# RESEARCH SUB-PROGRAM

## PARTITIONING OF SOLUTES FROM AGRICULTURAL FIELDS WITHIN THE HYDROLOGIC SYSTEM AT TWO SITES IN SOUTHERN ONTARIO AND THE SUBSEQUENT IMPACT ON ADJACENT AQUATIC ECOSYSTEMS

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**Prepared by:** Dr. David Rudolph and Dr. Gary Parkin (Kachanoski, Barton), Waterloo Centre for Groundwater Research, University of Waterloo, Waterloo, ONT N2L 3G1

**On behalf of:** Research Branch, Agriculture and Agri-Food Canada, Pest Management Research Centre (London)
1391 Sandford St.
London, Ontario N5V 4T3

# *FORWARD*

This report is one of a series of COESA (Canada-Ontario Environmental Sustainability Accord) reports from the Research Sub-Program of the Canada-Ontario Green Plan. The GREEN PLAN agreement, signed Sept. 21, 1992, is an equally-shared Canada-Ontario program totalling $64.2 M, to be delivered over a five-year period starting April 1, 1992 and ending March 31, 1997. It is designed to encourage and assist farmers with the implementation of appropriate farm management practices within the framework of environmentally sustainable agriculture. The Federal component will be delivered by Agriculture and Agrifood Canada and the Ontario component will be delivered by the Ontario Ministry of Agriculture and Food and Rural Assistance.

From the 30 recommendations crafted at the Kempenfelt Stakeholders conference (Barrie, October 1991), the Agreement Management Committee (AMC) identified nine program areas for Green Plan activities of which the three comprising research activities are (with Team Leaders):

1. Manure/Nutrient Management and Utilization of Biodegradable Organic Wastes through land application, with emphasis on water quality implications
   A. Animal Manure Management (nutrients and bacteria)
   B. Biodegradable organic urban waste application on agricultural lands (closed loop recycling) (Dr. Bruce T. Bowman, Pest Management Research Centre, London, ONT)

2. On-Farm Research: Tillage and crop management in a sustainable agriculture system. (Dr. Al Hamill, Harrow Research Station, Harrow, ONT)

3. Development of an integrated monitoring capability to track and diagnose aspects of resource quality and sustainability. (Dr. Bruce MacDonald, Centre for Land and Biological Resource Research, Guelph, ONT)

The original level of funding for the research component was $9,700,000 through Mar. 31, 1997. Projects will be carried out by Agriculture and Agri-Food Canada, universities, colleges or private sector agencies including farm groups.

This Research Sub-Program is being managed by the Pest Management Research Centre, Agriculture and Agri-Food Canada, 1391 Sandford St., London, ONT.  N5V 4T3.

Dr. Bruce T. Bowman, Scientific Authority
E-Mail: bowmanb@em.agr.ca

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# PARTITIONING OF SOLUTES FROM AGRICULTURAL FIELDS WITHIN THE HYDROLOGIC SYSTEM AT TWO SITES IN SOUTHERN ONTARIO AND THE SUBSEQUENT IMPACT ON ADJACENT AQUATIC ECOSYSTEMS

**FINAL REPORT**
**April 1994 - April 1997**

**Prepared for:**

**Dr. Bruce McDonald**
**Green Plan Projects Coordinator**
**Agriculture and Agri-Food Canada**
**70 Fountain St.**
**Guelph, Ontario**

**&**

**Dr. Bruce Bowman**
**Scientific Authority**
**Research Sub-Program, Canada-Ontario Green Plan**
**Agriculture and Agri-Food Canada**
**1391 Sandford St.**
**London, Ontario**

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## Executive Summary

Measurements of the hydrologic water balance of two agricultural field sites in southern Ontario (a hillslope comprised of loam soils near Kintore and a clay plain near Woodslee) were undertaken to determine the major transport pathways of contaminants (including nitrate, chloride, and atrazine) below the root zone. As well, a numerical model of two-dimensional water flow which includes tile drains was developed and tested at the Woodslee site. At the Kintore field site only, we also investigated the potential impacts of the contaminants on the biological health of a drainage ditch located near the perimeter of the study field.

The study fields were instrumented with meteorological stations and time domain reflectometry probes to measure potential and actual evapotranspiration, water flow metering systems at all tile drainage outlets and at upstream and downstream ditch stations (Kintore only), and groundwater monitoring wells to measure hydraulic heads below the water table.

To quantify the contaminant flux soil samples were collected from the A horizon and water samples were collected at all tile and drainage ditch monitoring stations and from all monitoring wells. Water and soil samples were analysed mainly for nitrate, ammonium, and chloride content; however, a limited number of samples were analysed for a standard suite of anions and cations, isotopes ($^{15}N$ and $^{18}O$),and atrazine.

Results indicate that at the Kintore site the tile drains are the primary transport path for nitrate to the drainage ditch. Concentration of nitrate in groundwater is highest in the upper slope area. There is good evidence that denitrification in the lower part of the field and beneath an uncultivated strip around the perimeter of the field substantially reduces nitrate concentrations. Groundwater contributes proportionately more chloride than nitrate to the drainage ditch in comparison to tile drains.

Results at the Woodslee clay plain site suggest that up to 20% of infiltration may bypass the tile drains and recharge deep groundwater. Preliminary results from the numerical model agree quite well with the measured quantity of tile drain effleuent; however, the model over-predicts the quantity of runoff and underestimates the amount of deep groundwater flow. One explanation for the discrepancies between model results and measurements is that the point measurements of soil hydraulic conductivity used in the model do not include macro-features such as fractures and are probably too low.

