wheat would be harvested May 15 of the next year and soybeans planted shortly thereafter. Soybeans would then be harvested by Nov. 1 and the land would usually be fallow until the following spring when corn would be planted again. The model treats only one year of the two year rotation. Therefore rooting depths for a corn-wheat rotation start with a fallow condition from Jan. 1 to April 15 of each year. Then corn rooting depths are assumed until harvest after which rooting depths for wheat are used until Jan. 1 when the sequence is repeated with fallow conditions-corn-wheat etc.

The average monthly flow rates per unit area for all five crop rotations are given in Figures 57 and 58. Figure 57 is for good subsurface drainage and Figure 58 is for poor subsurface drainage. The yearly average total water losses are given in Table 22 for the ten treatments. Good subsurface drainage increased the the total annual flow by an average of 3.42 cm/year (9.4%). For both subsurface drainage treatments a wheat-soybean rotation resulted in the least annual flows and the late corn-wheat rotation produced the highest annual flows. The differences in annual flow are mirrored by the differences in annual evapotranspiration (Figures 59 and 60 and Table 23). Winter differences in evapotranspiration (ET) are small as ET will nearly always equal potential ET (PET). For the corn only and wheat-soybean rotation in November and December, however, the ET was lower due to the lack of rooting depth. Periods of higher than average flow occurred prior to planting of late corn, between harvesting of wheat and planting of soybeans in the spring, between harvesting corn and planting wheat in the fall and for the corn only rotation in the winter months. Planting a

Table 21. Julian day (JULD) - rooting depth (ROOTD) relationship for the five different crop rotations in the form required as input to DRAINMOD, root depth in cm. [JULD(I),ROOTD(I),I=1,n].

1. Wheat - soybean rotation;

001	15.0	040	15.0	060	20.0	080	25.0	130	25.0	131	3.0
145	3.0	155	7.0	170	20.0	185	25.0	200	30.0	300	30.0
320	10.0	325	3.0	366	3.0						

2. Cover crop - corn - winter wheat rotation;

001	10.0	030	10.0	045	15.0	085	3.0	110	6.0	120	15.0
150	25.0	173	30.0	245	30.0	255	10.0	270	3.0	280	3.0
300	10.0	330	15.0	366	15.0						

3. Corn - winter wheat rotation;

001	3.0	070	3.0	110	6.0	120	15.0	150	25.0	173	30.0
245	30.0	255	10.0	270	3.0	280	3.0	300	10.0	330	15.0
366	15.0										

4. Corn only;

001	3.0	090	3.0	110	6.0	120	15.0	150	25.0	173	30.0
245	30.0	255	10.0	290	3.0	366	3.0				

5. Late corn - winter wheat rotation;

001	3.0	126	3.0	135	3.0	153	6.0	158	10.0	180	25.0
200	30.0	265	30.0	275	10.0	290	3.0	300	3.0	320	10.0
350	15.0	366	15.0								

cover crop over the winter had an effect similar to growing winter wheat in reducing total outflow.

The effects of the various rotations are dependent on the timing of crop growth with rainfall and PET. Over the long-term, average rainfall for May is about the same as the average monthly rainfall over the whole year (see Figure 61), while PET for May is relatively high. Thus the effects of varying rooting depths during May will be greater than during a high rainfall, low PET month (eg. October). However, the winter months, on average, exhibit higher flows and some form of cover crop over the winter months will help to make the most of what little ET there is. Water tables will be kept slightly deeper on average with a winter crop and thereby reduce average flows and possibly lower some of the peak flood flows.

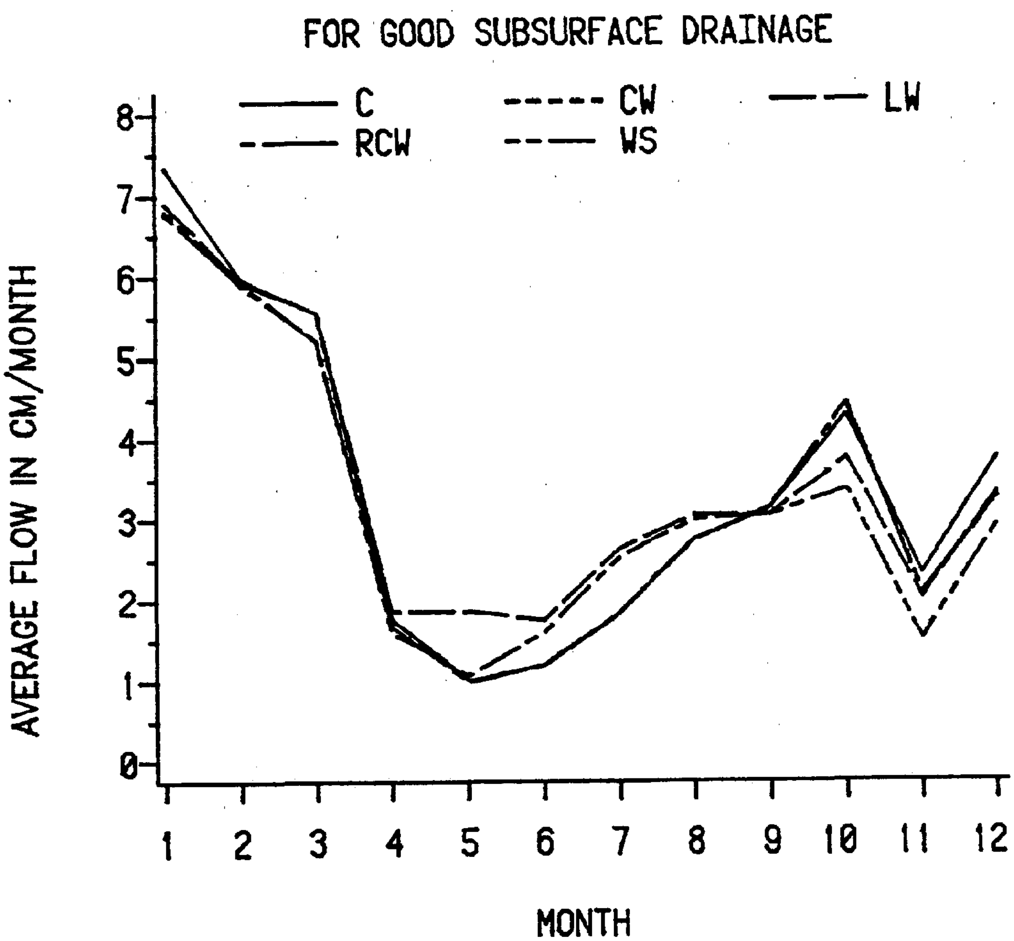

Figure 57. Average monthly flowrates per unit area for five different crop rotations for good subsurface drainage. Values are averages from 20 years of simulation. C is corn only, CW is corn-winter wheat, LW is late corn-winter wheat, RCW is cover crop-corn-winter wheat and WS is wheat-soybean rotation.

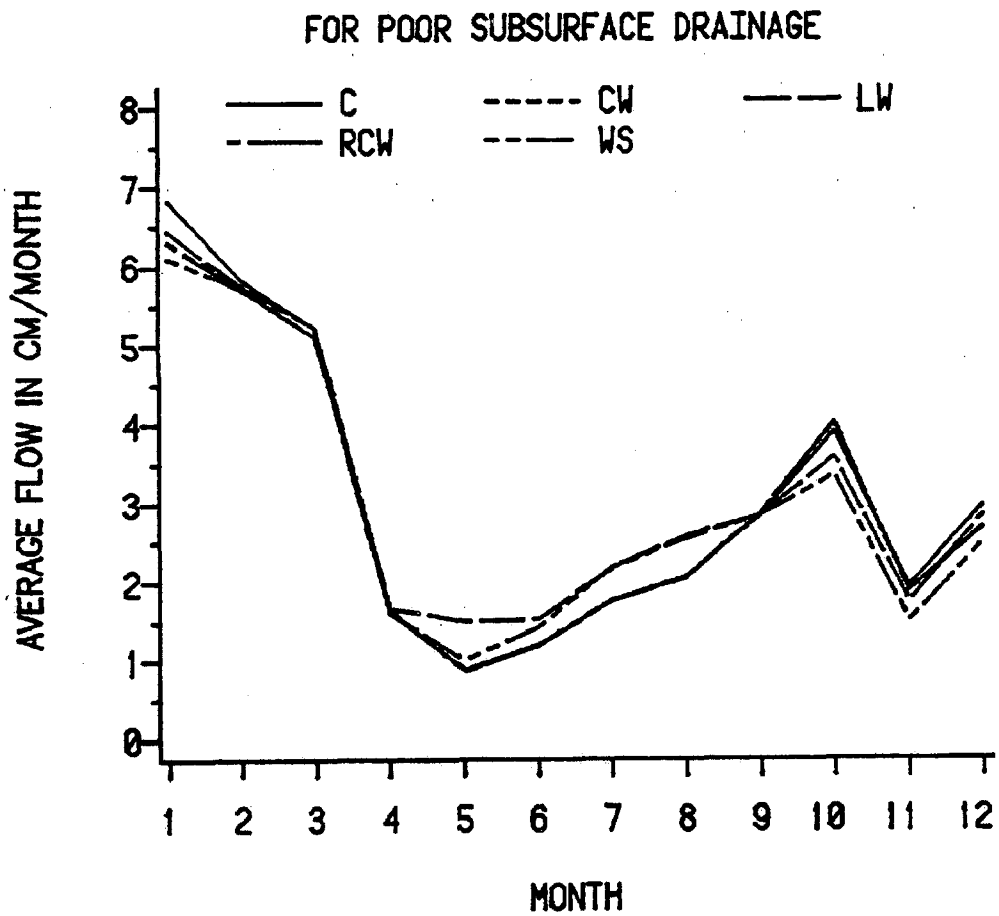

Figure 58. Average monthly flowrates per unit area for five different crop rotations for poor subsurface drainage. Values are averages from 20 years of simulation. See Figure 57 for key.

Table 22. Yearly average water loss for five different crop rotations and two different drainage treatments.

Rotation	Subsurface Drainage	Water Loss cm/year
Wheat-soybean	poor	35.8
	good	38.5
Cover crop-corn-wheat	poor	36.0
	good	39.2
Corn-wheat	poor	36.2
	good	39.7
Corn	poor	37.0
	good	40.9
Late corn-wheat	poor	37.8
	good	41.6

Table 23. Yearly average evapotranspiration for five different crop rotations and two different drainage treatments.

Rotation	Subsurface Drainage	ET cm/year
Wheat-soybean	poor	89.7
	good	87.2
Cover crop-corn-wheat	poor	89.5
	good	86.4
Corn-wheat	poor	89.3
	good	85.9
Corn	poor	88.6
	good	84.8
Late corn-wheat	poor	87.8
	good	84.1

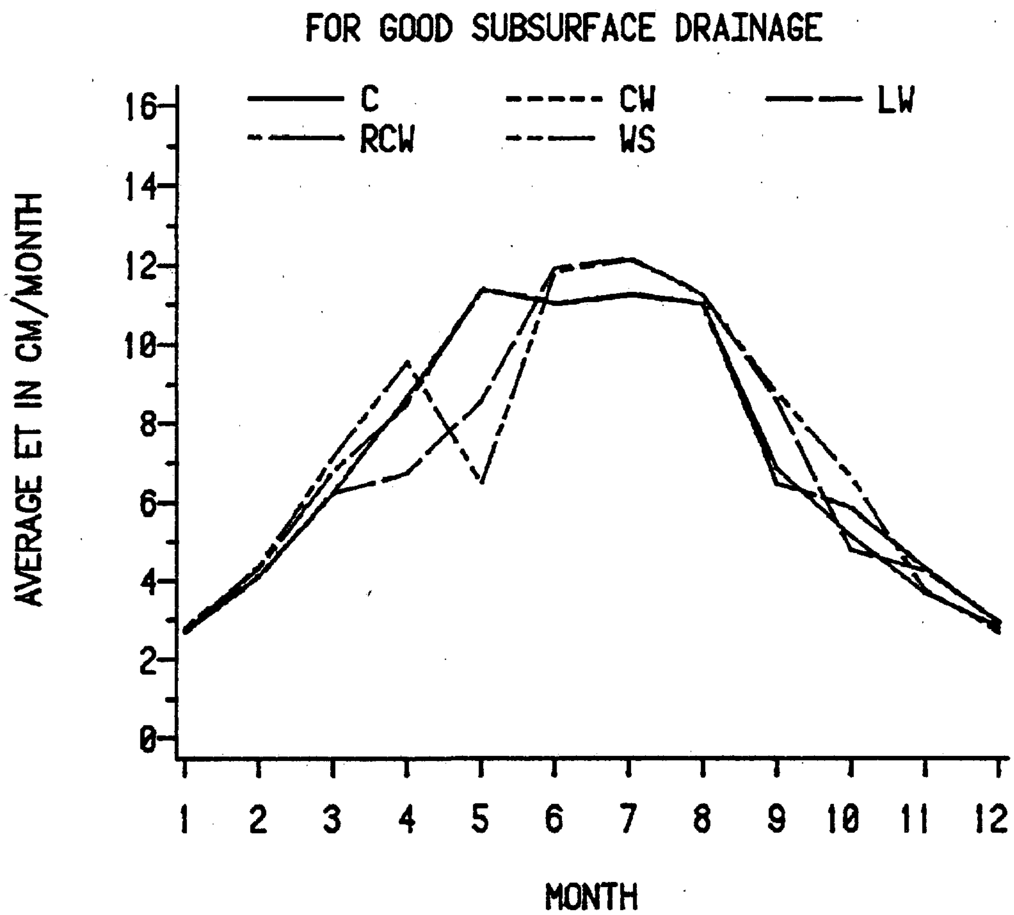

Figure 59. Average monthly evapotranspiration for five different crop rotations for the post-mining condition with good subsurface drainage. Values are averages from 20 years of simulation. See Figure 57 for key.

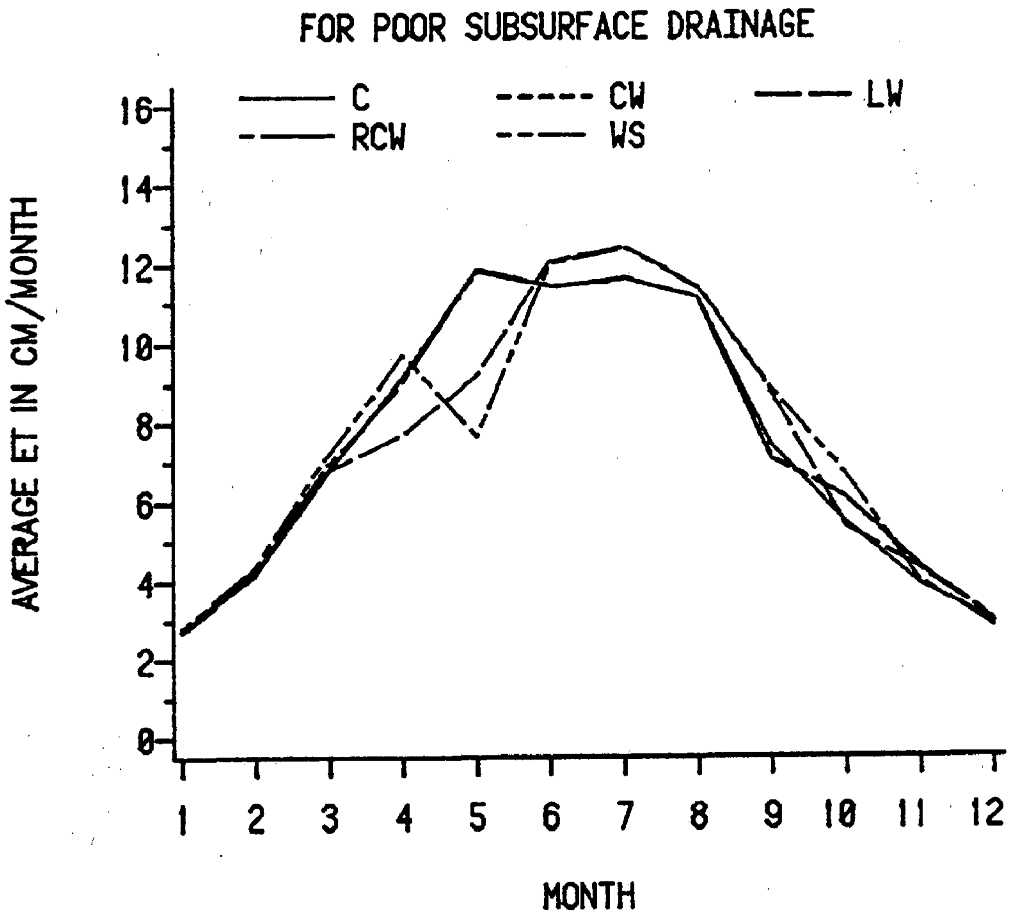

Figure 60. Average monthly evapotranspiration for five different crop rotations for the post-mining condition with poor subsurface drainage. Values are averages from 20 years of simulation. See Figure 57 for key.

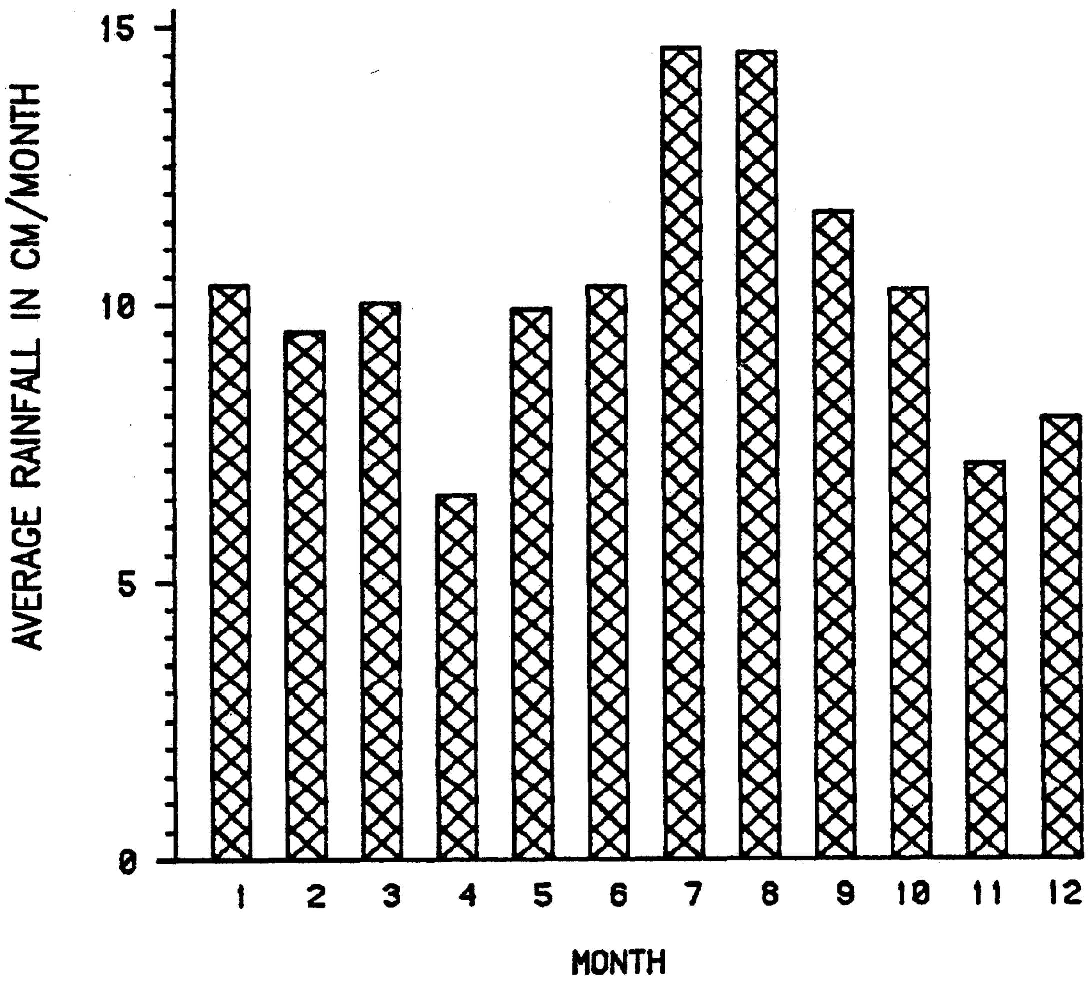

Figure 61. Average monthly rainfall from Elizabeth City. Values are 20 year averages.

Analysis of Hydrologic Effects During the Mining Process

From peat harvesting trials performed on the First Colony Farms permitted area over the last several years the following mining sequence has been proposed. Initially the vegetation is broken down and incorporated into the surface layer of peat using a large milling machine. The woody fraction of the vegetation is then available as fuel along with the peat. An extra field ditch is dug between the existing ditches, to increase surface and subsurface drainage, leaving a drain spacing of 50 m. The field is also sloped towards the ditches to promote surface runoff. It is proposed that six, 130 ha blocks will be under active mining at one time to keep the methanol plant supplied with peat. The area to be mined will be milled intensively to a depth of 10 cm (4") to break up any wood, and then left to dry for approximately one week. The pick-up procedure involves shallow milling, to fluff up and dry the surface few centimeters, followed by windrowing and then harvesting of what amounts to the surface 1 cm of peat dried to 40% moisture content. Weather permitting, shallow milling, windrowing and harvesting continues twice daily until the deep milled (10 cm) peat has been depleted. The deep mill is then repeated and the harvesting cycle begins again. Other operations such as root raking, rolling and wood shaving are performed as required. This procedure encourages maximum surface runoff with frequent milling to expose the wet peat and promote evaporation.

Input data for DRAINMOD to simulate the hydrology during mining are given in Table 24. The volume drained - water table depth relationship is given in Figure 24. The upward flux - water table depth relationship used is the same as for the before mining condition (Figure 23).

Simulations run for the active mining condition are primarily compared with the existing pre-mined condition simulations to show how hydrologic conditions might change as mining begins. Water table depth comparison for a year of average rainfall (1960) is given in Figure 62. The water table remains consistently higher during mining than before mining (by an average of 15.4 cm). Recall that the pre-mined condition assumed a constant root depth of 12 cm; a 4 cm root depth was assumed to simulate drying and removal of the surface layer during mining. With the vegetation removed, transpiration becomes zero and evaporation remains as the sole component of ET. Continual milling and exposure of wet peat could maintain high evaporation rates. However, as soon as the top few centimeters of peat have dried to the lower limit water content, subsequent evaporation will be inhibited. This effect will be most evident during the winter when no mining is planned and during very rapid drying conditions in summer when the harvesting cycle might not keep up with the drying rate. The important factor, so far as hydrology is concerned, is that the dry zone at the soil surface, which

Table 24. Summary of soil property and drainage system parameter inputs to DRAINMOD for during mining analysis.

INPUT	VALUE		
1. Soil properties.			
Depth to restricting layer	194 cm		
Saturated hydraulic conductivity, K	K = 70 cm/hr for depth < 4 cm K = 0.02 cm/hr 4 < depth < 104 cm K = 0.44 cm/hr 104 < depth < 194 cm		
Water content at lower limit for evaporation	0.45 cm^3/cm^3		
Saturated water content in root zone	0.74 cm^3/cm^3		
Average depth of evaporation (root) zone	4.0 cm		
Green-Ampt infiltration parameters;			
	Water table depth (cm)	A (cm^2/hr)	B (cm/hr)
	0	0	0
	10	0.23	5.0
	25	0.27	5.0
	50	0.34	5.0
	75	0.38	5.0
	100	0.42	5.0
	150	0.55	5.0
	500	0.55	5.0
2. Drainage system parameters (open ditches).			
Drain depth	100 cm		
Effective drain radius	19 cm		
Surface depressional storage	0.25 cm		
Drain spacing	50 m		

forms a reservoir for storage of subsequent rainfall never gets very deep. Once the top few cm have dried out, the surface layer

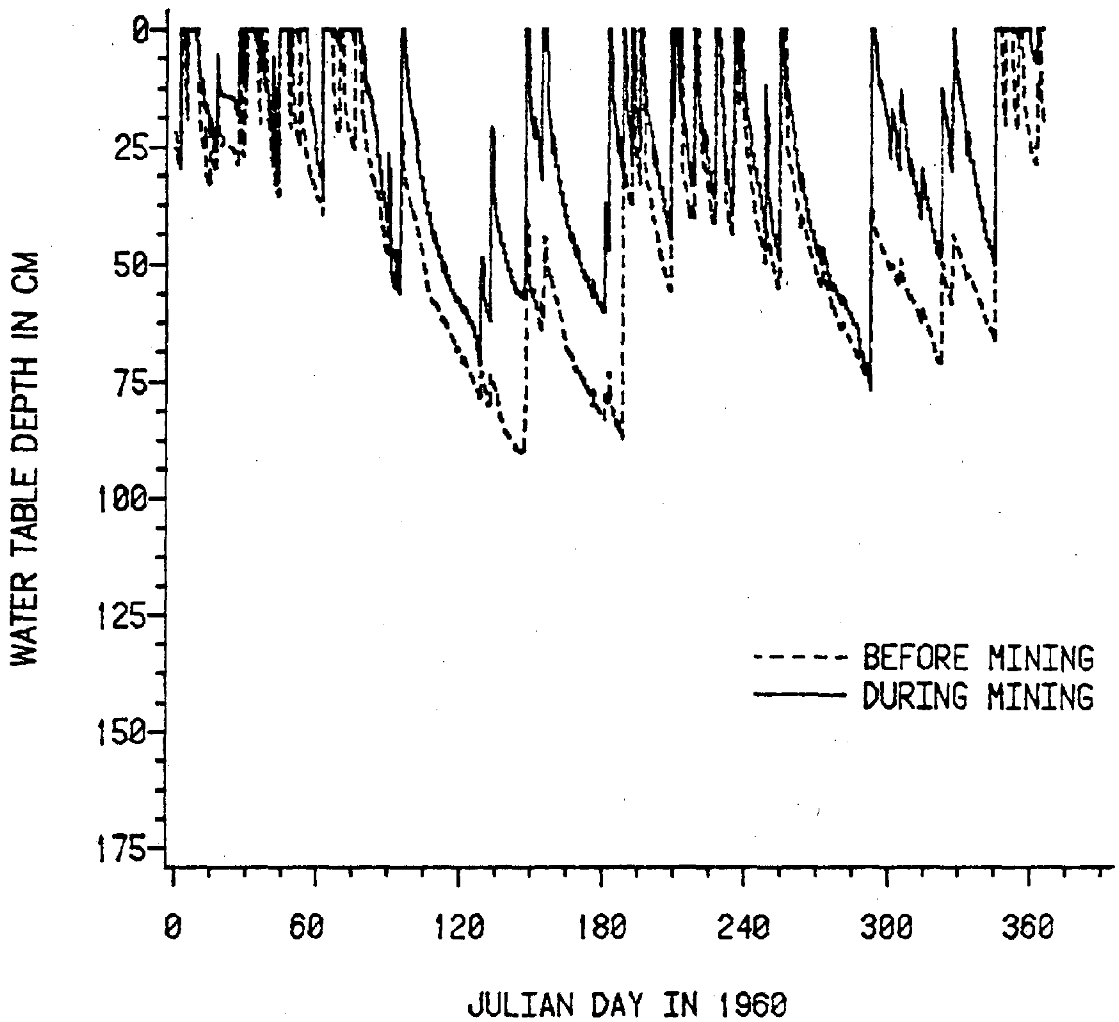

Figure 62. Before and during mining water table depth comparison for 1960.

is harvested allowing the layer underneath to begin to dry. This situation was simulated in the model by assuming an average surface depth of 4 cm which could be dried out to provide storage for infiltrating rainfall. When the dry zone extends to the 4 cm depth, the model predicts subsequent ET to be limited by the lack of soil water. Actually evaporation would probably not be limited because the surface layer would be removed by mining. However, this would have a similar effect on infiltration and runoff from subsequent rainfall events. The model assumes that storage is not available because ET is limited; whereas, storage is actually not available because the dry peat was removed (mined). Thus the actual ET from the mining site will be underestimated in the model, but the water table position relative to the mined surface as well as the subsurface drainage and the surface runoff should be accurately predicted. Because of the high water table relative to the mined surface, the surface and total runoff will be increased during mining over the before mining condition. This increase in total monthly flows is illustrated in Figure 63. The average predicted yearly water loss was 13.8 cm (29%) greater during mining than before mining.

Because of the cleared land surface and the high water tables, a much higher proportion of the total water loss during mining was surface runoff than for the pre-mining situation. Figure 64 illustrates the proportion of surface and subsurface drainage for the mined condition. On average the annual surface runoff was 44.4 cm and subsurface drainage was 16.9 cm during mining. This compares to an average annual surface runoff of 18.2 cm and subsurface drainage of 29.3 cm for the before mining case. Despite closer spacing of ditches the subsurface drainage during mining is less than before mining because of the much reduced drainage through the surface layer. It is expected that this reduction will not occur fully until approximately 15 cm of peat, containing milled vegetation and root material is mined. Figure 65 demonstrates the significant differences in surface runoff for before, during and after mining conditions. The after mining condition shown was for good subsurface drainage in a wheat-soybean rotation. Reducing the water that leaves the field as overland flow reduces peak flow rates as discussed in the previous sections.

Figure 66 shows the distribution of daily flows for the during mining compared to the before mining situation. For flows less than 10 l/s the during mining case exhibits slightly higher flow rates. This is because of higher average water tables and closer drain spacing increases the subsurface drainage. However, this difference is not great because of the very low conductivities and slow drainage rates. At flows between 10 and 60 l/s the pre-mining flow rates are higher than would occur during mining. This corresponds to a high water table and relatively rapid flow through the root zone for the pre-mining condition. With the assumed high conductivity for this layer, drainage rates with the water table between the surface and 25 cm deep will be higher in the pre-mining case. The most important differences are for flows in excess of 60 l/s. This is

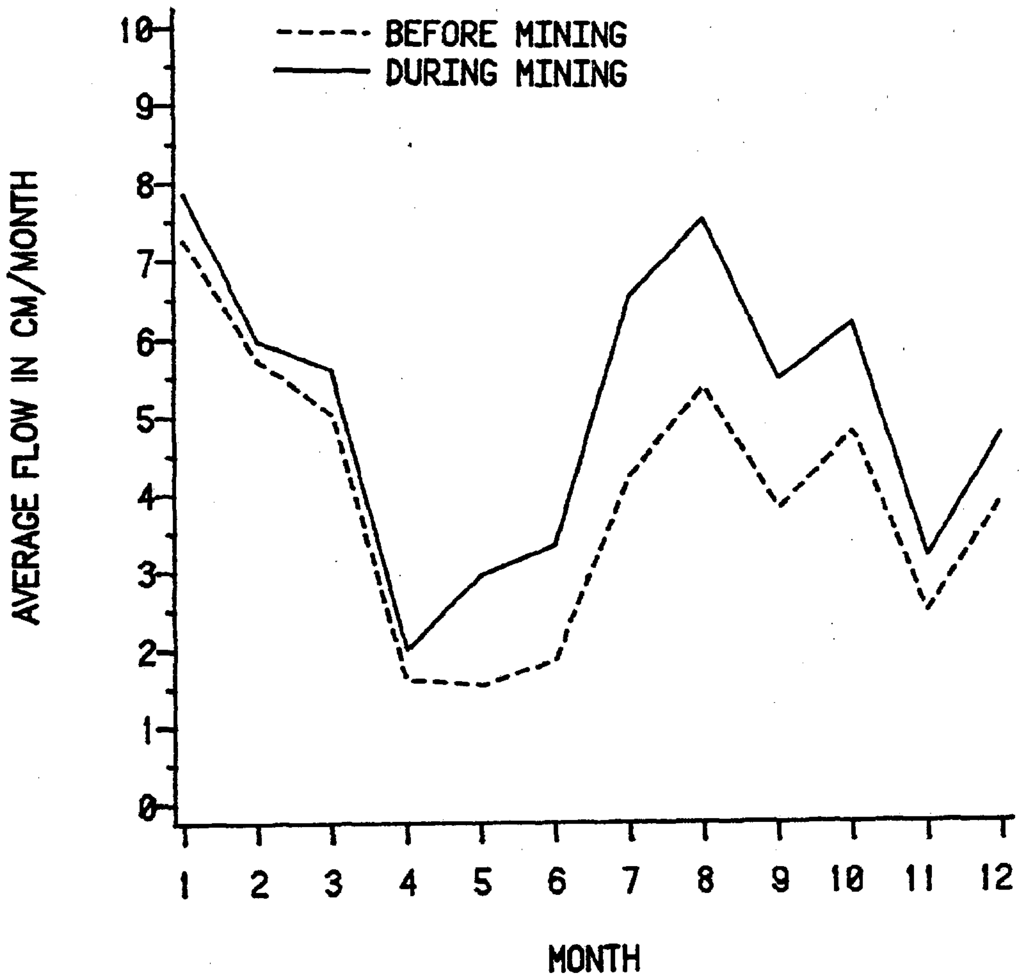

Figure 63. Before and during mining average monthly flow volumes per unit area. Values are monthly averages from 20 years of simulation.

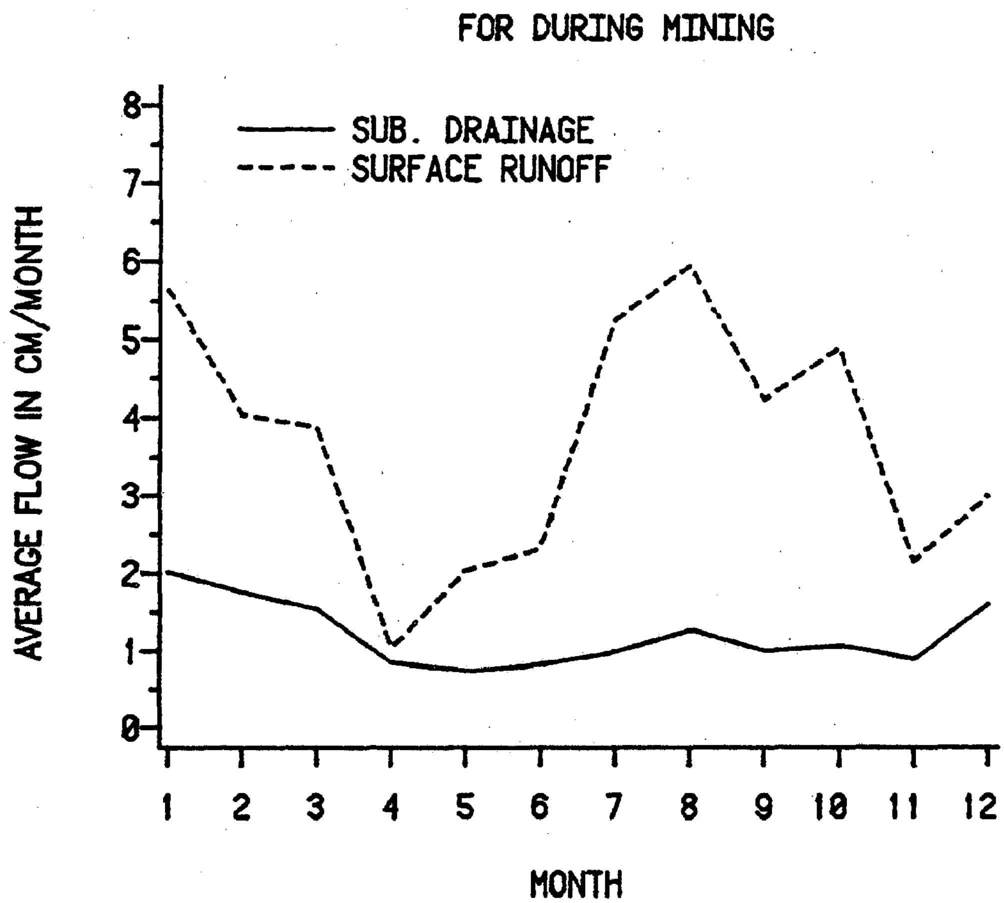

Figure 64. Comparison of subsurface drainage and surface runoff during mining. Values are monthly averages per unit area from 20 years of simulation.

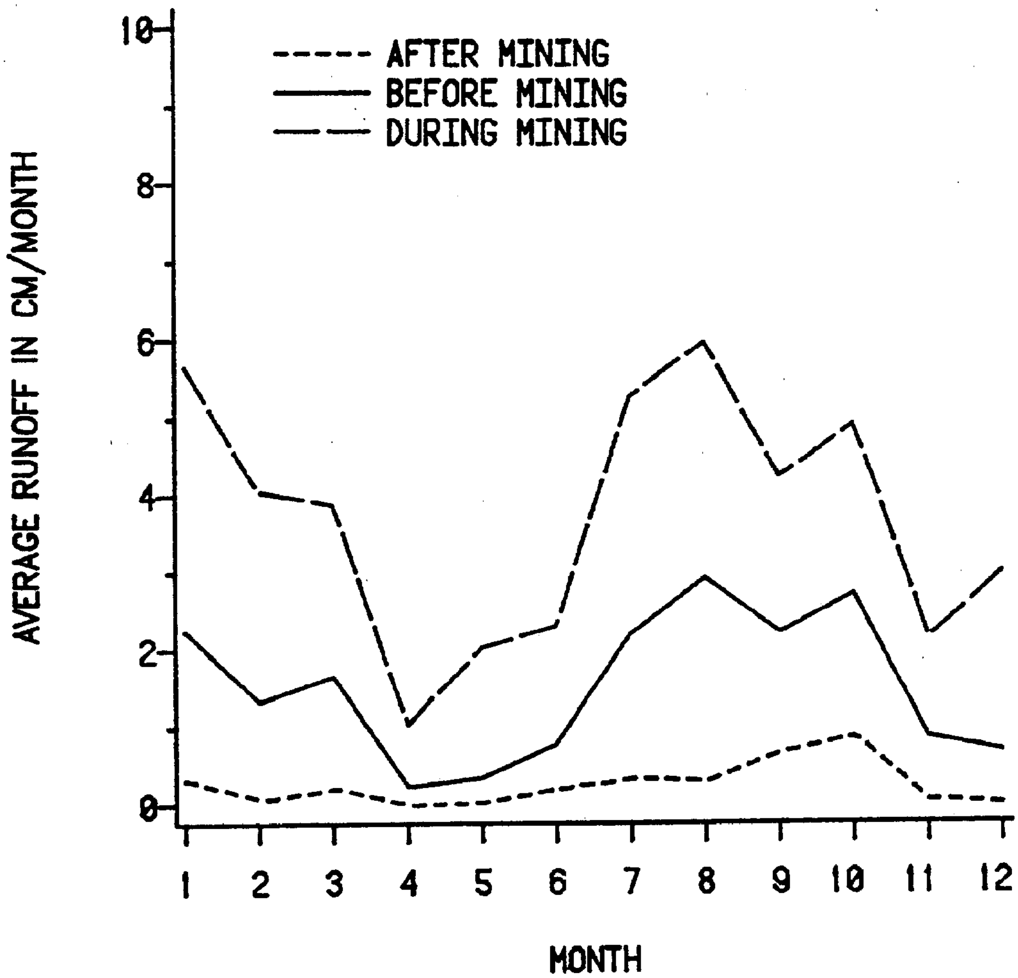

Figure 65. Average monthly surface runoff for before, during and after mining soil conditions. Values are averages from 20 years of simulation. The after mining condition was a wheat-soybean rotation with good subsurface drainage.

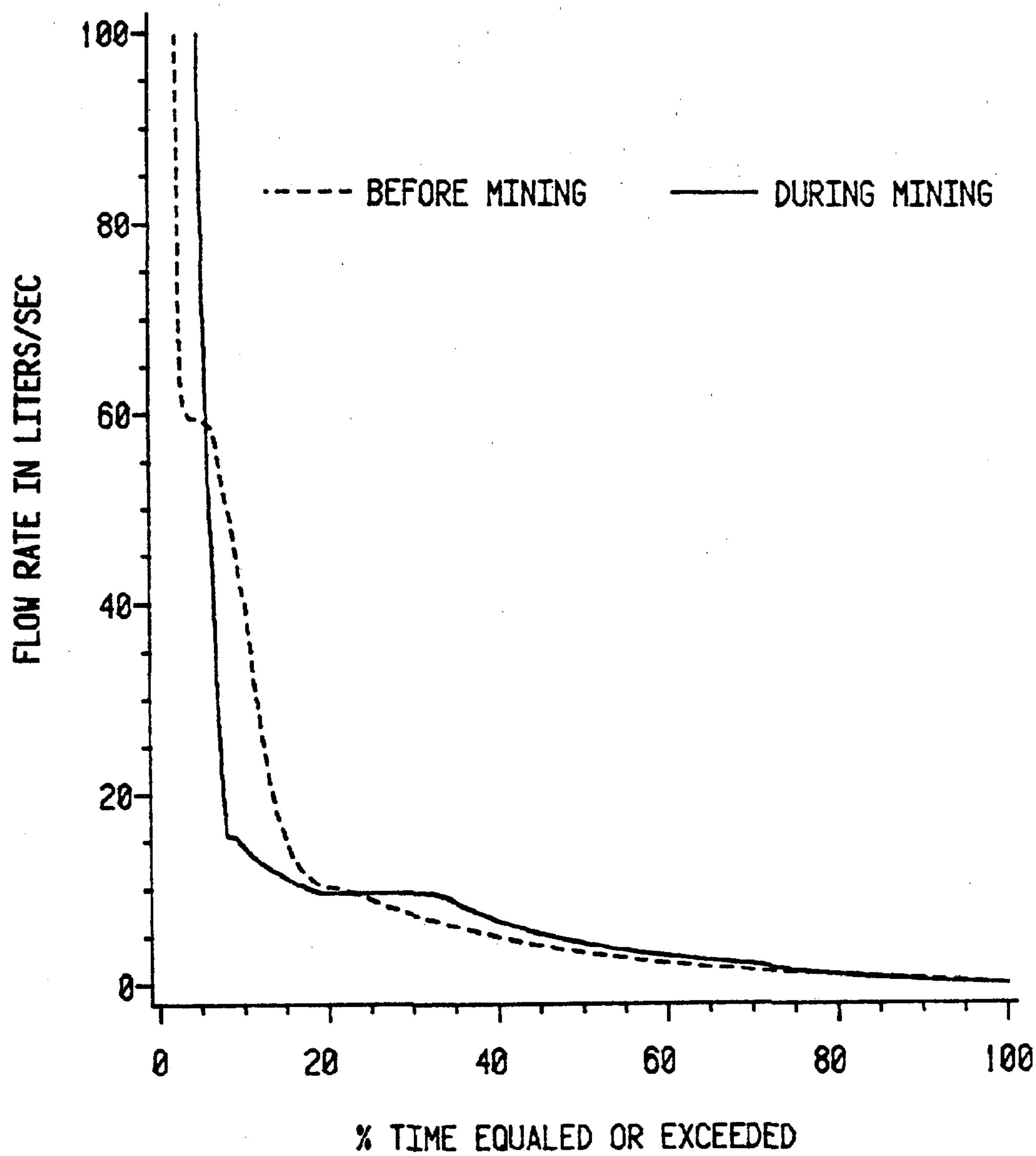

Figure 66. Comparison of flow duration curves for before and during mining. Values are average daily flows from DRAINMOD converted to collector canal outflows. The area contributing to flow is one block (130 ha). The vertical scale is truncated at 100 ℓ/s (see Figure 67 for points above 100 ℓ/s).

illustrated in Figure 67 on an expanded horizontal scale. For flow rates between 100 and 600 l/s for the active mining condition, increases in peak flood flows in excess of 50% can be expected over the pre-mining case. Note that this prediction is for collector canal outflows and these flows may be significantly moderated in the main canal (Figure 39). Also these high flows can be expected less than 6% of the time. The shallower water tables and reduced surface storage in the active mining case result in greatly increased overland flows both in frequency and magnitude.