Based on an investigation of biological effects on the drainage ditch at Kintore, each of the biotic indices (EPT richness index, Hilsenhoff BI) showed that water quality declined downstream. The Hilsenhoff BI indicated some organic pollution at all of the study sites in Logan Drain, but water quality was still considered to be good.

## Sommaire

On a mené des travaux de mesure du bilan hydrologique à deux sites agricoles du sud de l'Ontario (flanc de colline aux sols argileux près de Kintore et plaine argileuse près de Woodslee) afin de déterminer les principales voies de transport de contaminants (dont des nitrates, des chlorures et de l'atrazine) au-dessous de la rhizosphère. De plus, on a élaboré, et mis à l'essai au site de Woodslee, un modèle numérique bidimensionnel de l'écoulement de l'eau intégrant des canalisations de drainage. Au site de Kintore, on a étudié les effets potentiels des contaminants sur la santé biologique d'un fossé de drainage situé près du périmètre du site.

Aux deux sites, nous avons installé des instruments météorologiques et des réflectomètres avec indications temporelles afin de mesurer l'évapotranspiration potentielle et effective. Nous avons également installé des systèmes de mesure des débits à toutes les bouches d'évacuation des systèmes de drainage ainsi qu'aux postes du fossé de drainage en amont et en aval (site de Kintore seulement). Enfin, nous avons aménagé des puits de surveillance des eaux souterraines afin de mesurer la charge hydraulique au-dessous de la nappe phréatique.

Pour la quantification du flux de contaminants, des échantillons de sol ont été prélevés dans l'horizon A et des échantillons d'eau à tous les postes de surveillance du fossé et des tuyaux de drainage et dans tous les puits de surveillance. Ces échantillons ont fait l'objet d'analyses visant surtout à doser les nitrates, l'ammoniac et les chlorures. Toutefois, un certain nombre d'échantillons ont été soumis à des dosages d'une série standard d'anions, de cations et d'isotopes ($^{15}N$ et $^{18}O$) et de l'atrazine.

Selon les mesures effectuées dans la plaine argileuse près de Woodslee, jusqu'à 20% de l'eau qui s'infiltre dans le sol peut gagner les nappes d'eau profonde. Il existe une étroite corrélation entre les données préliminaires produites par le modèle numérique et les mesures de l'effluent du système de drainage; toutefois, le modèle surestime la quantité d'eau de ruissellement et sous-estime l'écoulement des nappes d'eau souterraines. Cela est peut-être attribuable au fait que les mesures ponctuelles de la conductivité hydraulique des sols utilisées par le modèle ne tiennent pas compte d'accidents de grande échelle, comme les fractures, et qu'elles sont probablement trop faibles.

D'après les résultats d'une étude des effets biologiques sur le fossé de drainage de Kintore, chacun des indices biotiques (indice de richesse EPT, indice biotique d'Hilsenhoff) indique une baisse de la qualité de l'eau en aval. L'indice biotique d'Hilsenhoff révèle l'existence d'une pollution organique à tous les sites du réseau de drainage de Logan, mais la qualité de l'eau était tout de même considérée comme acceptable.

## Personnel

**Principal Investigators:**

David L Rudolph
Waterloo Centre for Groundwater Research
Waterloo, Ontario
R. Gary Kachanoski
Department of Land Resource Science
University of Guelph
Guelph, Ontario
David Barton
Department of Biology
University of Waterloo

**Project Managers**

Gary W. Parkin
Ricardo Hirata
Anderson L. Ward
Department of Earth Science
University of Waterloo

Kosuke Noborio
Agriculture and Agri-Food Canada
Harrow Research Station

**Graduate Students**

Edwin Cey
Department of Earth Science
University of Waterloo
Neil MacLeod
Department of Biology
University of Waterloo

**Field Assistants**

Ian MacQueen
Mary Anne Hardy
Department of Earth Science
University of Waterloo

**Project Participants**

Craig Merkley
Ian Wilcox
Daryl Nethercott
Upper Thames River Conservation Authority
London, Ontario

Ramon Aravena
Waterloo Centre for Groundwater Research

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# Introduction

## i. Background

Nonpoint or dispersed source groundwater and surface water contamination is a growing concern in rural southwestern Ontario (e.g. Rudolph and Goss, 1993). There have been numerous local and regional-scale studies that have identified agricultural practices as the main sources of contamination (e.g. Gillham and Webber, 1969, Miller, 1979, Ryan, 1994; Svenson, 1996). Although classic hydrologic studies such as Dunne and Black (1970) have identified the major water transport pathways through a small watershed as overland runoff, tile drain flow, and groundwater flow, there are scant studies that have focussed on detailed delineation of water and solute transport pathways at the field scale (e.g. Drury et al., 1996). In particular, it is of critical importance to determine the main transport pathways out of the root zone and the natural processes of attenuation of reactive contaminants such as nitrate. Mounting evidence indicates that the fate of nitrate ultimately depends on the soil biogeochemistry (e.g. Robertson et al., 1996). There is a need to quantify which pathways are important to elucidate the components of the hydrologic cycle most sensitive to contaminant impact and guide remediation strategies.

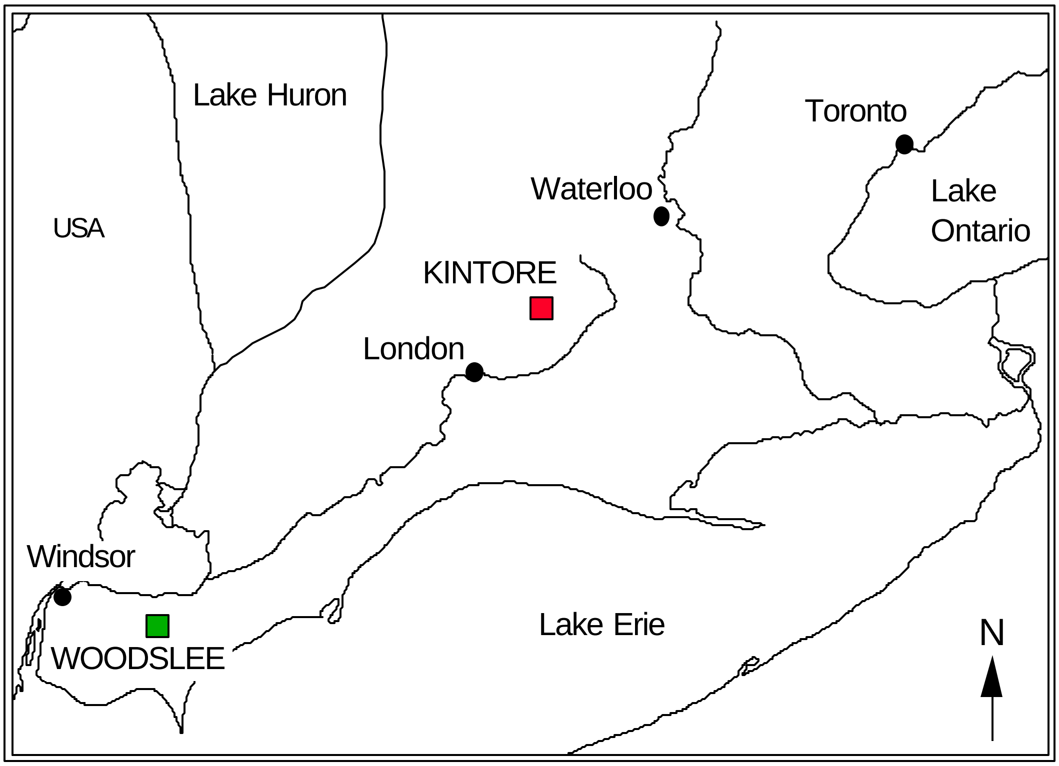
The overall environmental health of an agricultural ecosystem may be best judged by key indicators. For example, the state of the flora and fauna which reside in the adjacent surface water drainage systems may indicate detrimental impacts of agricultural activities. Preliminary work by environmental toxicologists clearly show the sensitivity of fish and other aquatic fauna to nitrate. Similar studies are also available concerning the impact of pesticide exposure to aquatic fauna. At the field-scale, however, very little data is available to assess the impact of agricultural chemicals on the overall aquatic ecosystem in adjacent streams. Of course, the actual concentrations of agricultural chemicals escaping to surface and deep groundwater is an indicator of the efficiency of the farm nutrient management practice. Again very little data exists at the field scale.

In the context of the Green Plan prospectus, the indicators most applicable to our study include: **1.4 Water Contamination Risk, 1.5 Agro-Ecosystem Biodiversity, 1.8 Nutrient Balance, and 1.10 Pesticide Risk.** To sustain our current levels of food production without significant environmental and economic impacts requires a deeper understanding of which indicators are most relevant. Our study is designed to address the questions outlined above within the context of field-scale monitoring. The specific objectives are outlined below.

**ii. Objectives**

Field investigations were carried out over a period of three years at two specific study sites, one on a private farm near Kintore, Ontario and the other near Woodslee, Ontario at the Eugene Whelan Agriculture and Agri-Food Canada (AAFC) Research Station (Fig. 1). These sites were chosen because they differ

Figure 1: Location of Kintore and Woodslee Study Sites in Southern Ontario



physiographically and extensive data bases were available through the Upper Thames River Conservation Authority (UTRCA) and AAFC, respectively. It is our intention that the data collected during this investigation compliment previous work by providing more detailed field-scale studies of water and solute transport through the entire hydrologic system.

Although exact field activities vary at each site, the objectives of the investigation are:

1. quantify the contaminant flux distribution at the field-scale over the annual cycle through a hydrologic water balance focusing on water partitioning between the unsaturated zone, saturated zone, tile drainage system, and the surface-water system.
2. evaluate the significance of spatial positioning within the field with regard to topographic setting, distance from the tile lines, depth to the water table and distance from the surface water drain.
3. document subsurface geochemical conditions that control the nitrification-denitrification processes again in relation to spatial positioning.
4. assess the health of the aquatic ecosystem in the surface water drain through analysis of flora and fauna along the course of the ditch (Kintore site only).
5. employ newly-developed mathematical models to simulate observed processes at the field scale, in order to develop predictive capabilities for agricultural land-use impact assessment (initially Woodslee site only).

**iii. Site Characteristics**

(a) Kintore

The 5.5 Ha field is located in the eastern sub-basin of the Kintore Creek watershed. It is part of a cash crop and hog farm. It is bordered on the eastern and southern sides by the Logan municipal drainage ditch, a natural woodlot on the western edge and another farm field to the north (Fig. 2). The Logan drain is 1 to 2 m wide and has a year-round baseflow and has been artificially straightened by an excavator. The headwaters are located approximately 1 km. to the northeast in an elevated wetland area. The open drain is confined mostly within a wooded area until it is exposed to the agricultural field at the study site. This lack of direct exposure to agricultural activities upstream was an important criterion in selecting the field site.

There is an uncultivated, but cleared vegetated strip about 10 m wide on both sides of the drain as it passes below the study field (Fig. 2). The water quality in the Logan drain has been studied extensively over the past ten years by the UTRCA.