These increased surface runoffs imply an increase in canal water turbidity as well as sediment collection in the canals. The increase in volumes of water loss, for individual storms as well as for over a whole year implies problems of increased flooding with canal banks being breached more often than before mining. However, the effects of mining an individual or series of blocks should be viewed in context with the whole system. A 50% increase in peak flow from a block may not be significant when merged with the flows from surrounding undisturbed or already reclaimed areas. This concept will be studied in the next section. It does indicate, however, that in order to minimize surface runoff and peak flood flows an area should be reclaimed as soon as is practical after harvesting is completed.

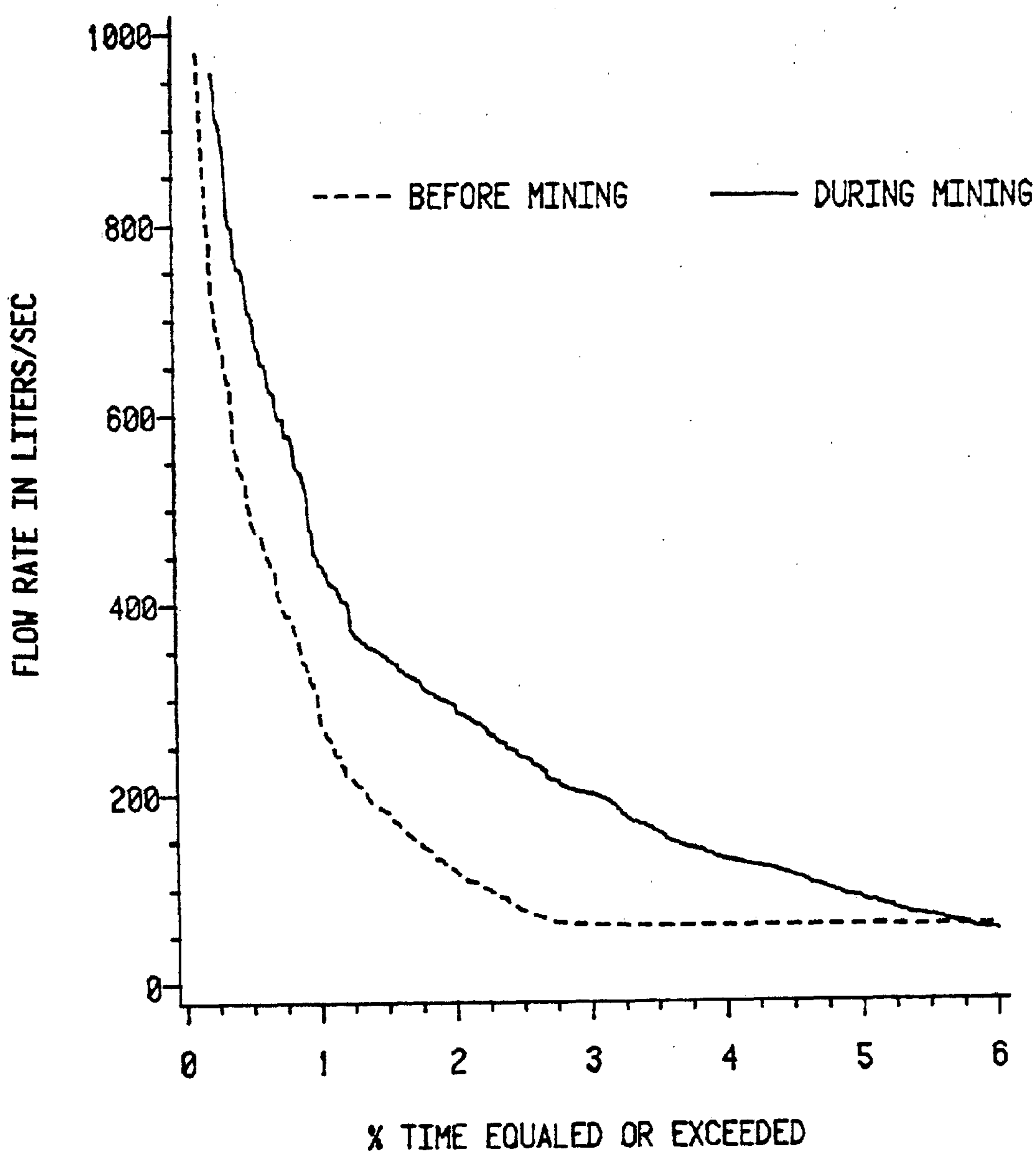

Figure 67. Comparison of flow duration curves for before and during mining. Values are average daily flowrates from DRAINMOD converted to collector canal outflows. The horizontal scale is limited to less than 6% time equaled or exceeded.

Analysis of Hydrologic Effects During the Mining Process

From peat harvesting trials performed on the First Colony Farms permitted area over the last several years the following mining sequence has been proposed. Initially the vegetation is broken down and incorporated into the surface layer of peat using a large milling machine. The woody fraction of the vegetation is then available as fuel along with the peat. An extra field ditch is dug between the existing ditches, to increase surface and subsurface drainage, leaving a drain spacing of 50 m. The field is also sloped towards the ditches to promote surface runoff. It is proposed that six, 130 ha blocks will be under active mining at one time to keep the methanol plant supplied with peat. The area to be mined will be milled intensively to a depth of 10 cm (4") to break up any wood, and then left to dry for approximately one week. The pick-up procedure involves shallow milling, to fluff up and dry the surface few centimeters, followed by windrowing and then harvesting of what amounts to the surface 1 cm of peat dried to 40% moisture content. Weather permitting, shallow milling, windrowing and harvesting continues twice daily until the deep milled (10 cm) peat has been depleted. The deep mill is then repeated and the harvesting cycle begins again. Other operations such as root raking, rolling and wood shaving are performed as required. This procedure encourages maximum surface runoff with frequent milling to expose the wet peat and promote evaporation.

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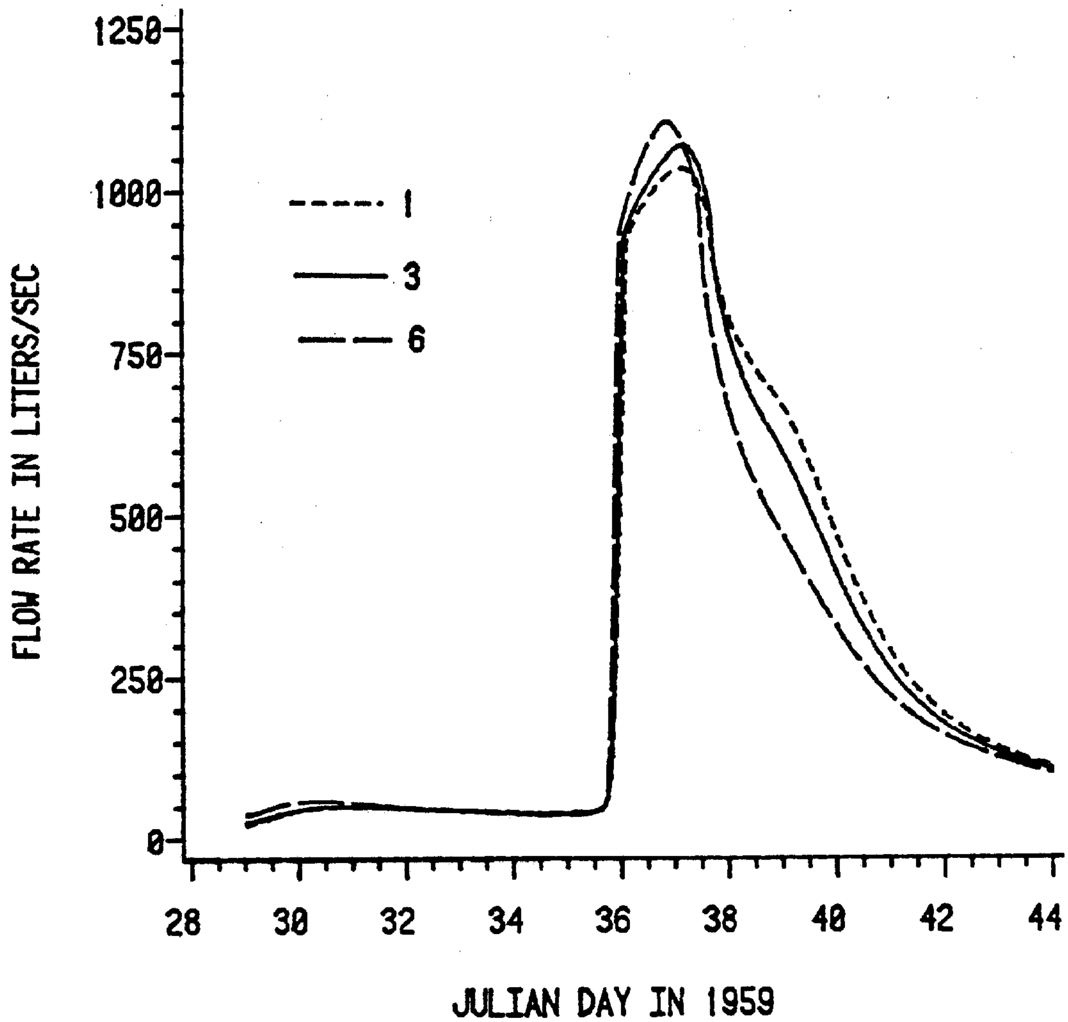

Figure 68. Hydrograph comparison of effects of one, three or six actively mined blocks contributing to flow in DeHoog Canal. All other blocks are in before mining condition. This is for a 75 day return period peak flow.

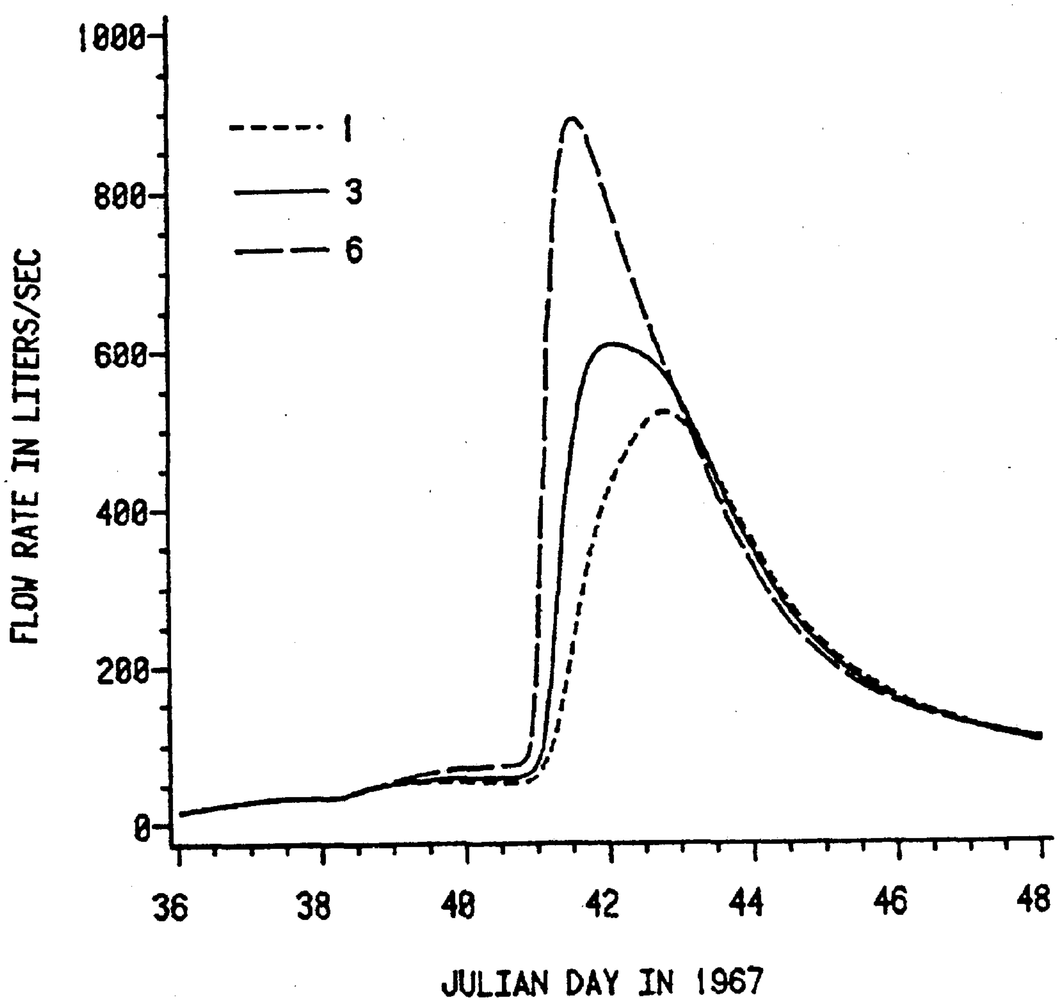

Figure 69. Hydrograph comparison of effects of one, three or six actively mined blocks contributing to flow in DeHoog Canal. All other blocks are in before mining condition. This is for a 50 day return period peak flow.

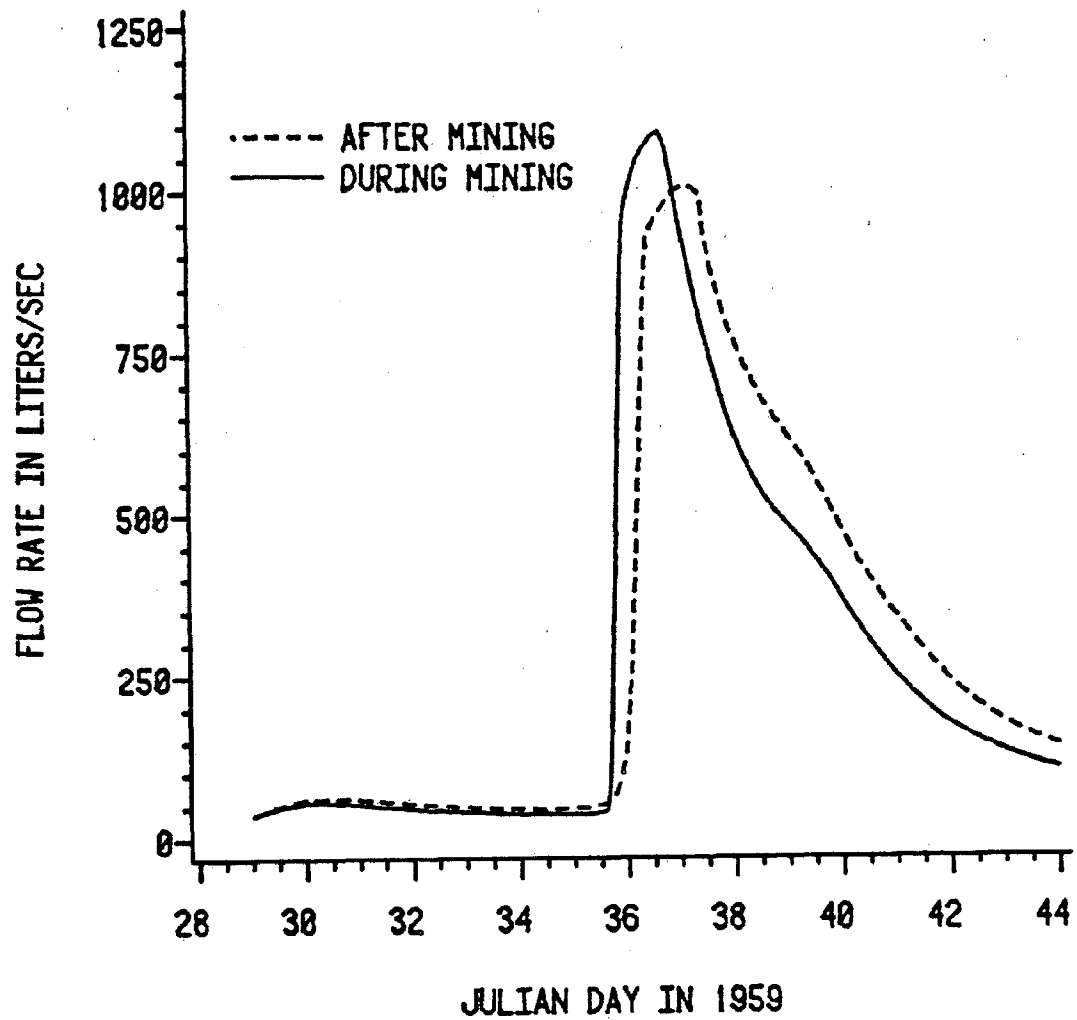

Figure 70. Comparison of DeHoog hydrographs between having six blocks in active mining condition and having the same six blocks in reclaimed condition with good subsurface drainage for a 75 day return period peak flood flow.

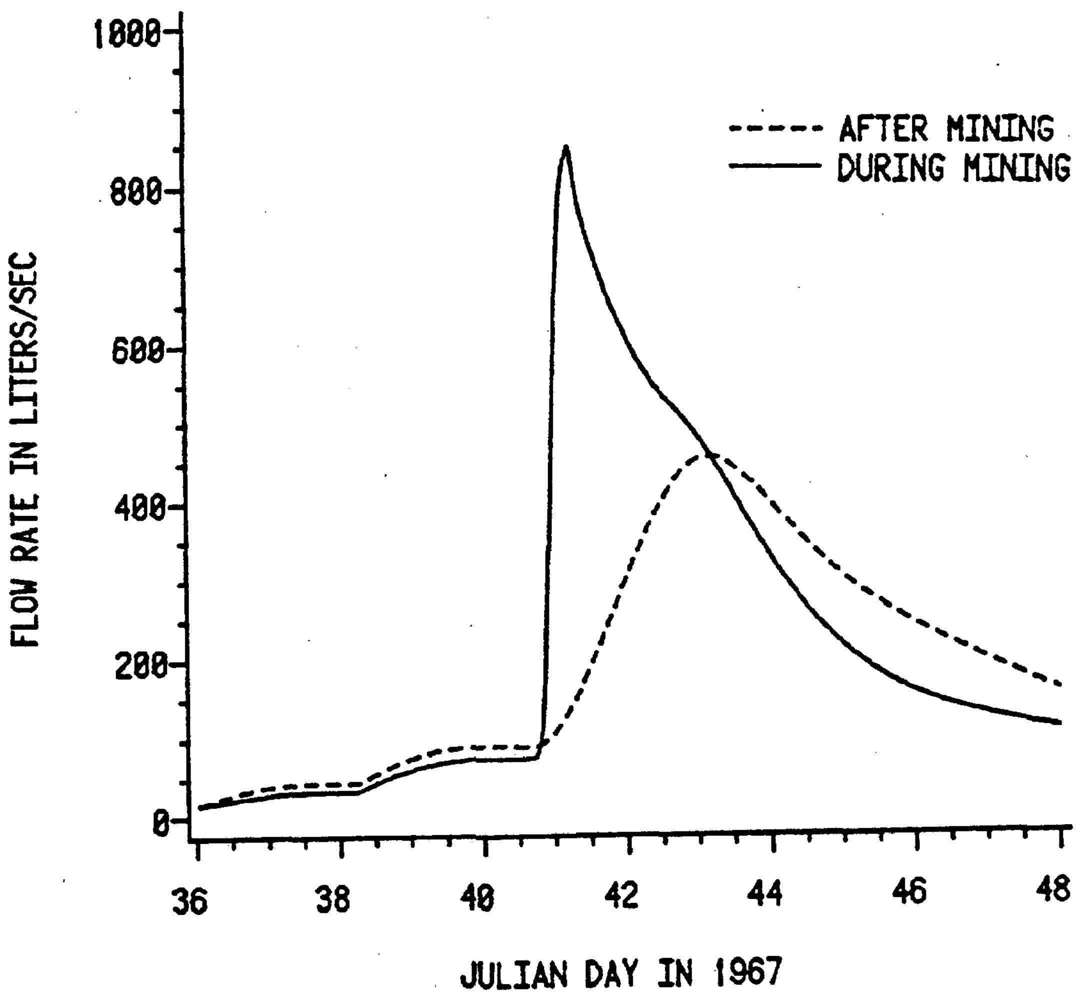

Figure 71. Comparison of DeHoog Canal hydrographs between having six blocks in active mining condition and having six blocks in reclaimed condition with good subsurface drainage for a 50 day return period peak flood flow.

RECOMMENDATIONS

Recommendations fall into three categories; recommendatons for model development and improvement, recommendations for future analyses using the models, and recommendations with respect to peat mining resulting from the studies presented in this report.

For the model, the major recommendation is for improvement in the computation of water storage in the canal banks and its subsequent release to the canals. With peat bogs in their undisturbed state this flow contribution may be well over fifty percent of the total subsurface drainage. As such its accurate representation is obviously important. The reported technique (empirical equations relating bank flow to an average water table depth derived from Boussinesq equation solutions), while a vast improvement over ignoring bank storage flow, has some shortcomings. The most noticable of these occurs immediately after a rainfall, on a deep water table, which is not sufficient to bring the water table to the surface. The model predicts an increase in bank flow dependent solely on the magnitude of the average water table rise. In reality a sudden, relatively large increase in flow will occur due to the filling up and following release of water from the bank storage reservoir which exists between the curved water table position before rainfall and a horizontal line from the point of maximum water table elevation and the canal.

At present in the models, all outfalls are considered to be unrestricted. The field ditches are assumed to exert negative pressure heads on the soil water equivalent to the depth of the ditch (i.e. the ditches are assumed to be empty of water at all times). The bank storage equations were based on a known set of circumstances and are not subject to this problem. Because the flood routing model predicts stage as well as flow rate it should be possible to link DRAINMOD to the flood routing model in such a way that ditch water levels could be passed from the flood routing model to DRAINMOD. Some iteration would be required but with small time steps this should not be prohibitive. Ideally, the various levels of flood routing should also be linked in order that the effects of any downstream obstruction (eg. pumping station, weir etc.) would be felt in the individual fields. Unfortunately, this recommendation is unlikely to be achieved with the models in their current form. Major revisions and possibly some simplifications would be required to keep the models within acceptable computation time limits.

For future modeling in peat bogs a better understanding of infiltration characteristics would be desirable. Infiltration properties of the surface layer of peat appear to change with wetting and drying. It would be very good to know the cause, extent and effects of these changes.

It is usual, for modeling purposes, to assume that a particular soil layer is impermeable when the hydraulic conductivity of that layer is less than one tenth of the hydraulic conductivity of the layer above. Because of the dramatic difference in hydraulic conductivities between the surface layer of peat containing cured peat as well as dead and decaying plant material and the saturated peat bog underneath this factor of ten guideline is easily exceeded. More work needs to be done to find out what effect this abrupt transition has on infiltration to the water table and on the movement of water in general.

It is planned to conduct simulations to observe the effects of forest plantings on the area hydrology. The current reclamation proposal for the First Colony Farms site include a 100 m strip of trees planted along each main canal running north-south and a 7 m wide strip along the collector canals running east-west. Apart from the obvious benefits towards erosion, shelter, wildlife refuge and future timber supply the transpiring capabilities of deep rooting trees may have an observable, favorable effect on the water balance and reduce total runoff.

There are a number of blocks on the FCF mining site which are essentially in a native condition. They are covered with scrub and small trees. The collector and main canals are in position but no field ditches have been dug. Total flow and water table measurements have been made on one of these sites during 1977 to 1979. The model could be tested on this site and then used to predict the long-term effects of mining and reclaiming these areas in general.

Arising from this study are a number of recommendations with regard to the actual mining procedure. The most important effect of mining is the increase in surface runoff from the sites in the active mining condition. In order to minimize the increases in canal flow rates, as few blocks as possible should be in the mining condition at one time. Also the mined blocks should be reclaimed as soon as practical upon completion of mining. It is expected that mining will continue from March to November each year. Blocks should not be left in the mining state over winter if possible. This could be prevented by timing reclamation to be undertaken in the fall.

If, during the initial mining phase, blocks continue to drain freely into the canals then as few actively mined blocks as possible should be allowed to discharge into the same canal. By spreading the increased runoff between several main canals downstream flooding will be reduced.

After mining is completed much can be done to control water loss from the fields. Installing a good subsurface drainage system will not only improve crop yields but will lower peak runoff rates for high rainfall events. A good subsurface drainage system will distribute water loss from a field more evenly over the year increasing the water storage capacity in the soil between rainfall events. Similar effects can be achieved with deeper rooting crops such as trees. Although such effects will be most pronounced over the summer months when potential evapotranspiration is highest, planting a cover crop over the winter months and having the ground fallow for as short a time as possible between crops will also be beneficial in increasing soil water storage and reducing peak flow rates.

SUMMARY

A water management computer model (DRAINMOD) was adapted for use on the First Colony Farms permitted peat mining area. The basis of the model is a water balance in the soil profile. Different model components evaluate various mechanisms of water movement and storage in the soil profile. The unit area water loss (surface and subsurface drainage) calculated by DRAINMOD was converted into average hourly flow rates from an eight hectare area. These flow rates were taken as lateral flow input to a numerical flood routing model. The flood routing model is based on the Saint-Venant equations of unsteady flows in open channels. The outflows from the field scale areas were routed down the collector canals. The collector canal outflows were, in turn, routed down the main canals to the watershed outlet.

DRAINMOD was used to observe the long-term hydrologic effects of mining the peat bogs and reclaiming them for agriculture on a per unit area basis. The DRAINMOD - flood routing model combination was used to determine actual canal flow rates at the mining area boundary for storms of varying magnitudes for a series of different land conditions.

The models were tested against field water table depths and collector canal flows collected on FCF from 1977 to 1979. The observed and simulated water table depths were in excellent agreement with an average daily deviation of 10.2 cm. The observed and simulated hydrographs of collector canal outflow were in good agreement for very low flows and for flows with the water table between 40 cm deep and the surface. The models appeared to overpredict flows when overland flow was a major contributor and the models tended to underpredict flows when a moderate to heavy rainfall event occurred when the water table was deep. Simulated daily and monthly average flow rates were in good agreement with observed data illustrating the reliability of the model's water balance.

Simulated comparisons of before and after mining hydrology revealed that the after mining hydrology will not be substantially different from before mining except that more hydrologic control will be available to the farmer. On average the water lost from the fields down the canals should be less after mining. Increased rooting depths from commonly used crop rotations would increase evapotranspiration and thus increase average water table depths and reduce average subsurface drainage. Adding good subsurface drainage to the reclaimed areas could increase base flow drainage rates and slightly increase total annual flow but would decrease peak flood flows.

During actual mining the volume of water entering the canals by overland flow will be substantially increased by an annual average of approximately 140% and total water lost to the canals

by subsurface as well as surface drainage will be increased by an annual average of about 29%. It is recommended that blocks be reclaimed as soon as possible after mining is completed.

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V. EFFECTS OF PEAT MINING ON GROUNDWATER PROCESSES

T. L. Foutz, R. W. Skaggs, R. G. Broadhead and R. B. Daniels*

INTRODUCTION

The effect of peat mining on groundwater processes, particularly flow to deep aquifers and flow to or from Lake Phelps, must be evaluated to determine the total impact of the proposed PMA project. The objectives of this study were to characterize the near surface sediments beneath the peat deposits and to evaluate the effects of peat mining on groundwater flow. First, field borings were made to determine the near surface stratigrapic sequence in the mining area. Bulk densities and hydraulic conductivities of various mineral units were measured. Next, the effects of mining and subsequent reclamation on groundwater processes in the mining site were analyzed by solving the Laplace equation for steady state flow conditions. A numerical finite element model was used to characterize groundwater movement for various mining scenarios and boundary conditions. The effects on subsurface flow to drainage canals, groundwater movement from Lake Phelps, and seepage to deep aquifers were studied. Results of these evaluations are presented herein.

PHYSICAL PROPERTIES OF MINERAL SEDIMENTS

Introduction

The sequence of near surface sediments, their lateral continuity, and the hydraulic conductivity of individual units had to be determined before the effects of peat mining on groundwater could be evaluated. A field study was conducted to determine the sequence and hydraulic conductivities of sediments within 17 m of the surface. Depths and conductivities of the deposits deeper than 17 m were determined from published and unpublished data obtained from the United States Geological Survey (U.S.G.S.).

Very few studies have been done on the classification and conductivities of the organic soils and underlying mineral layers in eastern North Carolina. Badr (1980) determined bulk densities, hydraulic conductivities, and other properties of the organic layers and underlying layers within about 2 meters of the surface for two peat soils in Washington and Tyrrell counties. Skaggs _et al_. (1980) described the results of borings to a depth of 13 meters on two soil profiles in the mining site south of Lake Phelps. Examination of their results shows that sandy loam

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layers could be expected under the peat on the west side of First Colony Farms, while silty clay loam to sandy clay loam can be expected on the east side. Miller (1980) studied the sediments exposed in the Stetson Borrow pit in Dare County. The stratigraphic units were classified and studies made on the physical and biogenetic structures.

The proposed mining site in Washington County is in the Swanquarter geological area and occupies a broad, flat, eastward-sloping plain representing an old ocean or estuary floor. Deep sediments are marine units with limestones, sand, and unconsolidated shell beds. The Swanquarter region has four major aquifer systems: Cretaceous sands; Eocene limestone; Miocene sands, marl and shell beds; and post Miocene age sand, marl, and shell beds (Nelson 1964). The more common names of these aquifers are Castle Hayne, Pungo River, Yorktown, and Quaternary deposits.

Very few hydraulic conductivity data are available for the deposits in the area of the proposed mining site as well as the deposits in the entire peninsula. Badr's (1980) saturated hydraulic conductivity values were determined in the laboratory using cores from the peat layers and from the mineral layers below the shallow peat. The transmissibilities of the four major aquifers have been estimated by the United States Geological Survey and are available in unpublished reports (Ronald W. Coble, personal communication, May, 1982).

Procedures

Field Borings and Section Descriptions

Borings were made to a depth of 17.8 m at 22 locations on the peat mining site to describe the sedimentary sequence and to measure the properties of individual units (Figure 72). The hole locations were originally arranged in an uniform grid pattern but as the study proceeded, the scheme was modified to accommodate conditions in adjacent areas. A Mobile B-30 drill rig with a 114 mm diameter bit was used for the borings.

The depth, thickness, texture, color, consistence, plasticity, cementation, and other features of each unit were recorded. Complete field descriptions of the profile for each of the bore holes are given by Foutz (1983). Samples were taken from each unit for bulk density and particle size determination. Six units that were continuous between a number of grid points were considered to have the major influence on groundwater movement. For each of these six units in situ hydraulic conductivity tests were performed at a minimum of three sites, and laboratory constant head tests were performed on core samples taken from two locations.

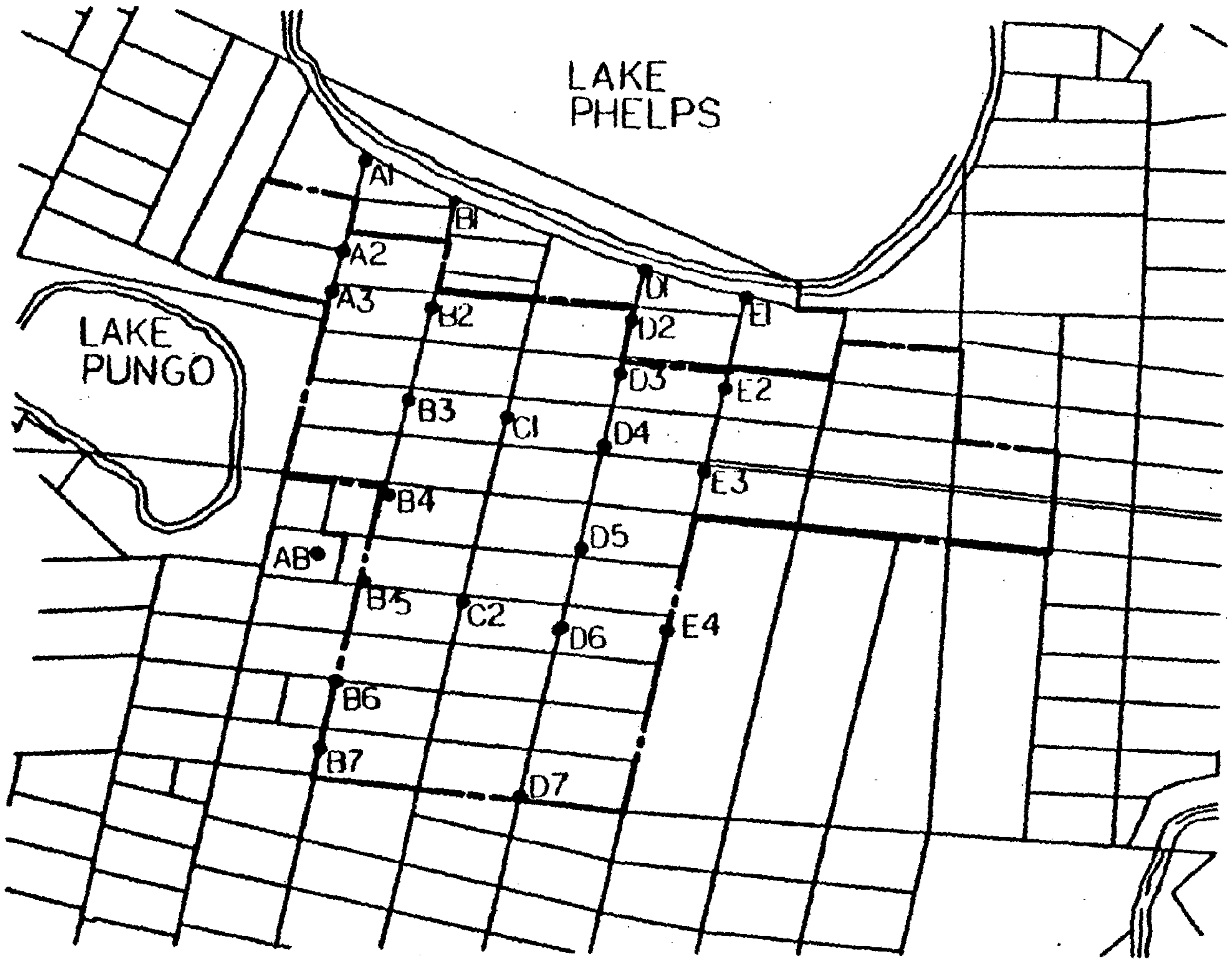


Figure 72. Location of field borings along Allen, Boerema, Clayton, DeHoog, and Evans Roads.

Soil Sampling

Samples of the mineral sediments below the peat deposits were obtained by two methods. First, samples were taken off the augers when initial borings were made. Samples from each of the layers were placed in plastic bags, sealed, and coded according to the hole site and depth. Second, samples were obtained by the use of the vibracore method as described by Laneskey et al. (1979). A 102 mm hole was drilled to the top of the mineral layer to be sampled. The tubing was lowered to the bottom of the hole and then vibrated into the layer until penetration stopped. The top of the tubing was sealed and the tubing with the sample was pulled from the hole.

Bulk densities were determined by cutting the cores obtained with the vibracore apparatus into sections about 100 mm long, measuring the volume, and determining their dry weight. Particle size distribution for each of the major profile layers was determined from hydrometer and sieve analyses as described by Day (1965).

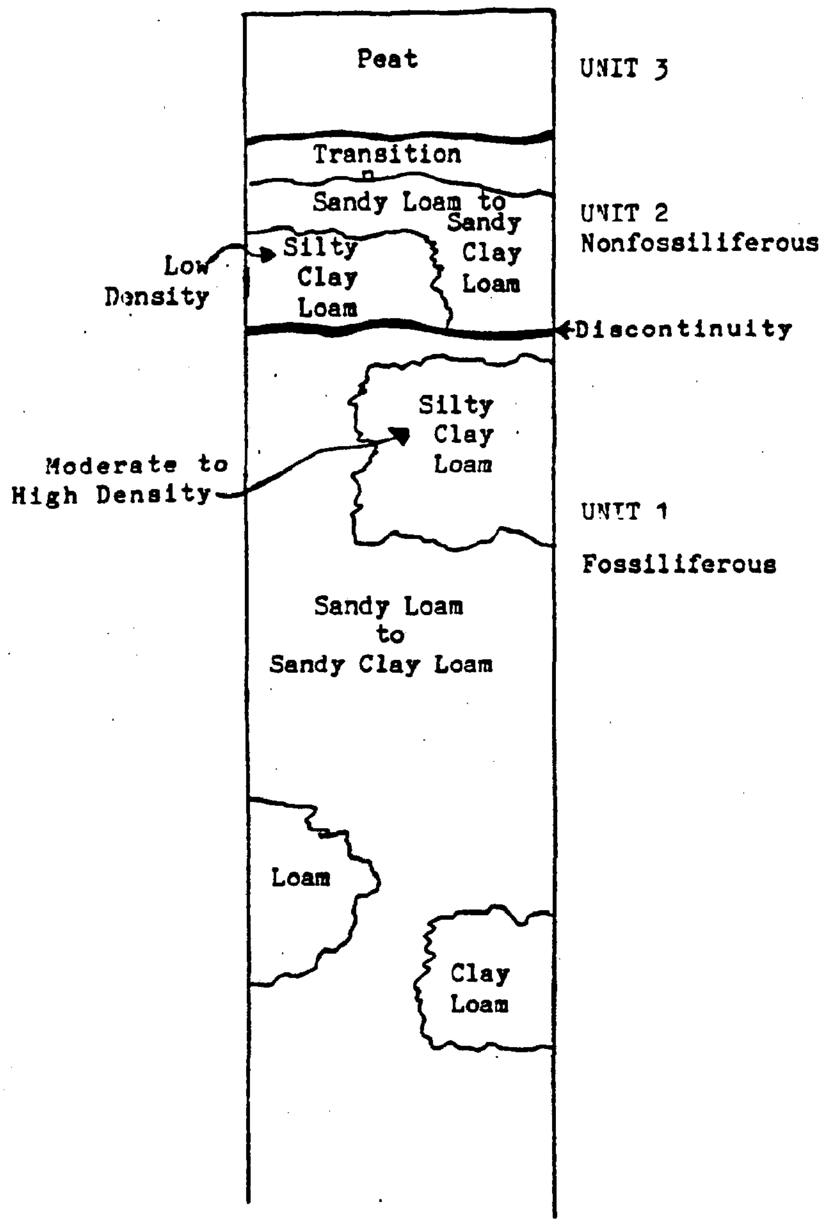
Saturated Hydraulic Conductivity

Two methods were used to measure the saturated hydraulic conductivity, K. The piezometer method, as described by Luthin and Kirkham (1968), was used to measure in situ K. The vibracoring method was used to install a 102 mm pipe into the layer which was to be measured. The soil was removed from inside the pipe and the water level allowed to rise to an equilibrium position. Then the water level was quickly lowered by bailing, and the rate of water table rise was measured and used to calculate the hydraulic conductivity. At least three pipes were installed and hydraulic conductivity measurements made in each of the six major near surface units identified in the area. Saturated hydraulic conductivities were also measured in the laboratory from the vibracore samples by using the constant head procedure as described by Klute (1965).

Results and Discussion

The results of the soil borings showed that the stratigraphy could be generally described as shown in Figure 73. Sediment sequences for the five north to south transects (Figure 72) are shown graphically in Figures 74-79.

The bulk densities for the six continuous layers are given in Table 25. The bulk density for soils of similar texture may vary widely as shown by the standard deviation values in this table. This variance may also explain, or at least it is



Unit 1-Beds of Sandy Loam to Sandy Clay Loam
and Silty Clay Loam to Clay
Embedded shell layers

Unit 2-Sandy Loam to Sandy Clay Loam to Silty
Clay Loam
Surface Layer

Unit 3-Peat Deposit

Figure 73. General description of the sequence of the mineral sediments below the peat deposits.

	KEY	
SYMBOL	NAME	ABBREVIATION
	ROAD FILL	—
	ORGANIC	O
	MEDIUM-FINE SAND	MFS
	SILTY CLAY LOAM	SCL
	CLAY LOAM	CL
	SANDY LOAM	SL
	SANDY CLAY LOAM	SCL
	SHELLS	- SL
	CLAY	C
	LOAMY SAND	LS
	TRANSITION	TR
	LOAM	L
	BASE OF HOLE	—

O - LOCATION NUMBER

Figure 74. Symbols used to describe the profile along Allen, Boerema, Clayton, DeHoog, and Evans Roads (see Figures 75 to 79).