The entire study field was systematically tile-drained about 10 years ago with 10 cm diameter perforated plastic pipe. Depth to tiles ranges from 0.30 to 1.0 m below ground surface. Originally, there were two outlet pipes to the Logan Drain. We have added four new outlets in order to partition the field into five drainage areas with different slope and hydrologic characteristics (Fig. 3).

The field topography is dominated by a simple, very gentle

Figure 2: Location of tile drains, soil pits, runoff and seepage measurement points & surface water sample sites

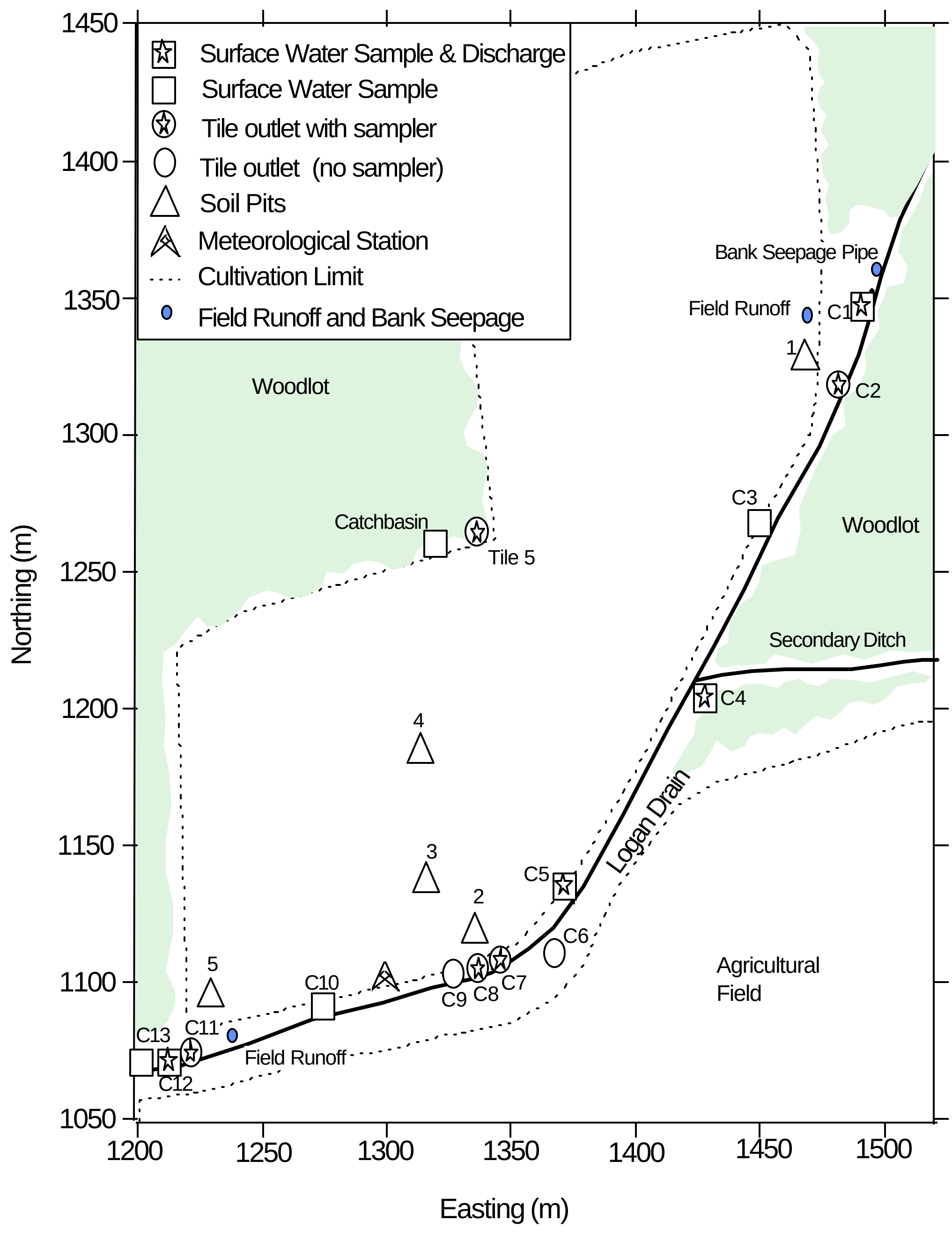


Figure 3: Tile drain outlets, tiles lines and areas drained.