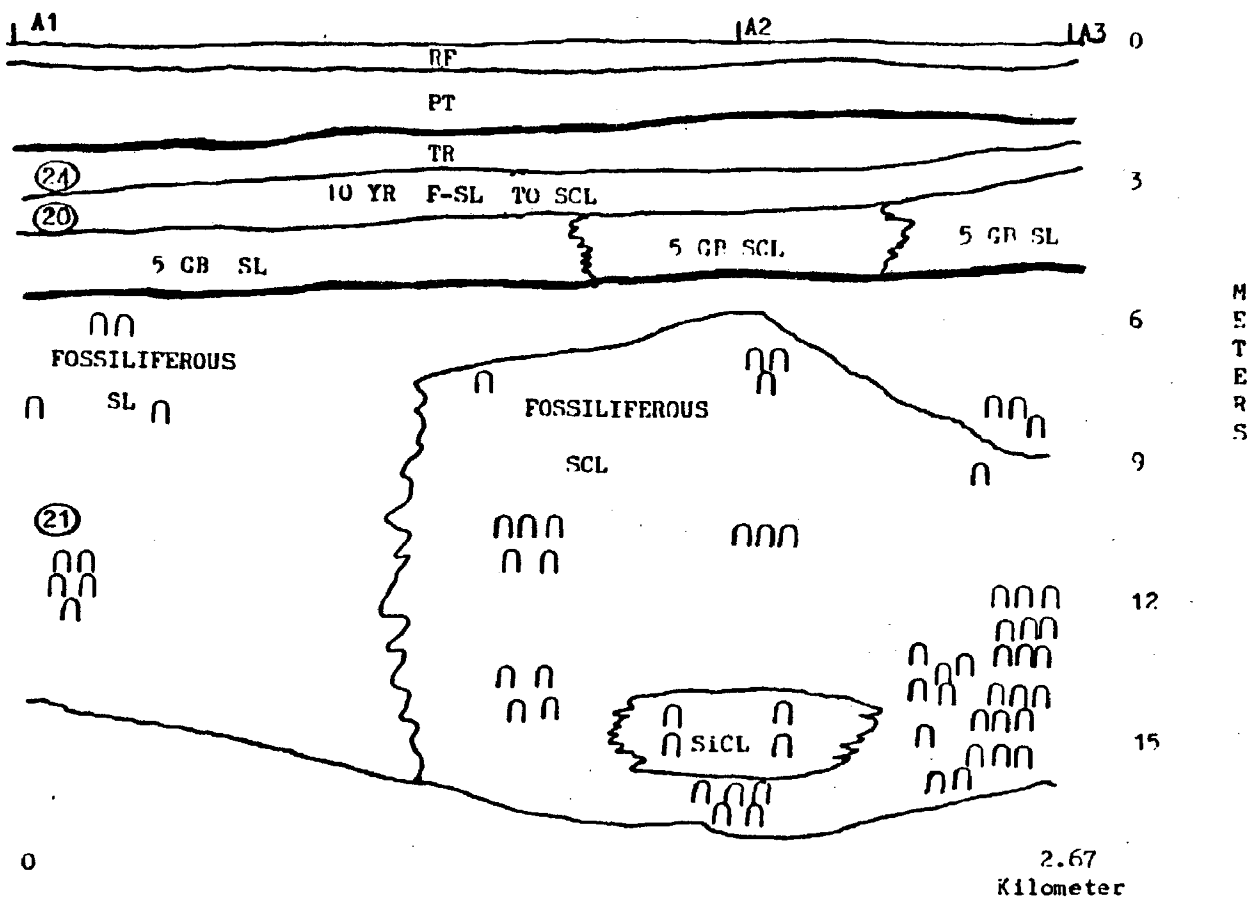


Figure 75. Description of the profile along Allen Road (refer to Figures 72 and 74).

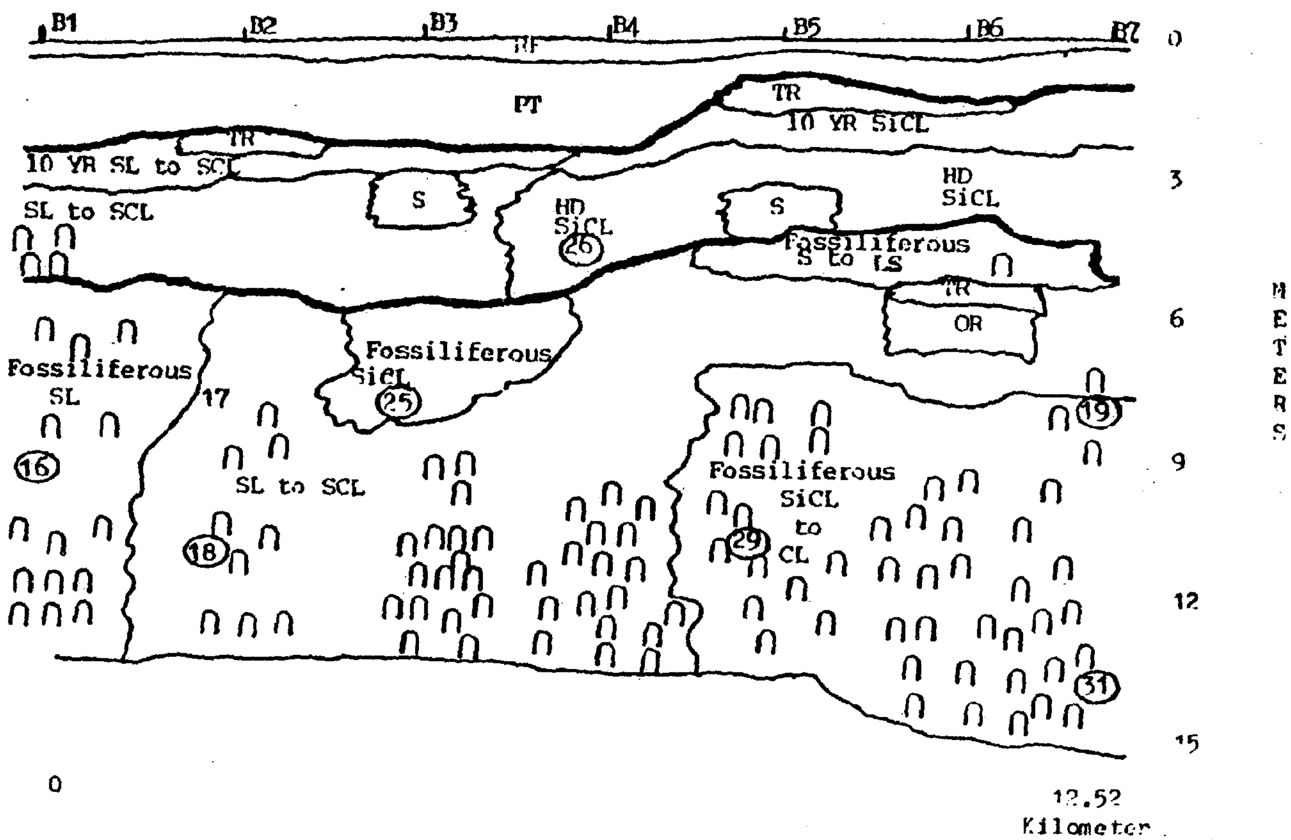


Figure 76. Description of the profile along Boerema Road (refer to Figures 72 and 74).

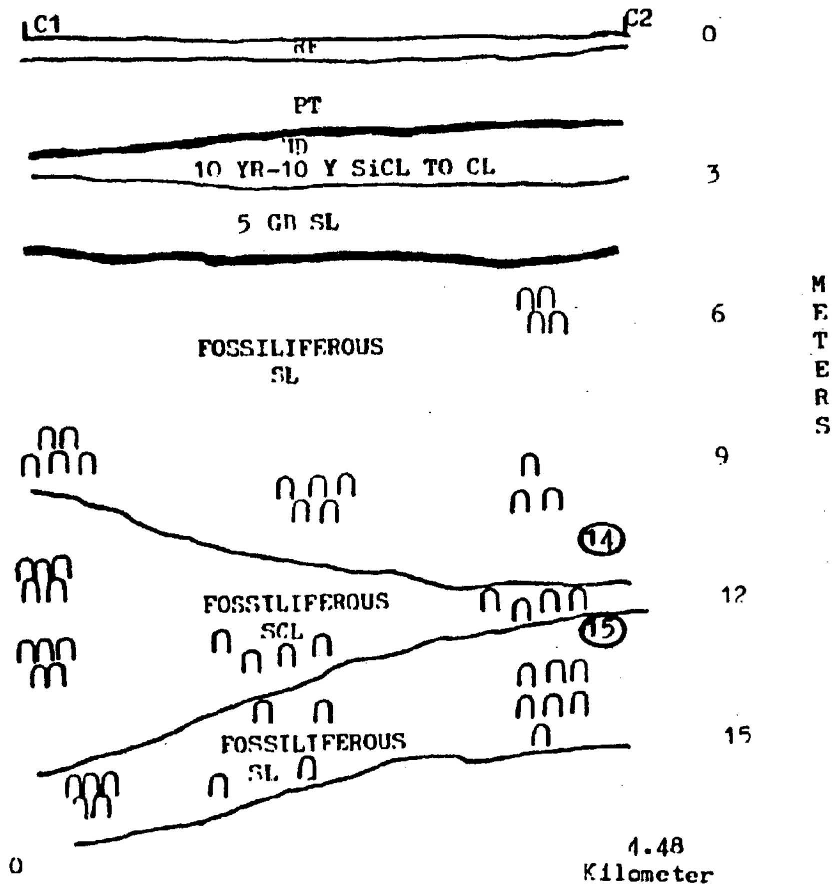


Figure 77. Description of the profile along Clayton Road (refer to Figures 72 and 74).

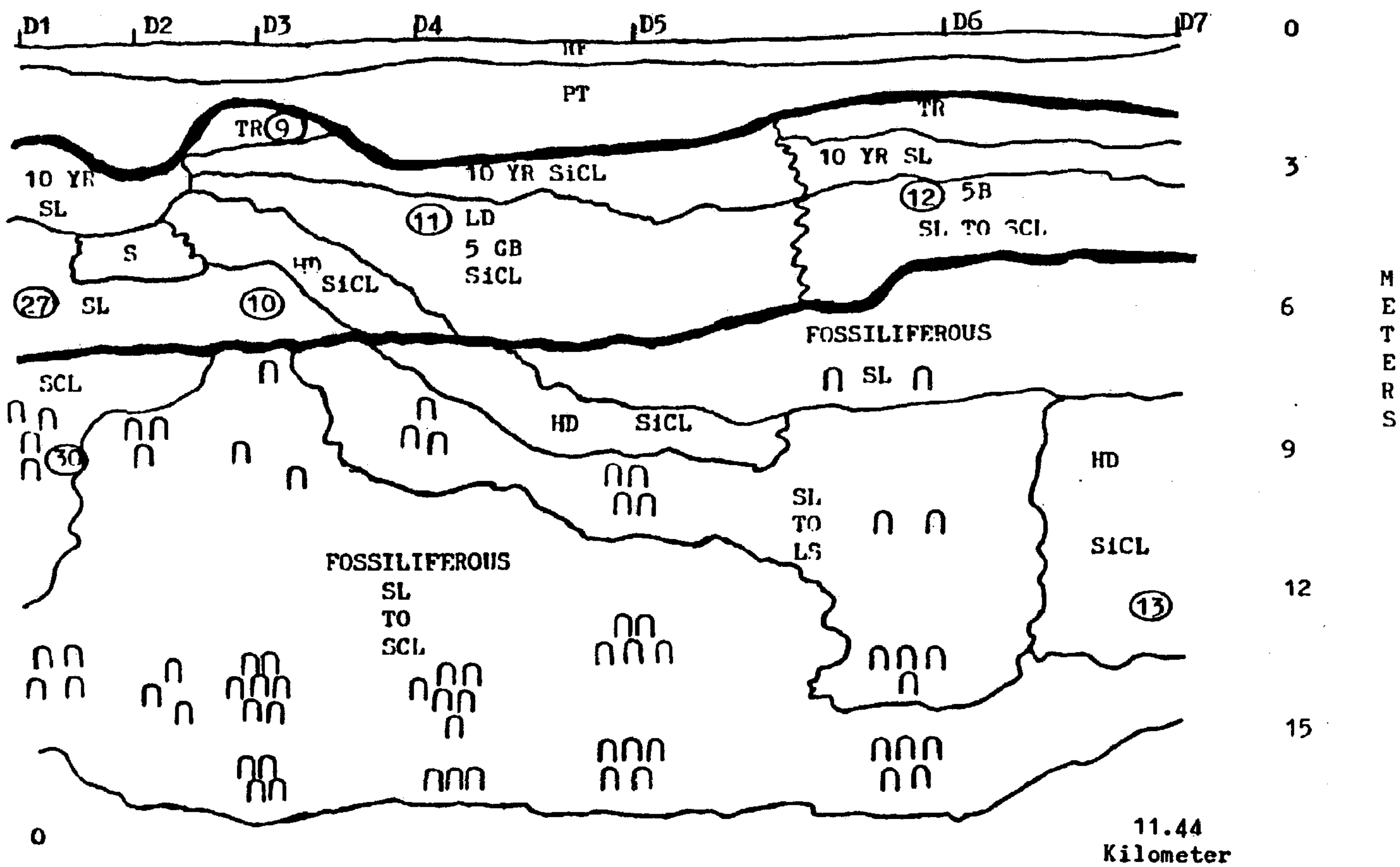


Figure 78. Description of the profile along DeHoog Road (refer to Figures 72 and 74).

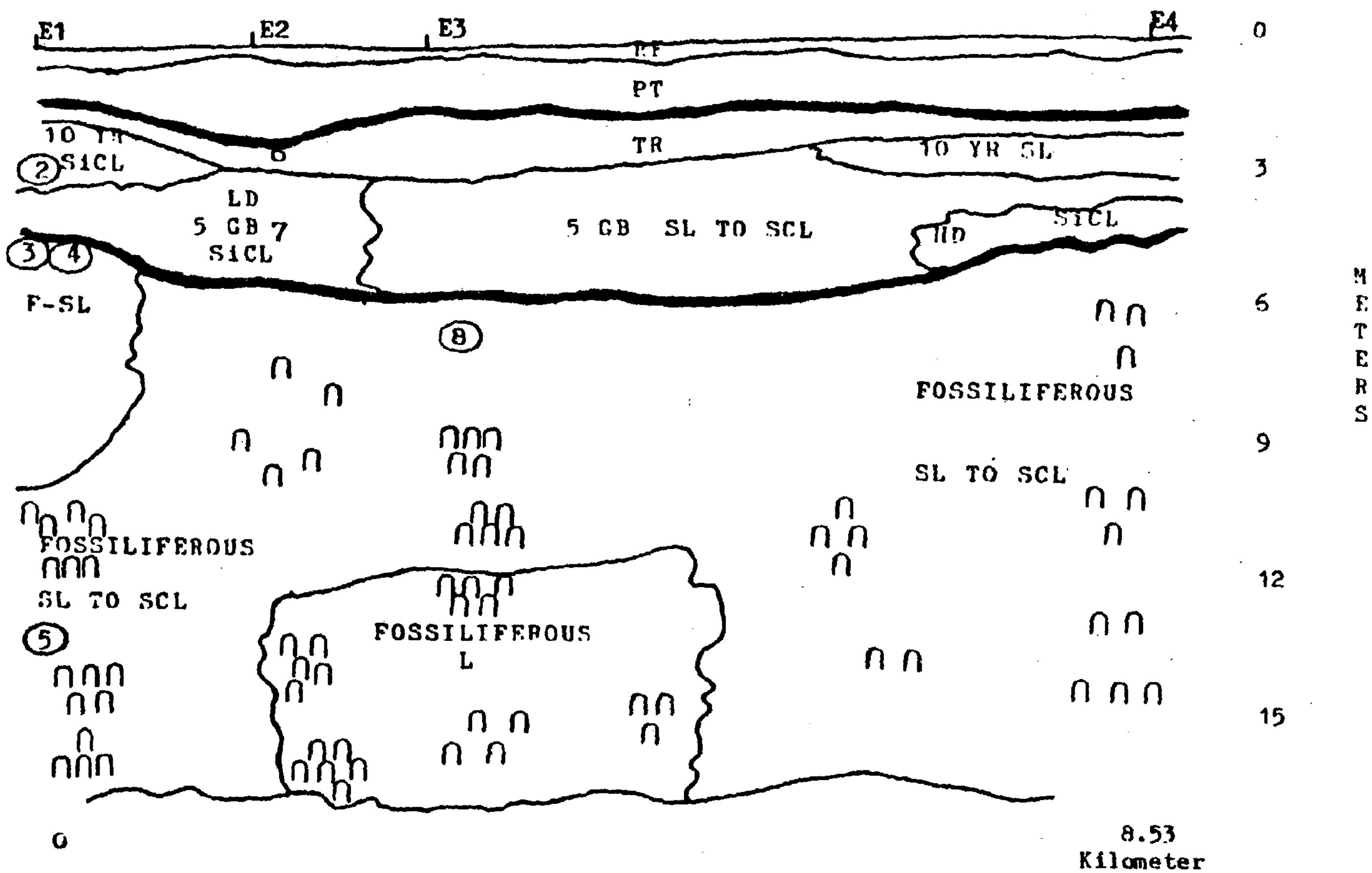


Figure 79. Description of the profile along Evans Road (refer to Figures 72 and 74).

Table 25. Bulk density values of mineral layers described below the peat deposits.

Textures	Mean Value gm/cm^3	Standard Deviation	Bore Hole Number*
Sandy loam	1.40	0.23	20,30, 8
Sandy loam with shells	1.44	0.31	21,16,27
Silty clay loam-I	1.69	0.41	12,4
Silty clay loam-II	1.37	0.10	6,11
Clay loam	1.19	----	31
Clay	1.09	0.08	14,15

*Location in Figures 74 to 76 where sample was taken.

consistent with, variation in hydraulic conductivities in a single continuous profile layer (Brady, 1973).

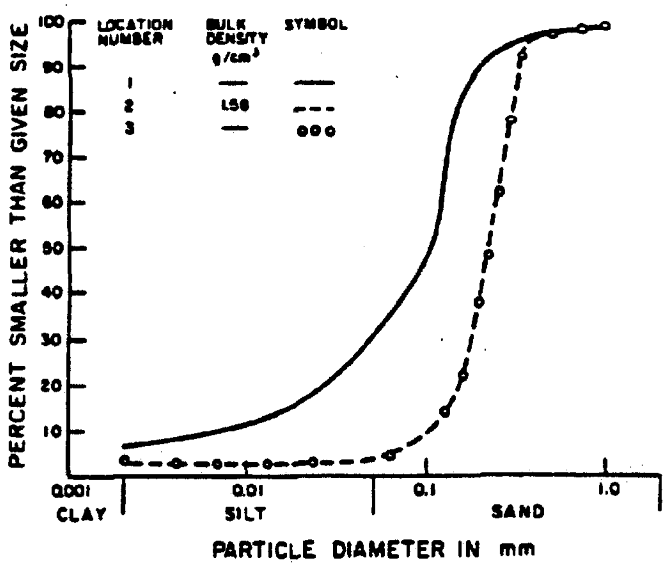
Particle diameter is plotted against the percentage by weight of soil particles less than the given diameter. Results for soil samples taken at the locations shown by numbers in Figures 75 to 79 are given in Figures 80 to 83.

The shapes and sizes of the soil particles affect water surface tension and therefore influence hydraulic conductivities (Bouwer, 1978). If characteristics such as bedding, water content, and water temperature are neglected, then a general assumption that sediments with smaller particles have smaller K values can be made. Therefore, beds with similar texture may have different hydrualic conductivity values.

The in situ hydraulic conductivities of the six major units are summarized in Table 26. The method of measuring conductivity will not create a variation in measurements as much as disturbance due to installation (Smiles and Youngs, 1965). Errors in field tests may be due to irregular auger holes, leakage around pipes, bad readings, and from environmental changes for readings made over an extended period. For the in situ tests, variations in the conductivity values for a particular textural group may be caused by interbedding of finer and coarser material or differences in bulk density or a combination of both.

Hydraulic conductivities obtained from laboratory constant head measurements for these six major layers were generally higher and were subject to larger variations than those obtained by the piezometer tests. Samples may have been disturbed by the vibracoring method and by removal from the profile. Even though care was taken, disturbance during transportation of the samples and loss of moisture may have occurred. Any one of these factors could have caused a loss of soil-pipe interface, thus increasing the flow rates measured in the laboratory and invalidating the K values obtained. Additional problems in resaturating the samples and conducting the K measurements in the lab led to the rejection of these values and the use of the values given in Table 26 in subsequent analyses.

Hydraulic conductivities of the deeper aquifers and confining clays as determined by U.S.G.S. (Ronald W. Coble, personal communication, May, 1983) are given in Table 27.