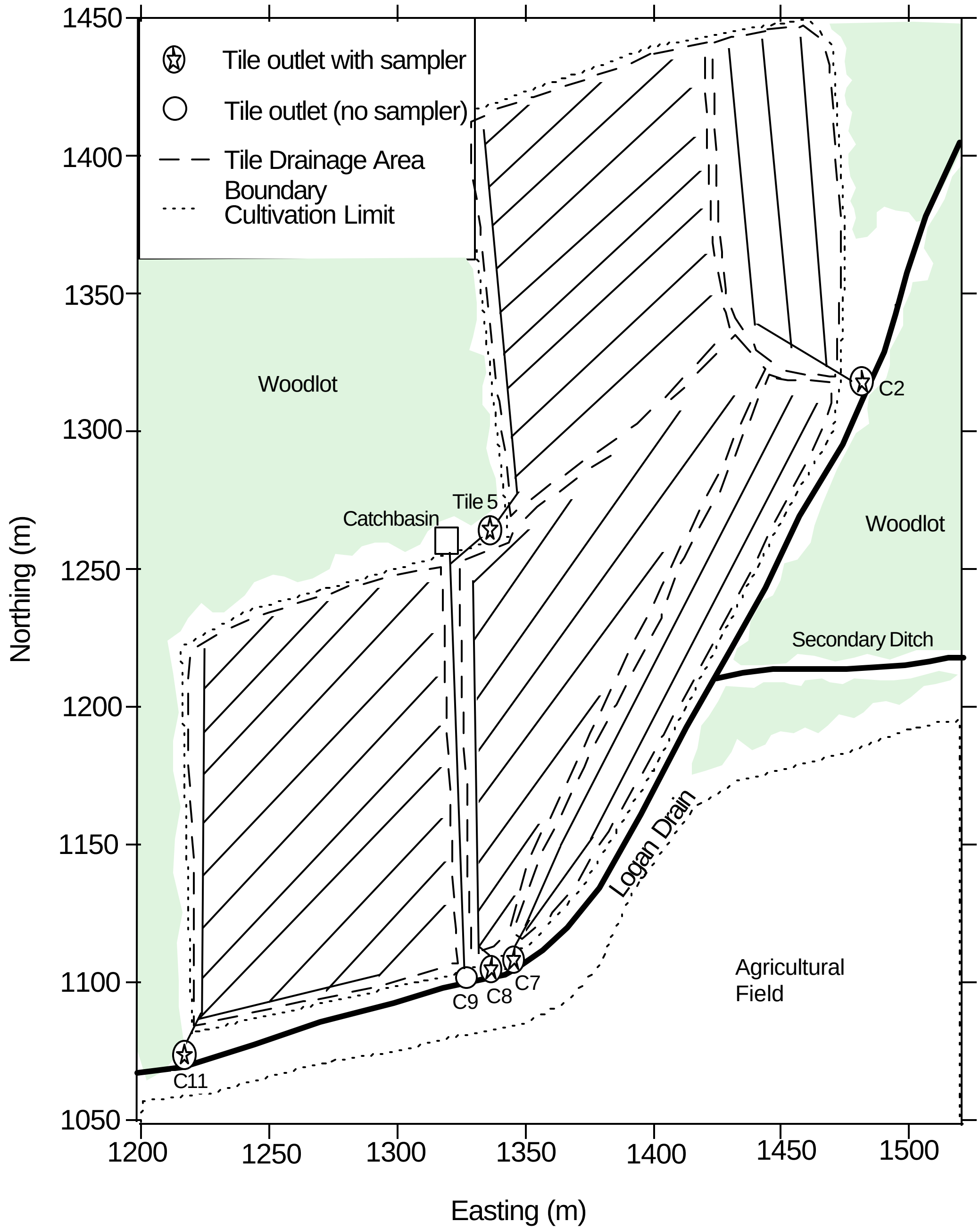


Figure 4: Topography of Field Site

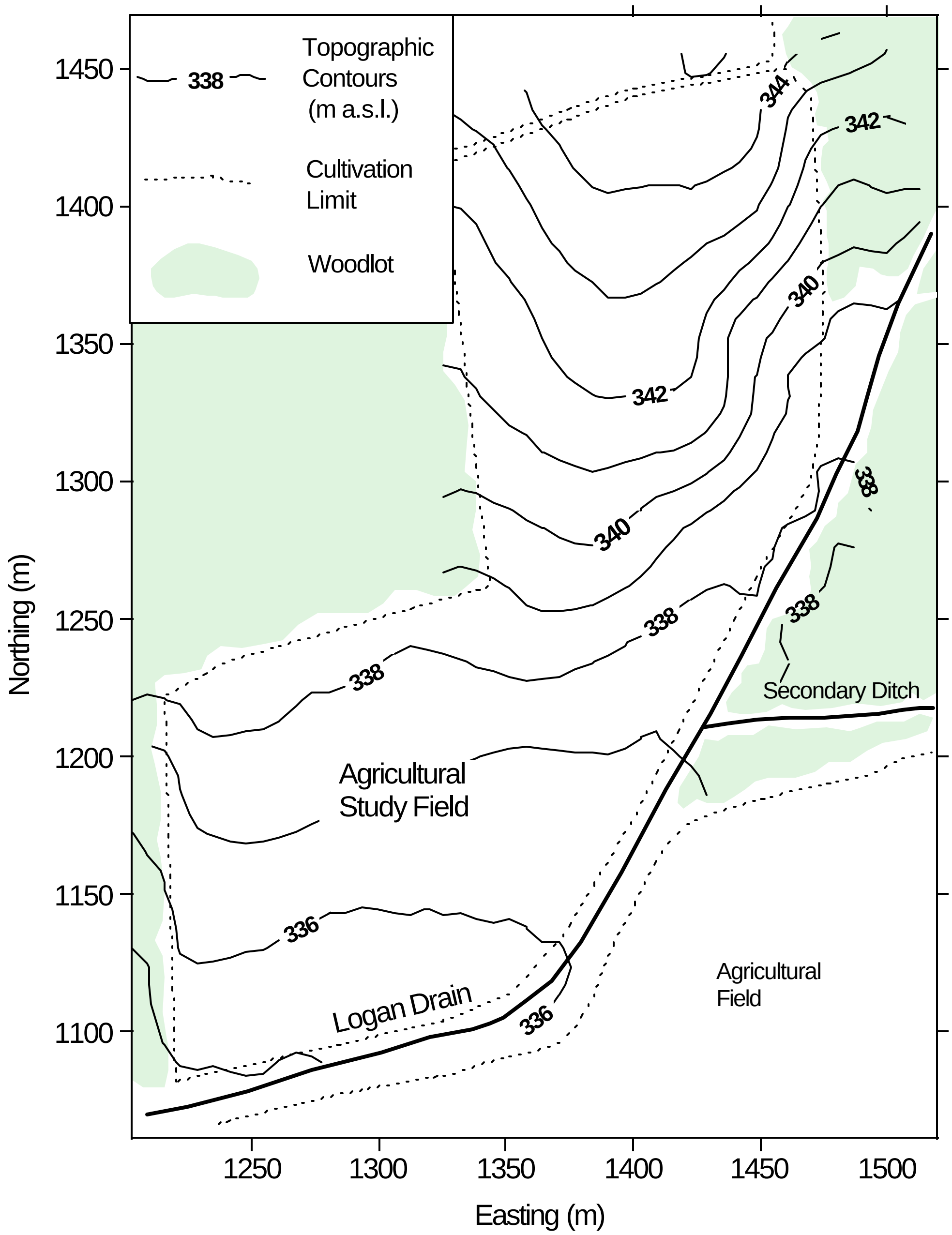
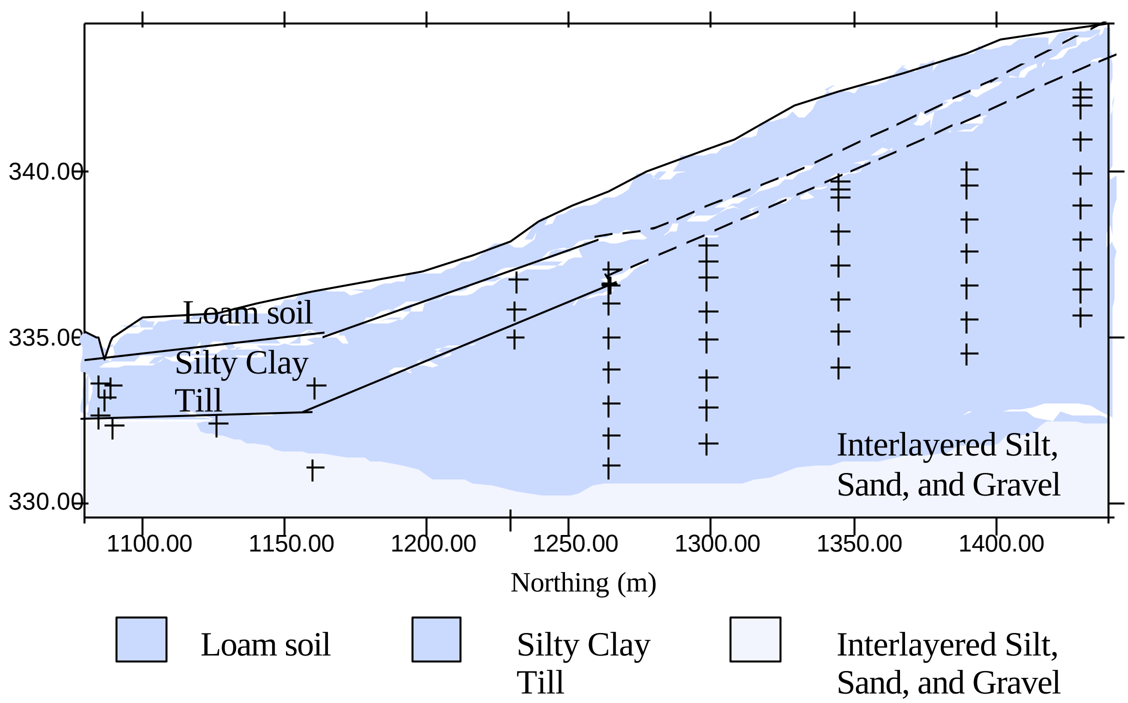


Fig. 5. Geological cross-section from south to north through the study field (The + represents monitoring well screen locations).



slope from north to south with an overall drop in elevation of about 9 m (Fig. 4). In the northeast quadrant there is a more complex gentle slope from west to east. Soil series are Embro and Crombie silt loam (Wicklund and Richards, 1961). The regional surficial geology is dominantly sand-silt till. The site is between the Arva and Dorchester Moraines (Sado and Vagners, 1975). Coring at the field site reveals the geology to be a complex system of waterlain and subglacial diamictons with interlayered stratified drift of probable glacifluvial and glacilacustrine origin (Fig. 5). A dense, grey silty clay till was found consistently between 0.5 to 2 m below the ground surface in the lower part of the field. It confines a more permeable sand and gravel layer below producing local flowing artesian conditions. Indeed hydraulic head values as high as 2 m above ground surface were measured at one location. Core descriptions including soil texture, saturated hydraulic conductivity, and moisture release curves are given in Appendix 1.

The field was seeded with barley, alfalfa, and timothy in the spring of 1994. The barley was harvested in August and liquid swine manure was applied from a mobile tank of capacity 13,600 litres. Similar manure applications occurred again during the spring and fall of 1995 and the spring of 1996. In the late fall of 1995, the alfalfa\timothy crop was killed by herbicide and tilled with a Soil Saver conservation tillage device. The spring of 1996 saw the crop change to corn. A cocktail of atrazine, triazine, dichroma, dimethanamid, and bentazon was applied by a sprayer for weed control. As well, a quantity of solid muriate of potash, phosphate, and liquid ammonium nitrate and urea fertilizers were applied with the seeding and spraying events, respectively.