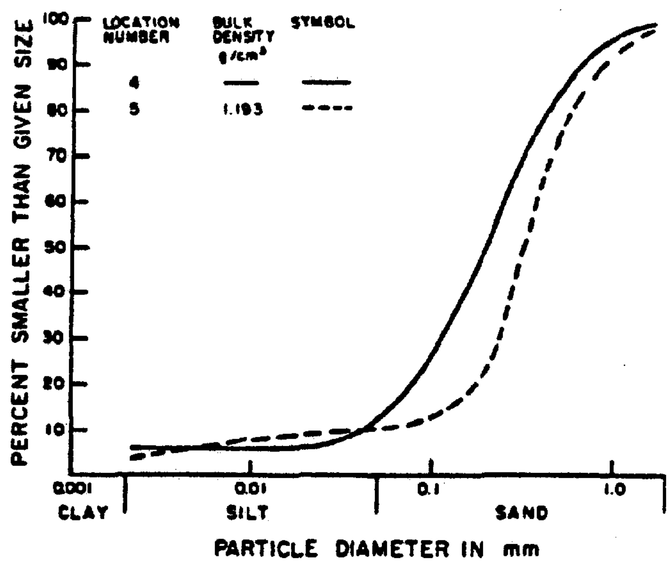
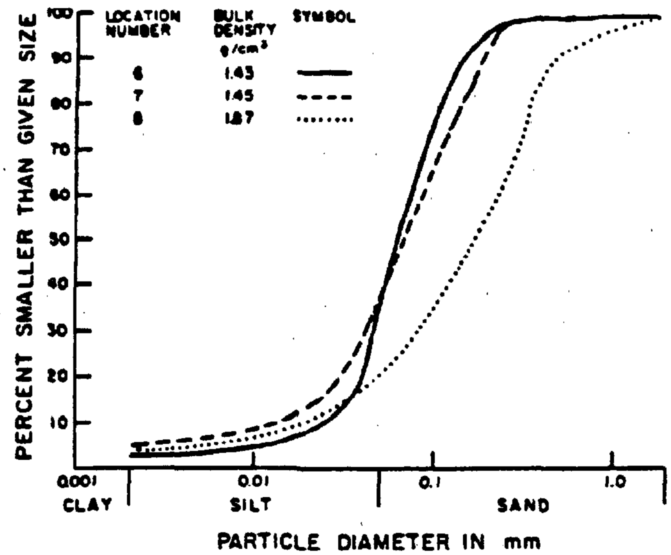


Figure 80. Particle size distributions for samples 1 through 5. The samples were obtained at locations specified in Figures 75 through 79.



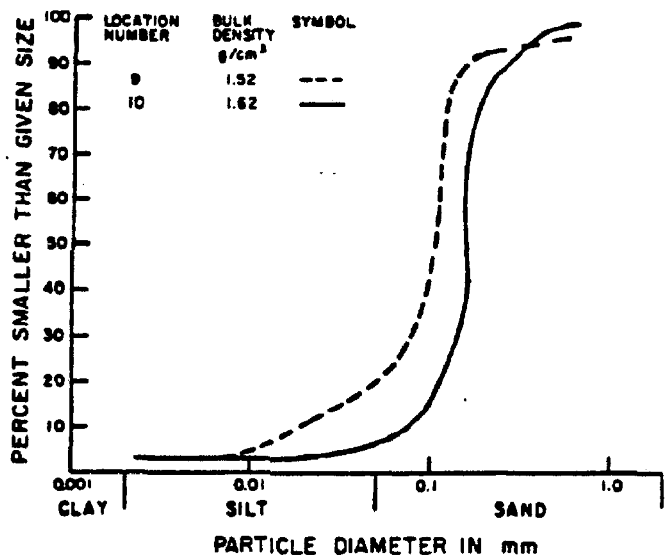


Figure 81. Particle size distributions for samples 6 through 10. The samples were obtained at locations specified in Figures 75 through 79.

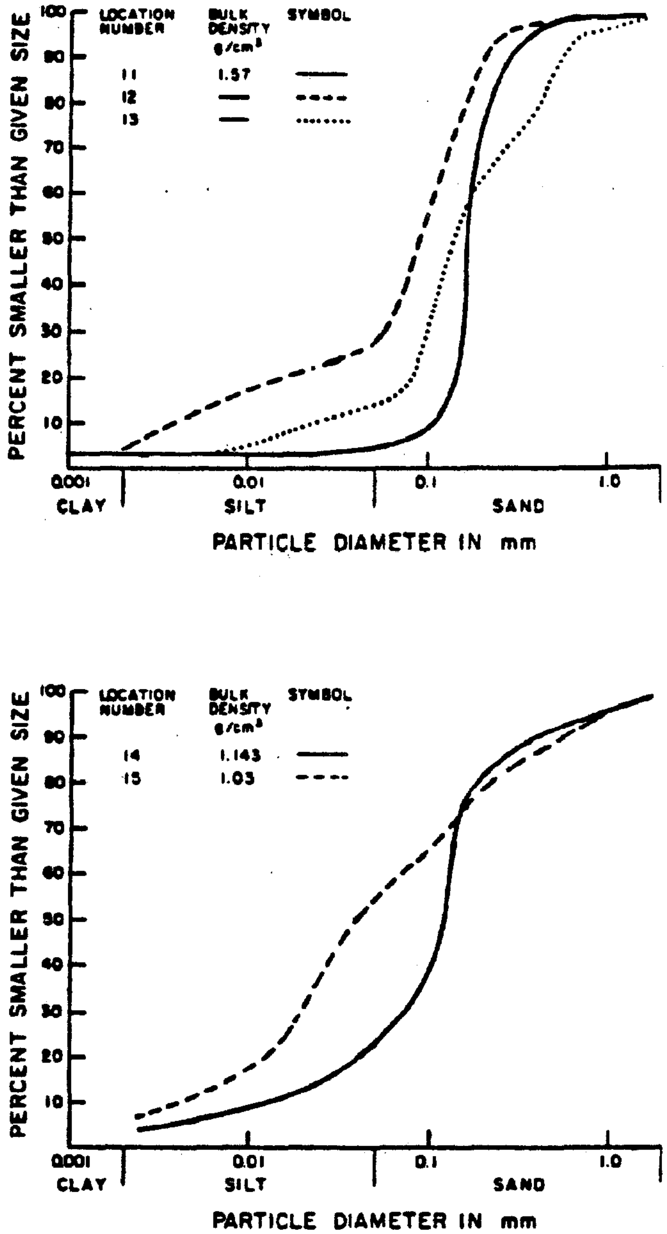


Figure 82. Particle size distributions for samples 11 through 15. The samples were obtained at locations specified in Figures 75 through 79.

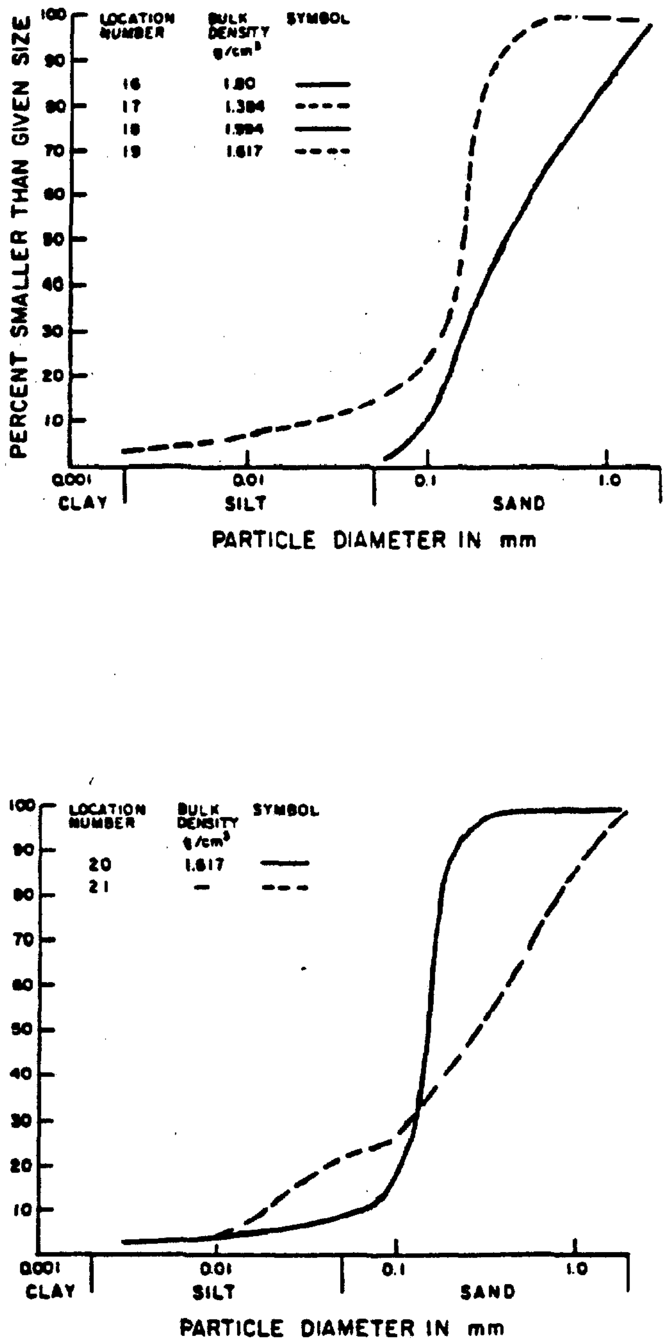


Figure 83. Particle size distributions for samples 16 through 21. The samples were obtained at locations specified in Figures 75 through 79.

Table 26. Saturated Hydraulic Conductivities Measured by the Piezometer Method (cm/hr).

Soil type	Piezometer test Maximum K	Minimum K	Mean K	Standard deviation	Location* number
Sandy loam	0.023	0.0002	0.004	0.008	20,9,26,3
Sandy loam with shells	0.328	0.026	0.105	0.149	21,27,28
Transition	0.508	0.362	0.416	0.079	22,23,24
Silty clay loam-I	0.009	0.0	0.0039	0.0038	25,12,29
Silty clay loam-II	0.112	0.022	0.0668	0.0643	11,6

*Sample taken from locations shown in Figures 75 to 79.

Table 27. Aquifer Hydraulic Conductivity Obtained from U.S.G.S. (personal communications with Ronald W. Coble).

Aquifer layer	Conductivity (cm/hr)
Confining clay above the Yorktown	1.8 (10^{-4})
Yorktown	0.40
Confining clay above the Pungo River	1.1 (10^{-4})
Pungo River	0.35
Confining clay above the Castle Hayne	3.2 (10^{-5})
Castle Hayne	2.40

ANALYSIS OF GROUNDWATER MOVEMENT

Introduction

The water movement analyzed in this study was for two dimensional, steady state conditions. Unsaturated flow was neglected. A two-dimensional profile view of the groundwater system is given in Figure 84. The governing equation for flow in two dimensions may be written as

$$\frac{\partial}{\partial x}\left[K_x \frac{\partial H}{\partial x}\right] + \frac{\partial}{\partial y}\left[K_y \frac{\partial H}{\partial y}\right] = 0$$

where H is the hydraulic head, K_x is the saturated hydraulic conductivity in the x-direction, and K_y is the saturated hydraulic conductivity in the y-direction. This equation is known as the Laplace Equation and is derived from Darcy's Law and from the principle of the conservation of mass. The soil was considered to be isotropic in the analyses conducted, that is $K_x = K_y$.

The average water table depth before mining varies from 0 to about 100 cm (Skaggs et al., 1980), so the thickness of the unsaturated zone is small compared with the saturated zone where most of the subsurface flow occurs. Since the flow in the unsaturated zone is much smaller than in the saturated zone, unsaturated flow was neglected. Hydraulic gradients and flow rates are obviously affected by the position of the water table. However, extreme limits of summer and winter water table elevations, as measured by Skaggs et al. (1980), can be used in steady state analyses to establish the range of groundwater flow rates.

Procedures

Analyses were conducted for two transects, A-B and C-D, illustrated in Figure 85. The boundary conditions along transect A-B are indicated in Figure 84. The position of the water table was assumed to vary between known limits, so the hydraulic head at the surface could be assigned for a given case. The hydraulic heads at Lake Phelps, and at the interface of the Castle Hayne aquifer were based on data presented by Heath (1975). Transects A-B and C-D, were chosen so that they would extend well beyond the mined area considered in the analyses. Thus it was assumed that the flow across the vertical boundaries, e.g. the right and left boundaries, was not affected by the mining process and was considered to be zero. For transect A-B, the left boundary was taken at a point near the middle of Lake Phelps and the right boundary was located at the edge of the mining area. Transect A-B was 2100 m long and C-D was 12,900 m long.

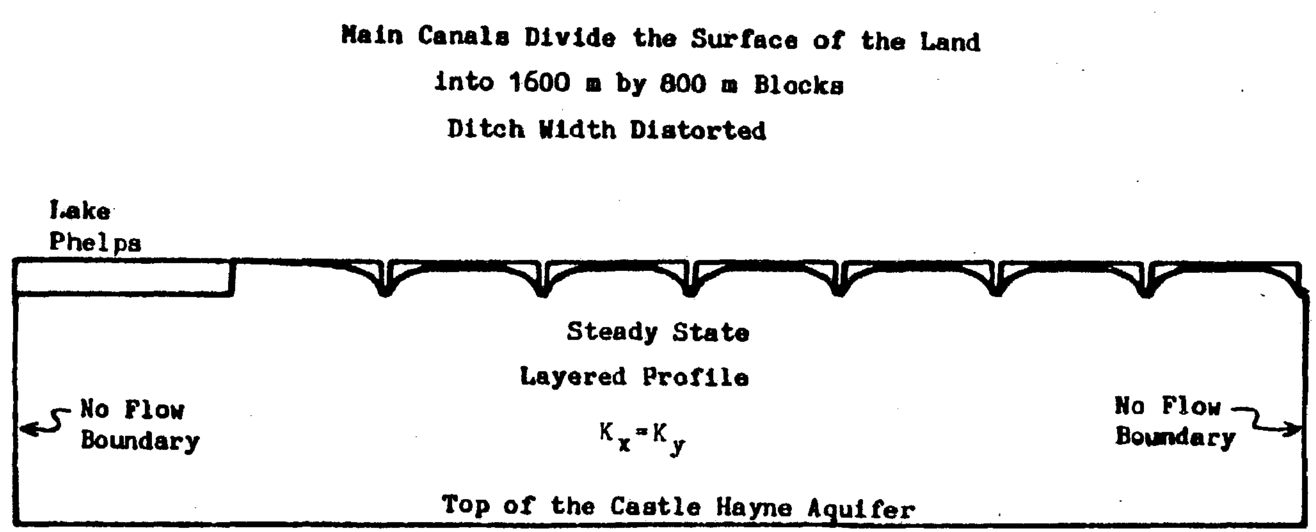


Figure 84. Typical cross section of the mining area. Drainage canals divide the surface of the land into rectangular blocks.

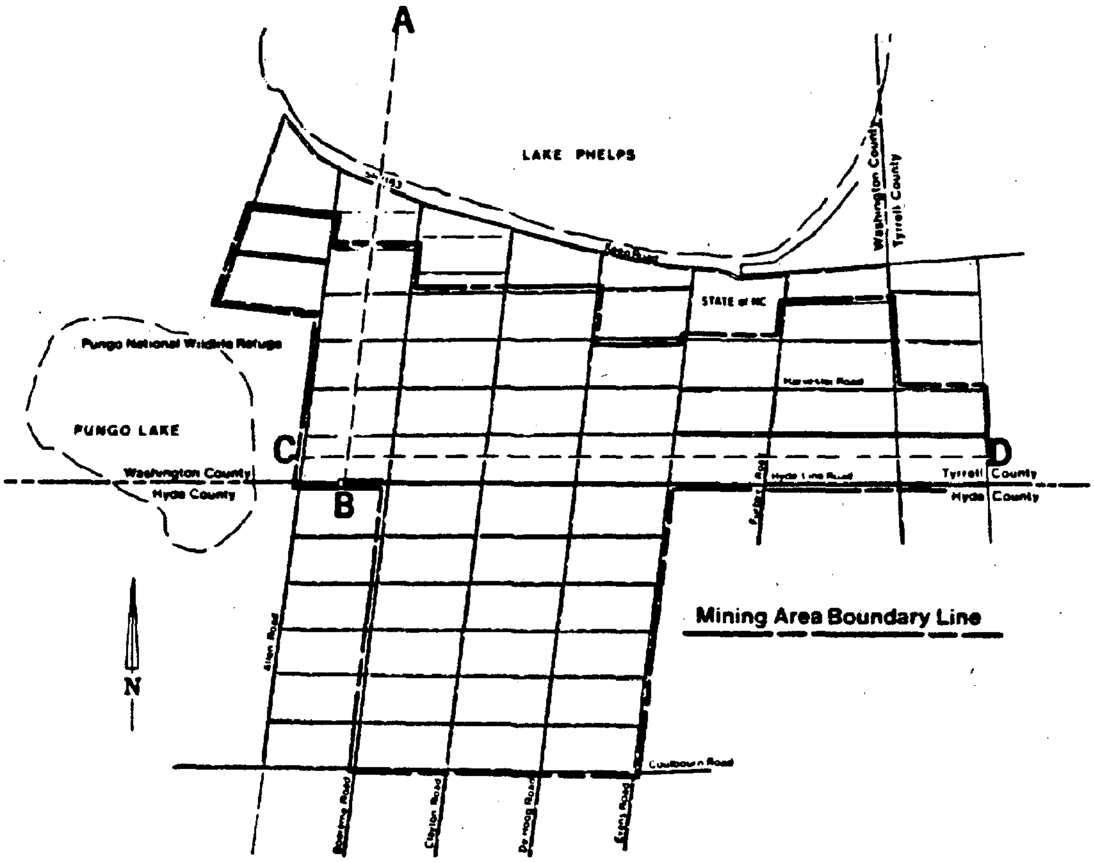


Figure 85. Location of First Colony Farms peat mining operation. Mining site boundaries are shown by bold lines. Main canals run north-south. Collector canal are spaced 0.8 km apart and run east-west. Field ditches are not shown.

Groundwater flow was described by solving the Laplace equation along transects A-B and C-D using soil properties discussd in previous sections and boundary conditions given above. The Finite Element Method (F.E.M.) was the procedure used to solve the governing equation. A computer model developed by Dhatt and Touzot (1981) makes use of this method with quadrilateral elements and Hermitian elements in one, two, and three dimensions. Quadrilateral elements were used in these solutions due to their flexibility in finding both potentials and gradients. Triangular and square shaped elements were used in the two dimensional analysis. The hydraulic fluxes were found at points inside the individual elements by the Gauss Integration Method. Sergerlind (1976) gives an applied presentation of the Finite Element Methods while Norrie and DeVeries (1978) give a theoretical explanation.

The validity of the finite element method (F.E.M.) used in this analysis was tested by comparing F.E.M. solutions for open ditch drainage situations to analytical solutions. The cases considered involved steady state drainage for a two-dimensional, uniform, completely saturated profile with a ponded surface. The F.E.M. solutions were within 2.5% of the analyticial solutions for both flow rates and hydraulic heads in all cases considered. Details of the comparisons as well as a review of literature on finite element methods are given by Foutz (1983).

Results and Discussion

Several different mining schemes were considered in evaluating the effects of peat mining on groundwater processes. Present drainage canals divide the land area into 1600 m by 800 m blocks. These blocks form convenient units for analyzing the effects of mining. Solutions were obtained for cases where a single block is mined, where four adjacent blocks are mined and where peat is removed from all blocks in a transect. In addition, the effects of deepening the main drainage canals to provide a better drainage outlet, of holding the water level in the canals high and of different field water table depths were analyzed.

Profile layers considered in the F.E.M. solutions are shown schematically in Figure 86, and the hydraulic conductivity values for the layers are given in Table 28. Based on data presented by Heath (1975), the piezometric head at the top of the Castle Hayne aquifer was assumed to be 0.64 m above mean sea level or 110.64 m above the top of the Castle Hayne, the ground surface in this area is approximately 5 m above mean sea level (115 m above the top of the Castle Hayne), and Lake Phelps is about 3.2 m deep.