(b) Woodslee

A detailed description of the physical and hydrological characteristics and instrumentation at the Woodslee site was given in Drury et al. (1996). The site consists of a thick sequence of low permeability glaciolacustrine clays. The local topography is essentially flat. In terms of instrumentation, the only additions pertinent to this study were 16 multi-level groundwater monitoring wells. Each well consists of 6 - 8 sampling levels down to a maximum of 5 m depth. Water levels were recorded on a monthly basis and samples were collected for nitrate, ammonium, and chloride analyses. Measurements of groundwater pH, Eh, electrical conductivity, and dissolved oxygen content were done on a seasonal basis. Soybeans and corn were grown at Woodslee during 1995 and 1996, respectively. Soybeans were planted in the spring of 1995 along with a single application of inorganic nitrogen fertilizer. During 1996, corn was grown at the Woodslee site. Nitrogen fertilizer was applied to the field on two separate occasions. In this study only a water balance was completed at Woodslee.

**iv. Structure of the Report**

We have divided the body of the report into two main sections - one devoted to the Kintore site and the other to Woodslee.

For the Kintore section, parts 1 and 2 contain information on hydrological and geochemical aspects. It includes a description of the relevant instrumentation, results of the water, nitrogen, and chloride balances. Part 3 gives a brief account of herbicide leaching. Part 4 describes the methods and results from the study of the biological health of surface waters.

Part 1 of the section devoted to the Woodslee site contains methods and results of the water balance measurements. Part 2 contains a description and a test of a newly-developed numerical model of unsaturated - saturated water flow with tile drainage. Following the sections on the Kintore and Woodslee sites is a comparison of the water balance results from the two sites. In addition to the main body of the report, Appendix A contains one stand-alone scientific paper ready for submission to a scientific journal. The title of the paper is Efficiency and Accuracy of Different Methods to Measure Potential Evapotranspiration Under Temperate Climatic Conditions.

**I Kintore Site**

**Part 1. Water Balance**

**1.0 Introduction**

A conceptual water-balance equation for a single agricultural field with tile drains is

$$P + G_I + Q_I = ET + G_O + Q_O + \Delta S_s + \Delta S_G + T_O \qquad \textbf{(1)}$$

where P is precipitation in the form of rain and snowmelt, $G_I$ is
the influx of groundwater,$Q_I$ is the influx of surface water, ET is the actual evapotranspiration from the soil moisture, $G_O$ is the outflow of groundwater, $Q_O$ is the outflow of surface water, $\Delta S_S$ is the change in storage of surface water, $\Delta S_G$ is the change in storage of groundwater, and $T_o$ is outflow through the tile drains. The mathematical formulation of equation (1) is trivial; however, the measurement of each component is not nearly as straightforward. An accurate measurement of the hydrologic water balance, even for a monocropped agricultural field, requires an interdisciplinary approach using an array of appropriate instrumentation.

There are numerous examples of field-scale hydrologic water balance investigations (e.g. Dunne and Black, 1970; Wilcock and Essery, 1984; Hudson, 1988; etc.). With a few exceptions, the previous studies have only dealt with one or two components in equation (1). The aim of our study is to quantify all of the terms in equation (1) at the field scale.

The impetus behind our investigation is the growing concern over the negative impact of non-point source contaminants from agricultural sources (Rudolph and Goss, 1992). A water balance calculation is key to identifying the major contaminant transport pathways from cultivated fields to adjacent surface waters and groundwater. The contaminant's residence time and the biogeochemistry of the soil will ultimately determine the effluent concentrations. The field instrumentation and data collection protocols are presented below.

## 2.0 Methodology

### 2.1 Atmosphere

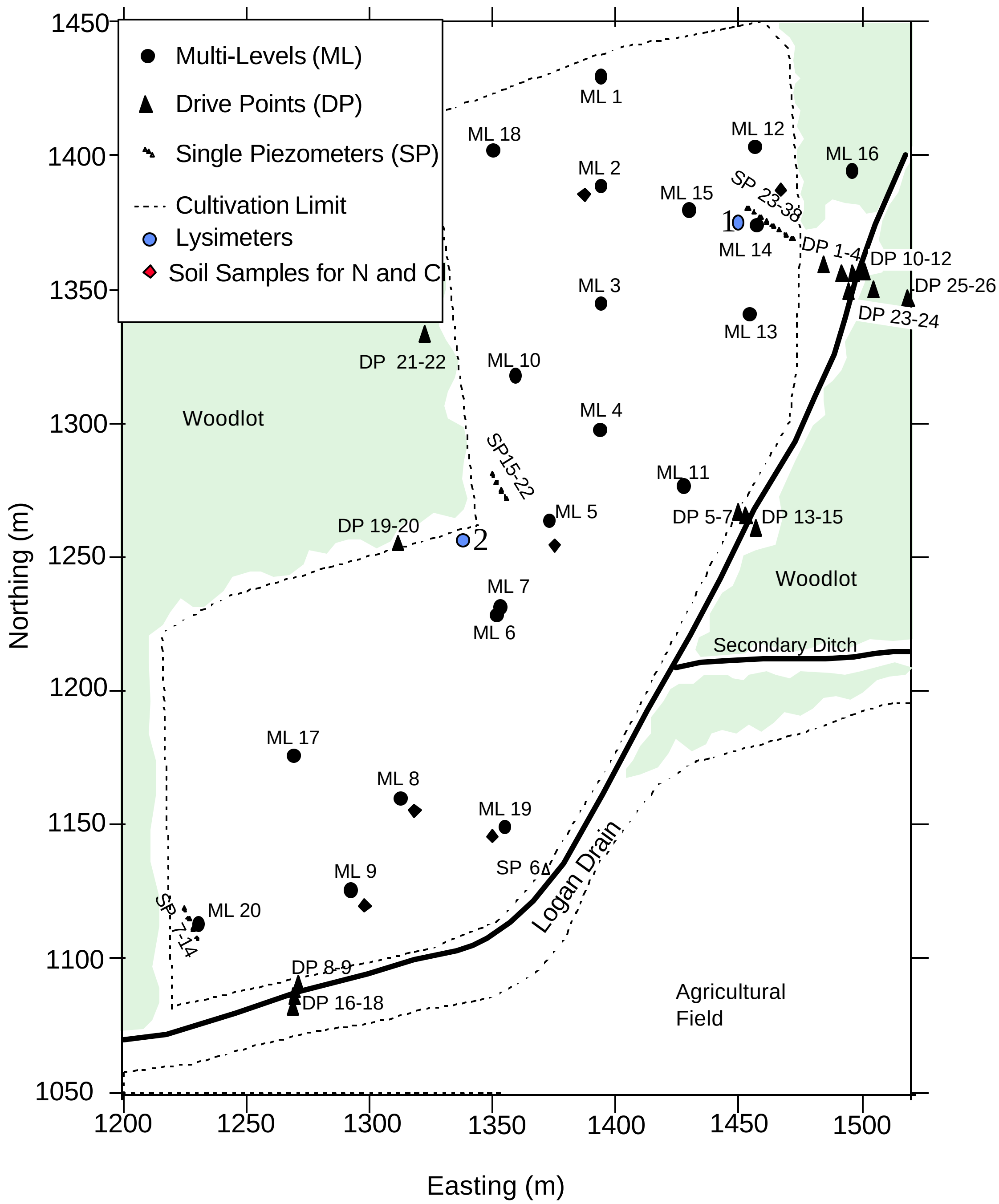
A micro-meteorological station was installed in the uncultivated strip between the study field and the Logan drain (Fig. 2). The met station consists of a cup anemometer to measure wind speed, a silicon pyranometer to measure incoming solar radiation, a combined relative humidity and temperature probe, a tipping-bucket rain gauge, and a Class-A evaporation pan. Appropriate hourly totals or averages were recorded on a Campbell Scientific 21X datalogger system. Two snow stakes were installed at different locations to estimate snow depth. In addition, each month from November to April, we measured snow depth at ten random locations across the field. Snow density was measured at the same time using a gravimetric technique on core samples.