Cases Considered

Solutions were obtained for the transects A-B and C-D in Figure 85. Transect A-B began in Lake Phelps at a point 2400 m

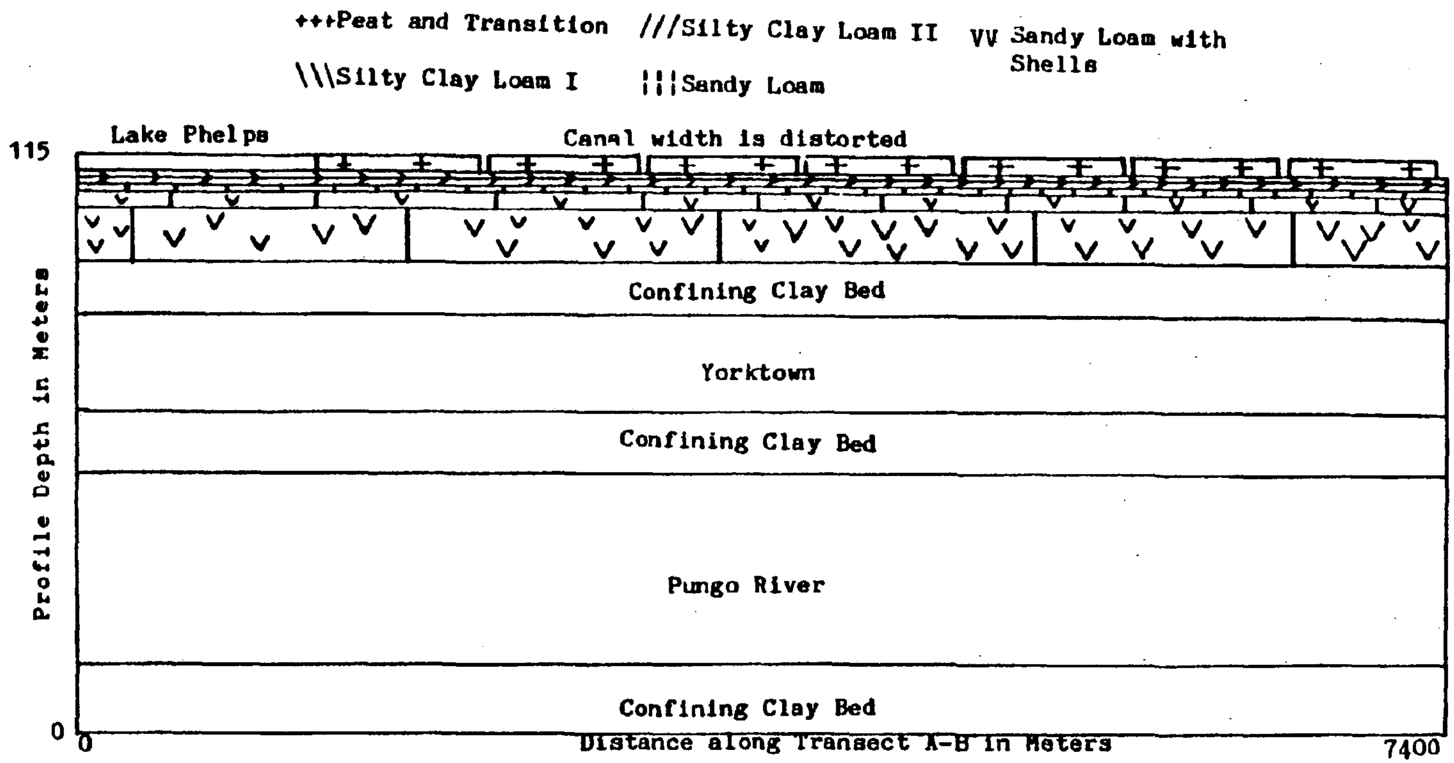


Figure 86. Profile description along transect A-B. Layers shown were based on borings made in this study. The figure is not to scale (refer to Table 28 for layer depths).

away from the shore and extended south 7400 m. It crosses seven drainage canals with point B located at the middle of the last canal. The soil surface next to Lake Phelps was 4.2 m above sea level, and at point B was 5 m above sea level.

Transect C-D extended 12,900 m from east to west across nine drainage canals in the mining site. Point C is located in the middle of the first canal, and point D was in the middle of the ninth canal. The elevation for the soil surface of this transect was 5 m above sea level for the entire transect.

Table 28. Soil layers and their hydraulic conductivities

Layer	Thickness m	Conductivity cm/hr
Organic	2.58	0.0200***
Transition	0.12	0.4320*
Silty clay loam - II	0.05	0.0668*
Silty clay loam - I	0.08	0.0087*
Sandy loam	7.17	0.0011*
Sandy loam with shell hash	5.00	0.1820*
Confining clay bed above the Yorktown	6.00	1.8 (10^{-4})**
Yorktown	18.0	0.400*
Confining clay bed above the Pungo River	10.0	1.1 (10^{-4})**
Pungo River	62.9	0.3500*
Confining clay bed above the Castle Hayne	3.10	3.2 (10^{-5})**

*found by field test performed in this study.

**obtained from U.S.G.S.

***obtained from Badr (1980).

Solutions were obtained for the cases shown schematically in Figure 87 for transect A-B and in Figure 88 for transect C-D. Two solutions were obtained for conditions prior to mining. In one case boundary conditions were chosen to represent the situation where no drainage canals existed in the area and the water table was coincident with the surface. The other case represented the existing situation with drainage canals as shown in Figure 87a. Mining cases considered included mining and lowering the surface of a single block, four adjacent blocks, and all of the blocks in a transect. A summary of the cases considered is given in Table 29.

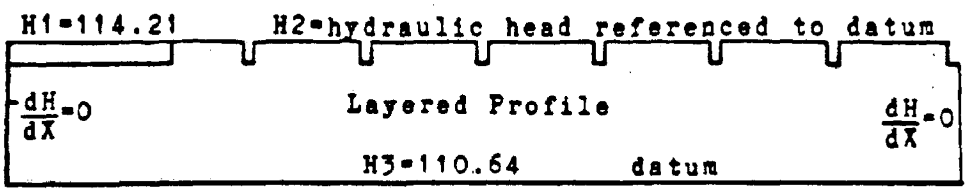
Effect of Mining on Deep Seepage

A summary of predicted rates of deep seepage to the Castle Hayne aquifer is given in Table 30. Case 1 indicates the potential deep seepage rate that would occur along transect A-B in the absence of drainage canals and with the water table near Lake Phelps at the soil surface. This represents conditions which might have occurred during wet periods before to any drainage activities in the area. Even though this case gave the highest seepage rates of all conditions analyzed, the predicted rate was only 0.043 cm/year. The low rate is primarily due to the low K values of the clay beds overlying the Yorktown, Pungo River and Castle Hayne aquifers (Table 28).

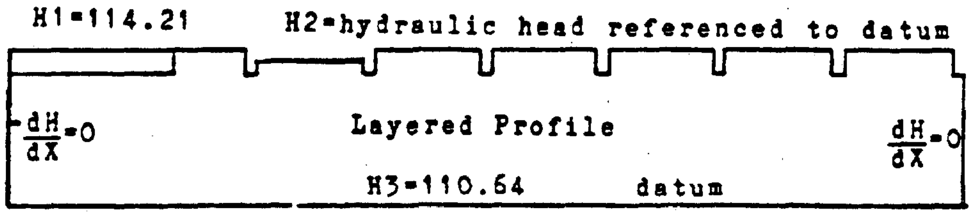
Equipotential lines for case 1 are shown in Figure 89. As expected in the absence of drainage canals, the equipotential lines are nearly horizontal with the largest potential drops occurring in the clay beds.

Cases 2 and 3 represent actual conditions prior to mining with a summer water table depth of 1.0 m. The drainage canals were assumed to be 3.2 m deep and to have a water depth of 0.3 m. Near the canals the water table was assumed to have an elliptical shape, rising to a depth of 1.2 m at a distance of 30 m from the canal. The water table was assumed to rise linearly from that point until it reached a depth of 1.0 m at a distance of 200 m from the canal. This water table position represented relatively dry summer time conditions as reported by Skaggs et al. (1980). Deep seepage was somewhat higher for transect A-B (case 2) than for transect C-D (case 3) because a portion of transect A-B contains Lake Phelps which has a higher water level and greater seepage than the land area. Equipotential lines for transect A-B are plotted in Figure 90. Note that gradients are higher under Lake Phelps than in the adjacent land where lowered water levels in the ditches cause decreased hydraulic heads. Because of the lowered water levels in the ditches, the equipotential plots show horizontal gradients toward the ditches as expected.

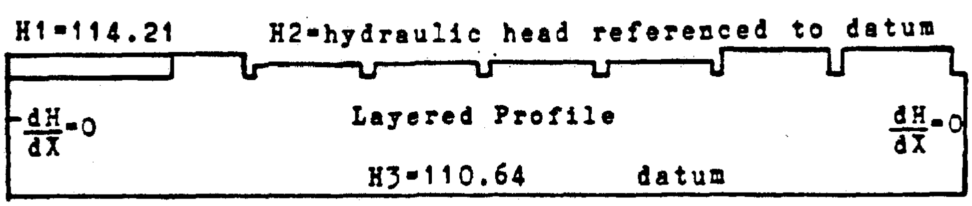
The effect of mining one 800 m by 1600 m block of land located near Lake Phelps (case 4, Table 30) was to reduce deep seepage from an average over the transect of 0.039 cm/yr to 0.038 cm/yr, a decrease of 2.6%. When four adjacent blocks were mined



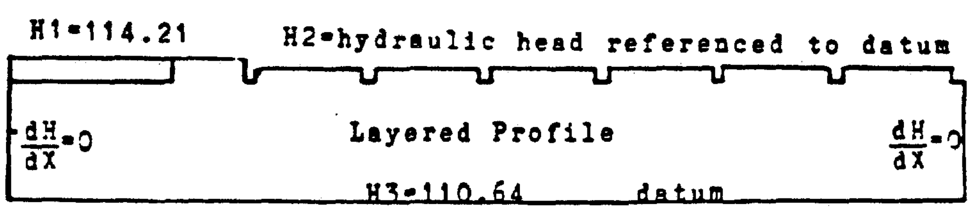
a) Premined



b) One block mined

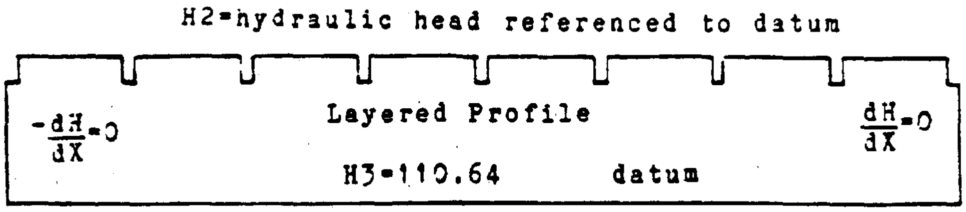


c) Four blocks mined

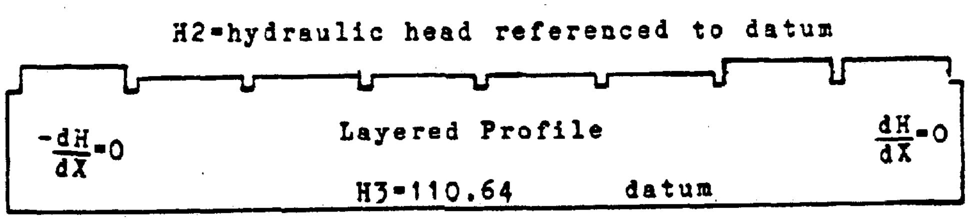


d) All blocks mined

Figure 87. Boundary conditions assumed along transect A-B. Canal width is distorted. Vertical scale is much greater than the horizontal scale.



a) Premined



b) Four blocks mined

Figure 88. Boundary conditions assumed along transect C-D. Canal width is distorted. Vertical scale is much greater than horizontal scale.

Table 29. Boundary conditions considered in evaluating the effect of mining on groundwater flow.

Case Number	Mining Condition	Seasonal Water Table Condition	Canal Depth	Transect
1	Before mining	At soil surface	No canal	A-B
2	Before mining	Summer	3.2 m	A-B
3	Before mining	Summer	3.2 m	C-D
4	Peat mined between canals located 2640 m and 3425 m from point A in transect A-B	Summer	3.2 m	A-B
5	Peat mined between 2640 m and 5300 m from A along transect A-B	Summer	3.2 m	A-B
6	Peat mined between 3220 m and 9660 m from C along transect C-D	Summer	3.2 m	C-D
7	Peat mined between 2640 m and 7400 m from A along transect A-B	Summer	3.2 m	A-B
8	Before mining	Summer	4.2 m	A-B
9	Peat mined between 2640 m and 5300 m from A along transect A-B	Summer	4.2 m	A-B
10	Before mining with a sand layer between Lake Phelps and the first drainage canal	Summer	3.2 m	A-B
11	Peat mined between 2640 m and 5300 m with a sand layer between Lake Phelps and the first drainage canal	Winter	3.2 m	A-B

Table 30. Groundwater flow rates to the Castle Hayne aquifer for the various boundary conditions assumed.

Case Number Refer to Table 30	Flow Rate cm/yr	Percentage Change from Premined Case for the same transect
1	0.043	---
2	0.039	---
3	0.025	---
4	0.038	2.6%
5	0.037	5.1%
6	0.021	16%
7	0.0367	5.9%
8	0.039	0%
9	0.037	0%
10	0.039	0%
11	0.040	1.3%

(case 5, Table 30) the decrease in seepage was 5.1% compared to the pre-mined case (case 2). In all cases mining is assumed to lower the land surface 1.5 m, and the water table depth at distances greater than 200 m from the drainage canals is assumed to be 0.85 m deep. For cases 4 through 7 the canals were assumed to be at the same elevation as before mining; i.e., it was assumed that the present canal depth will provide the necessary drainage outlet after mining. Again the water table close to the canals was assumed to be elliptical rising to a depth of 1.2 m at 30 m from the canal.

Equipotential lines for cases 4 and 5 are plotted in Figures 91 and 92, respectively. Comparison with the pre-mined case (Figure 90) shows that mining reduced the potentials under the mined area and increased horizontal gradients from the surrounding areas. Vertical gradients under the mined area are reduced which results in the reduction of deep seepage. For example mining four adjacent blocks caused the 112.5 m equipotential to rise from an elevation of about 55 m above the datum (top of the Castle Hayne), as shown in Figure 90, to near the surface, about

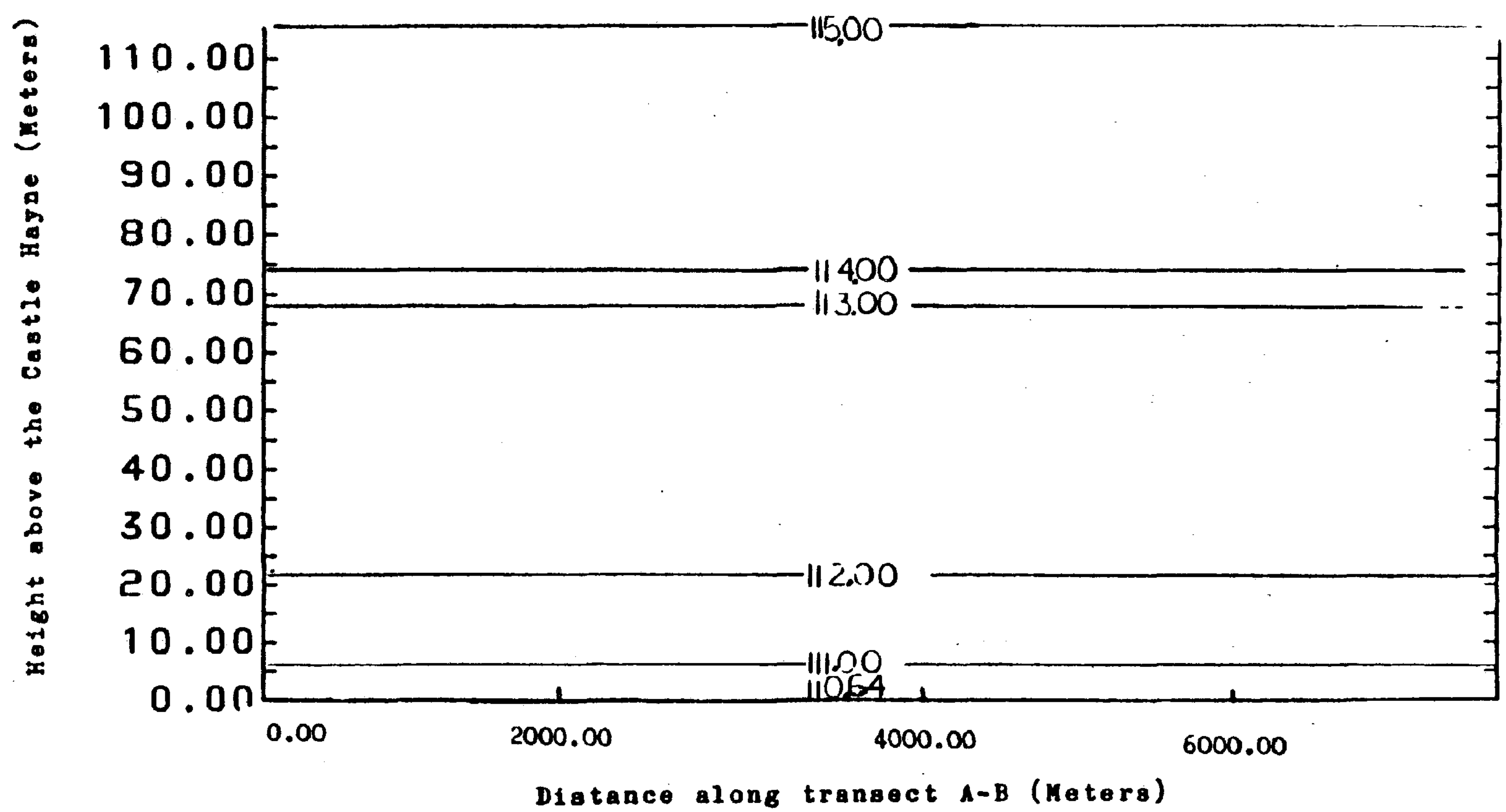


Figure 89. Equipotential lines for the case of no drainage canals with the water table ponded at the surface.

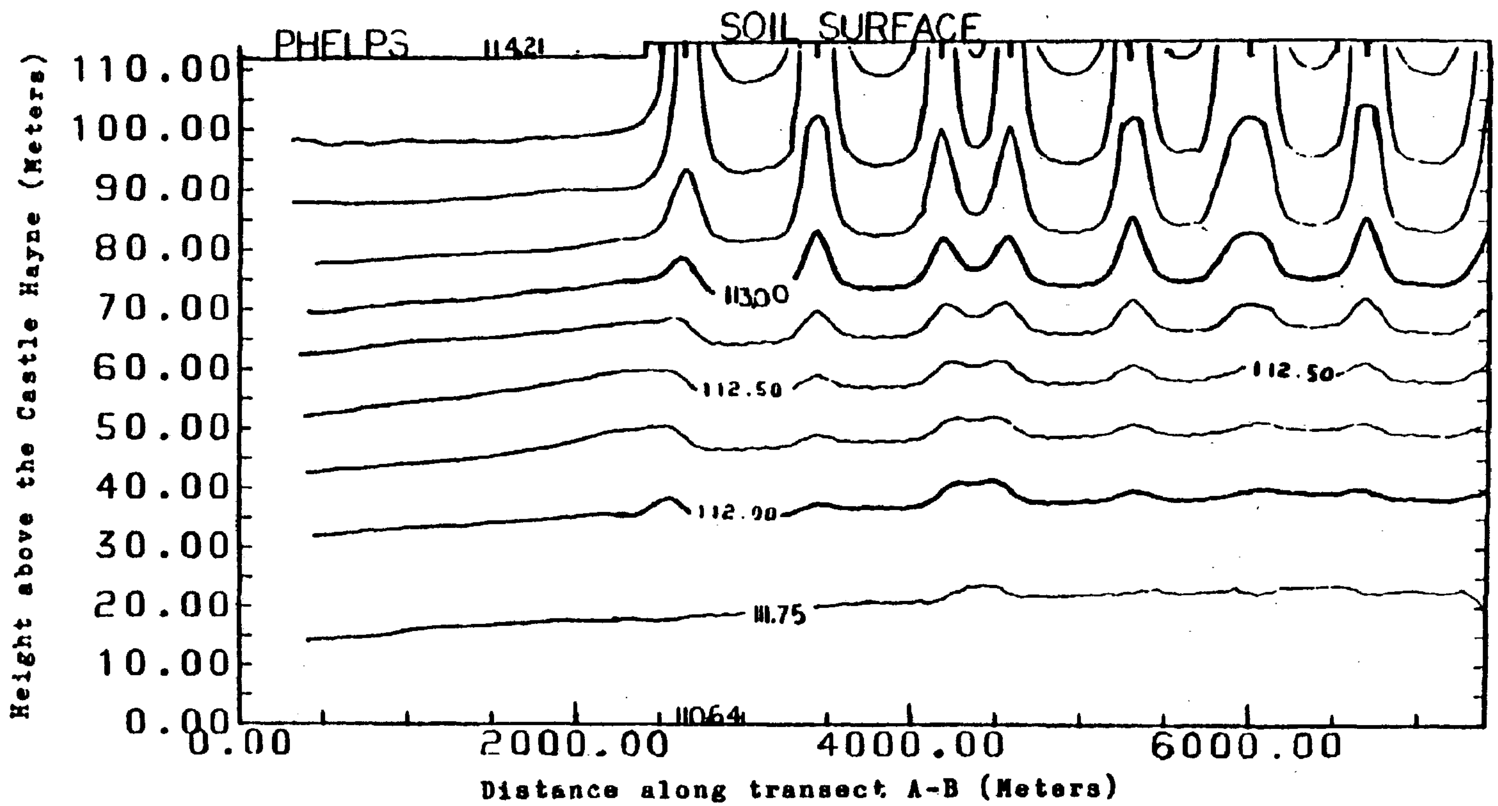


Figure 90. Equipotentials in m referenced to the top of the Castle Hayne aquifer for the pre-mined case along transect A-B. Head at the Castle Hayne was assumed to be 110.64 m (0.64 m above sea level).

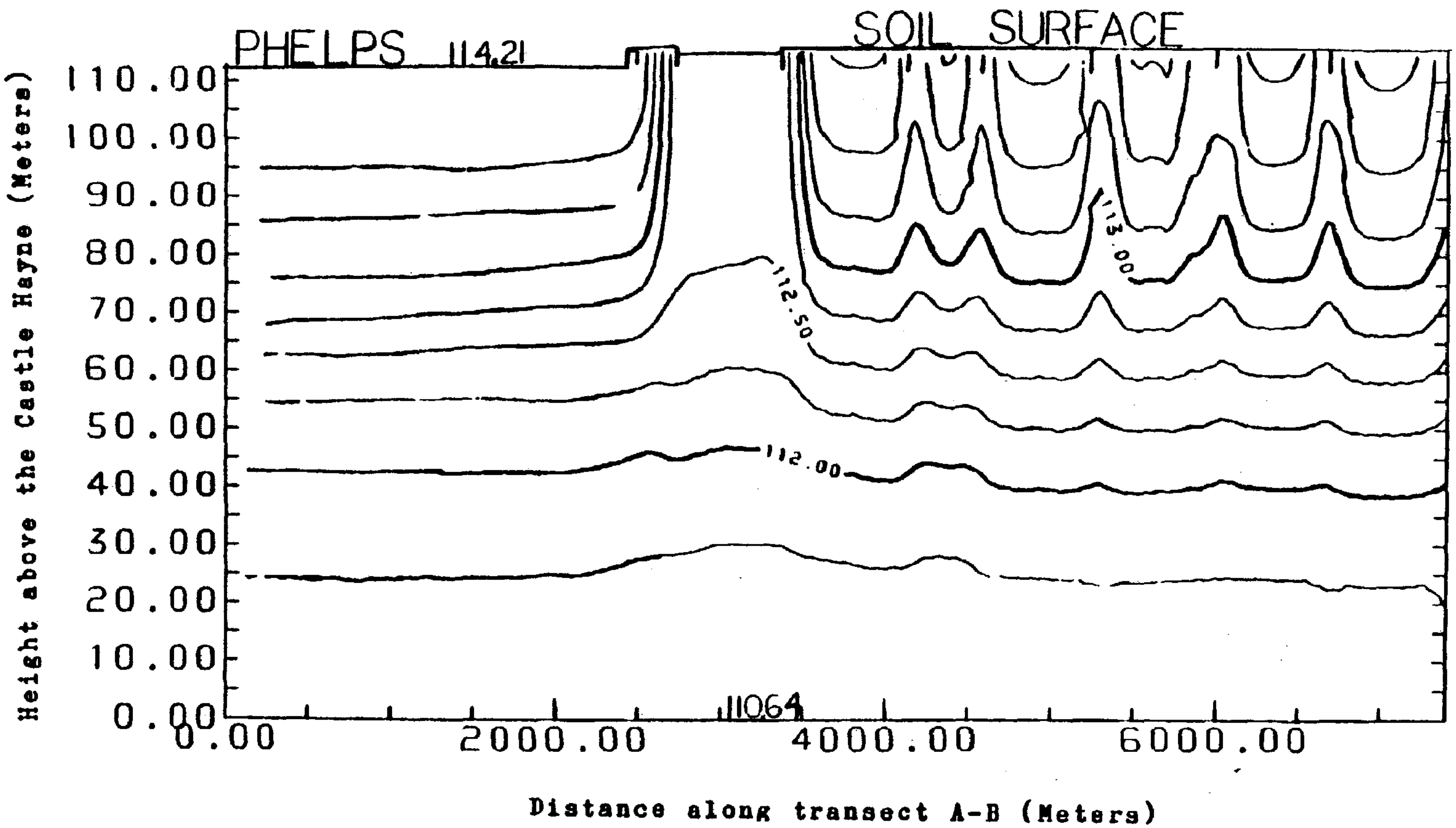


Figure 91. Equipotentials in m referenced to the top of the Castle Hayne aquifer for case 4 where one block is mined in transect A-B. Head at the Castle Hayne was assumed to be 110.64 m (0.64 m above mean sea level).

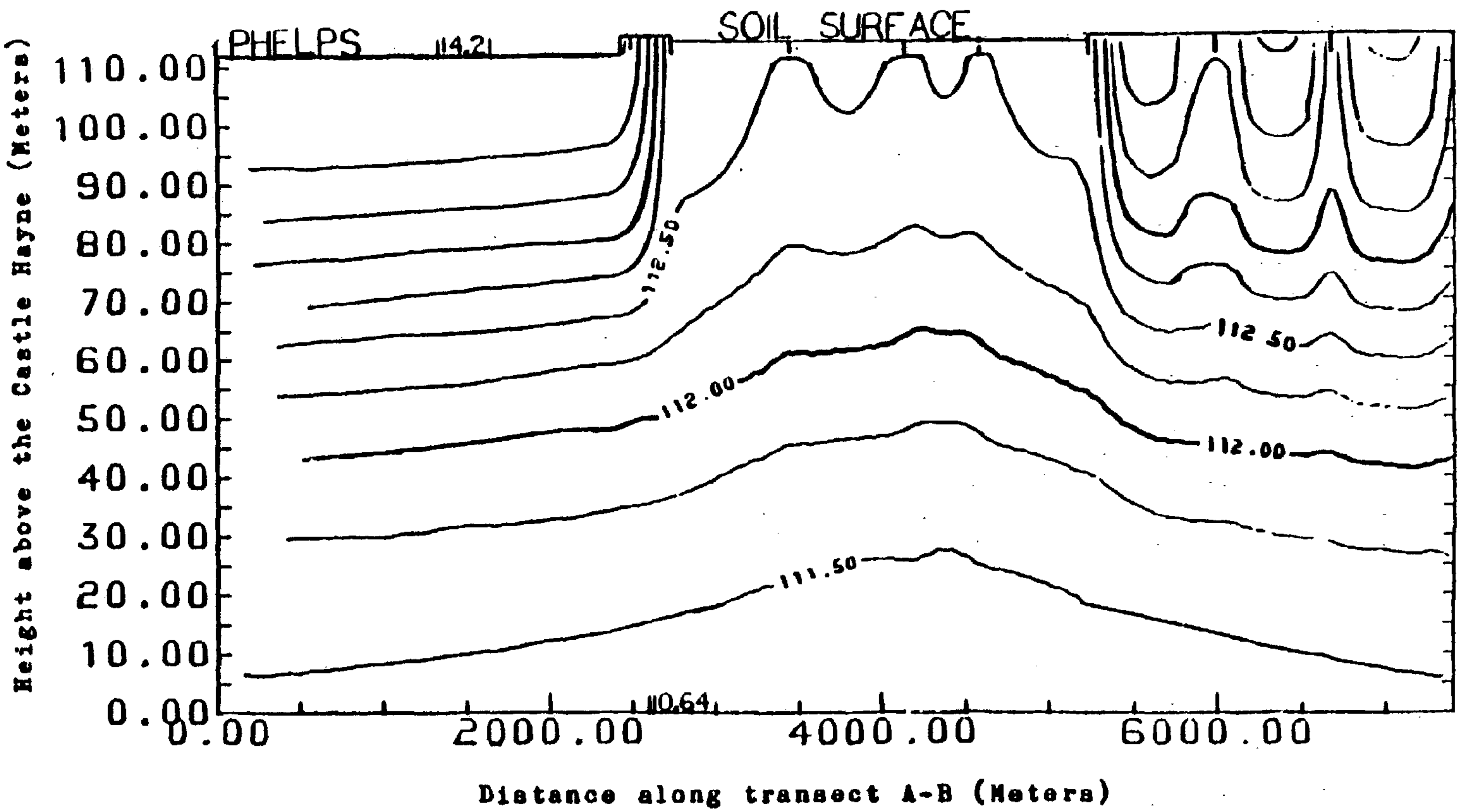


Figure 92. Equipotentials in m referenced to the top of the Castle Hayne aquifer for case 5 where four adjacent blocks are mined in transect A-B. Head at the Castle Hayne was assumed to be 110.64 m (0.64 m above mean sea level).

110 m above the datum (Figure 92). However, at one-block removed from the mined area, the equipotential is approximately at its original position (Figure 92).

The effect of mining four blocks along transect C-D was to reduce deep seepage from 0.025 cm/yr to 0.021 cm/yr. Deep seepage rates were smaller than for transect A-B because transect C-D did not go through Lake Phelps. The change in deep seepage rates due to mining was greater than for transect A-B for the same reason.

Case 7 considers the effect of mining all blocks in transect A-B (Figure 84d). The predicted average deep seepage rate for the transect was 0.0367 cm/yr which was 7.4% less than the pre-mined condition. The equipotential lines are plotted in Figure 93. As noted for the previous case, reduction of the surface elevation due to mining reduces the hydraulic heads and vertical gradients in the mined area. Horizontal gradients near Lake Phelps are increased (compare with Figure 90), but the effect of the lake on the equipotentials does not extend beyond the first block.

Results presented above indicate that deep seepage in the proposed mining area is small and is probably negligible for practical purposes. The largest percentage change due to mining was predicted for transect C-D where mining four of the eight blocks in the transect reduced deep seepage by 16%. A rough estimate of the percentage change in deep seepage as a result of mining the whole area can be made by assuming vertical flow only and simply examining the change in the hydraulic gradient. If the original surface is 115 m above the top of the Castle Hayne and the average water table is 1.0 m deep, the difference in hydraulic head would be $H = 114 - 110.6 = 3.4$ m. Lowering the land surface by 1.5 m would reduce the hydraulic head near the surface to $114 - 1.5 = 112.5$ and $H = 112.5 - 110.6 = 1.9$ m. Thus H would be reduced by $3.4 - 1.9 = 1.5$ m or 44% from its original value. Therefore, seepage rates would also be reduced by the same percentage as the gradient, 44%. Our predicted effects on seepage rates are less than 44% for two reasons. Firstly, the hydraulic heads at and near to the canals are not lowered due to mining. Secondly, unaffected areas such as Lake Phelps in transect A-B and adjacent unmined areas in transect C-D moderate the effects of lowering the land surface and reduce the average seepage rate. Nevertheless, vertical seepage from a unit area in the center of a block located in the interior of the mining area could be reduced by approximately 44% due to a decrease in vertical hydraulic gradients. It should be emphasized however, that the predicted seepage rates are extremely small for all cases, so that mining effects on deep seepage are negligible for practical purposes.

It should be noted that an independent investigation of this process (PMA, 1983) determined that a gradient reversal would occur in the surface aquifers resulting in an unwelling of water into the mined areas. This result was dependent upon their

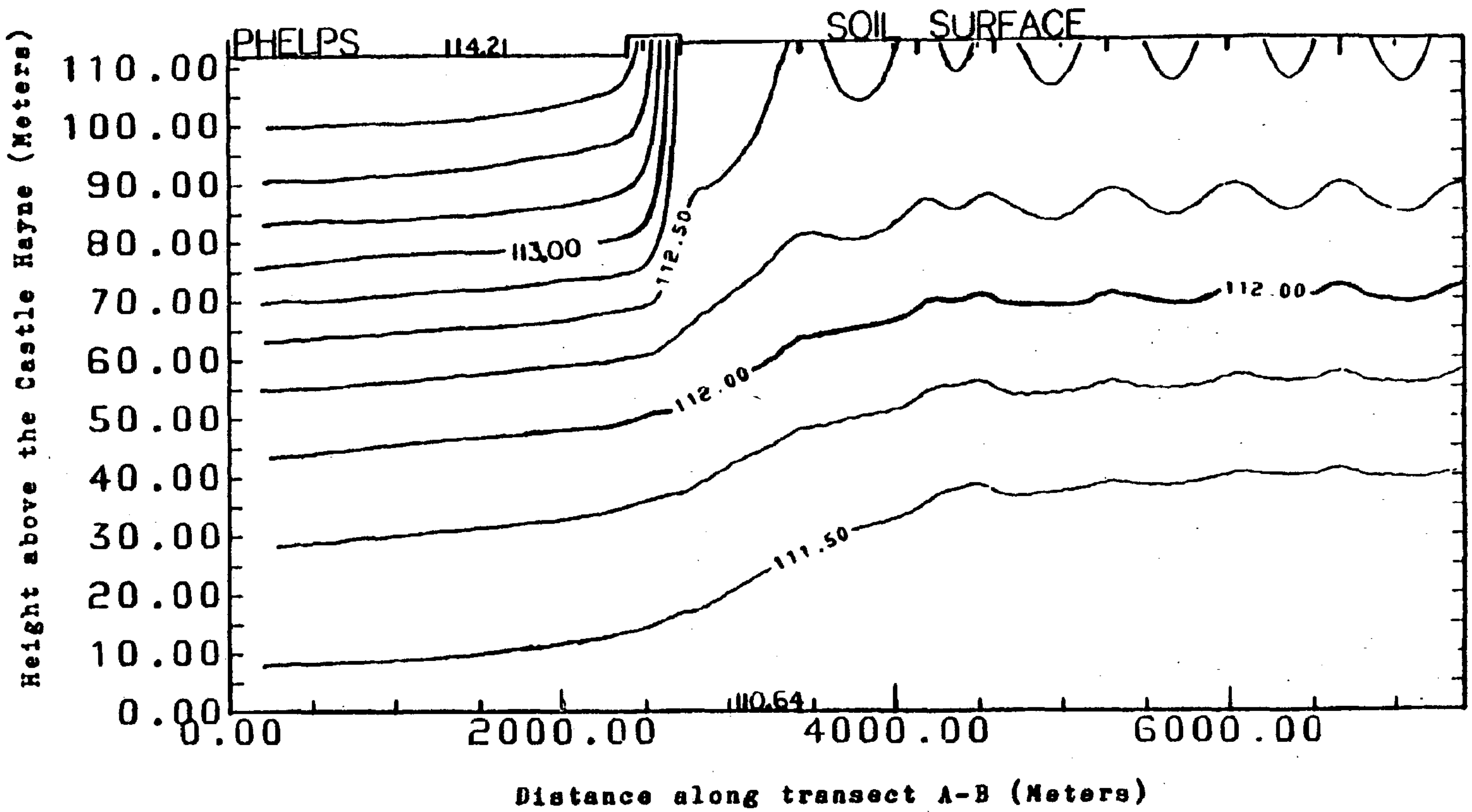


Figure 93. Equipotentials in m referenced to the top of the Castle Hayne aquifer for case 7 where all blocks are mined in transect A-B. Head at the Castle Hayne was assumed to be 110.64 m (0.64 m above mean sea level).

assumption that the clay lenses above and below the sandy loam unit were continuous. In the boring analysis reported herein, these lenses were not found to be continuous. The unwelling prediction is also dependent upon the lateral conductivities of the sandy loam and sandy loam with shell hash units being high enough to maintain an adequate supply of water to those units. PMA reported hydraulic conductivities of those units several orders of magnitude higher than those measured in the field and laboratory study (Table 26). The results presented herein are also limited by assumptions, including profile simplifications and boundary conditions as stated earlier. However, it is believed that the mining process is slow enough to allow the aquifers to readjust to a new equilibrium thus preventing any continuous unwelling effect.

Effect of Mining on Seepage from Lake Phelps

A main drainage canal is approximately 200 m from Lake Phelps and runs parallel to the southern shore of the lake, (Figure 85). The effects of mining on the ground water movement from the lake to this canal and others in the area were evaluated by comparing predicted seepage rates out of Phelps for conditions before and after mining. Seepage rates from Lake Phelps for varying boundary conditions are summarized in Table 31.

The recharge to Lake Phelps can be described by the water balance equation,

Recharge = Rainfall + Surface Runoff - Evaporation + Seepage to or from the lake + Outflow from the lake.

The seepage component was calculated by solving the Laplace equation as discussed previously. The surface runoff into the lake was assumed negligible. Based on data presented by Heath (1975), the average annual rainfall in the area was considered to be 129 cm (51 inches) and the average annual lake evaporation was considered to be about 91 cm (36 inches). Based on these values, the recharge represented by the difference between rainfall and evaporation was estimated to be 38 cm/yr. The water level in the lake is held at a relatively constant elevation by a control structure located in the outlet canal on the North side. As shown in Table 31 seepage from the lake amounts to approximately 0.2 cm/year for all cases considered. Therefore drainage from the lake through the outlet canal amounts to about 37 to 38 cm/yr on the average.

The direction of seepage from Phelps prior to mining (case 2) can be determined from the equipotential lines in Figure 90. The vertical boundary of the lake was assumed to be 3.2 m high. The lateral flow rate across this boundary was 0.128 cm^3/cm/hr and the vertical seepage rate across the bottom area of the lake was 0.19 cm/yr. The length of the boundary canal adjacent to the

Table 31. Flow rates of water seeping out of Lake Phelps for various boundary conditions assumed.

Case Number Refer to Table 29	Flow rate in cm/yr** Lateral seepage from lake*	Vertical seepage through lake bottom
2	0.0032	0.19
4	0.0033	0.19
5	0.0033	0.19
7	0.0033	0.20
8	0.0034	0.19
9	0.0034	0.20
10	0.0148	0.19
11	0.0150	0.19

*Values carried to four significant digits to show changes.

**Based on a shore length adjacent to the area of 9650 m and a lake surface area of 3.35×10^7 m^2.

mining site is 9650 m and the area of the lake is 3.35×10^7 m^2. Therefore the predicted lateral seepage from the lake to the section of the boundary canal adjacent to the site was 0.0032 cm/yr (Case 2, Table 31). Thus the lateral seepage from the lake is negligible in comparison to the vertical seepage which is itself an extensively small value.

The effects of peat mining on seepage from Phelps were evaluated for the three mining schemes described for cases 4, 5, and 7 in Table 29. The predicted lateral seepage rate across line E-F for all three mining schemes was 0.131 cm^3/cm/hr, which amounted to 0.0033 cm/yr compared to 0.0032 cm/yr for the before mining Case 2. The seepage across the bottom of the lake varied by no more than 0.01 cm/yr from the pre-mined value of 0.19 cm/yr. Thus, the removal of 1.5 m of peat from the soil surface results in a negligible change in the total seepage from the lake.

Sandy beds could exist between Lake Phelps and the adjacent drainage canal, although there was no direct evidence of their presence from the bore holes. The effect of such a condition was considered by assuming a sand bed with high conductivity extending from the bottom of the lake to the first two drainage canals. The K value of the sand was assumed to be 3 cm/hr. The layer was 3.2 m below the soil surface and was 12.0 cm thick. These solutions provided information on a situation in which potentially high seepage rates could occur. Again it should be emphasized that there was no evidence that such a continuous bed exists, but there is the possibility that such discontinuous lenses could affect seepage rates.

The assumed sand bed did not affect seepage through the bottom of Lake Phelps or deep seepage on an areal basis, but it dramatically increased the predicted lateral seepage from Lake Phelps to the boundary canal by 360%. Lowering the land surface by mining had a similar effect as before, however, increasing total seepage by less than 0.1% of the pre-mined case (Case 11, Table 31).

SUMMARY AND CONCLUSIONS

Data from drill holes to a depth of 17.8 m at 22 locations on the peat mining site on First Colony Farms were used to describe the sediments beneath Allen, Boerema, Clayton, DeHoog, and Evans Roads. Six units were continuous between most of the drill holes and were considered to have the major influence on groundwater movement. Samples were taken from these six units for bulk density and particle size analyses. The saturated hydraulic conductivities of the layers were obtained in the field by the piezometer method and by the constant head method in the lab.

The sediments found below the peat deposits were a transition zone consisting of organic and sandy material, sandy loam, sandy loam with fossils, and two types of silty clay loam. The bulk densities of the layers ranged from 1.37 g/cm^3 to 1.71 g/cm^3.

Three piezometer tests per unit were conducted to measure hydraulic conductivities for the six mineral layers. Two samples per layer were obtained for the constant head conductivity test. The average K values were 0.51 cm/hr for the transition zone, 0.33 cm/hr for the sandy loam with shells, and 0.023 cm/hr for the sandy loam. The conductivities for the silty clay loams ranged from 0.008 cm/hr to 0.022 cm/hr. The hydraulic conductivities for layers deeper than 17.8 m were obtained from unpublished data of the U.S.G.S.

The Finite Element Method was used to evaluate the effects of peat mining on groundwater flow. Deep seepage to the Castle Hayne aquifer, seepage from Lake Phelps, and flow to the drainage canals were determined by solving the Laplace equation. Boundary conditions representing before and after mining situations were used in the solutions to this equation.

Due to the low hydrualic conductivity values of the profile the highest deep seepage rate to the Castle Hayne aquifer obtained for a case representing before mining conditions was 0.039 cm/year. Four mined conditions were assumed to exist along a transect which crossed seven drainage canals that divide the peat into 1600 m by 800 m blocks. Three cases were used to represent conditions that might exist if a single block, four adjacent, or all blocks along the transect were mined. The deep seepage rates for these mined conditions were 0.038 cm/yr, 0.037 cm/yr and 0.0367 cm/yr, respectively. The deep seepage rates were not affected by deepening the drainage canals 1 m.

Horizontal seepage rates from Lake Phelps were calculated for two-dimensional conditions. These rates were assumed to exist along the boundary of the mining site therefore a recharge rate over the entire lake area could be calculated. The total seepage out of Phelps, under conditions representing the use of canals for agriculture, was 0.193 cm/yr. The lateral seepage rates from the lake were approximately 2% of the vertical seepage rates through the bottom of the lake. By lowering the bottom of the canals 1 m, the total seepage out of the Lake Phelps increased by less than 0.2%.

The assumption of a continuous sand layer between Lake Phelps and the canal nearest the lake did not effect deep seepage to the Castle Hayne but did increase flow out of Phelps to 0.215 cm/yr. A combination of the sand layer condition and a condition with four adjacent blocks mined had negligible increased effect on the total seepage from the lake.

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