**2.2 Unsaturated Zone**

Thirty-two time domain reflectometry (TDR) probes were installed to estimate soil water content in three transects between tile drains at odd-numbered SP monitoring wells (Fig. 6). The 2 mm diameter stainless steel probes were installed vertically from the soil surface to the bottom of the A and B horizons, approximately 0.20 and 0.40 m below soil surface. The method of Topp et al. (1980) was used to estimate the soil water content. Measurements were taken with a Tektronix 1502 series cable tester throughout the summer months of 1995 and 1996. Measurements were taken weekly during 1995; however, this time interval was determined to be too long to accurately estimate actual evapotranspiration. In the summer of 1996, measurements were taken three times per week.

Twenty-five porous cup tensiometers were installed in conjunction with the TDR probes. Measurements of soil-water potential at 20, 40, 60, and 80 cm depths were recorded with a hand-held electronic pressure transducer at the same time as the TDR measurements of water content. Estimates of the hydraulic gradient from adjacent tensiometers were used to determine if net evapotranspiration or infiltration of water was occurring in the soil profile. Note that lateral unsaturated flow was not considered in this study.

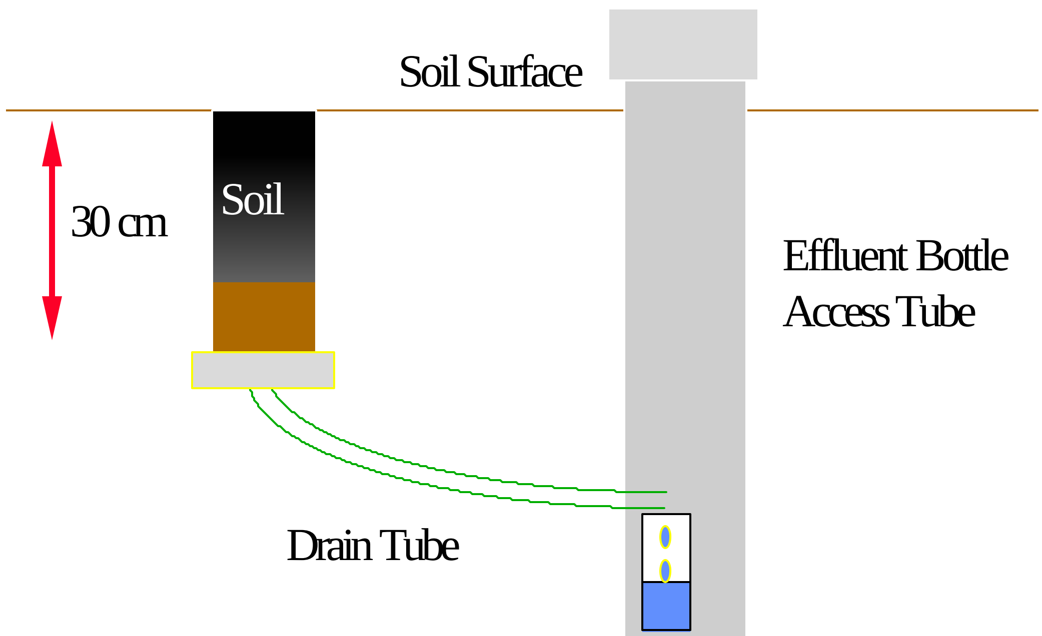
Figure 6: Location of monitoring wells, lysimeters, and soil samples.



In addition to TDR and tensiometers installed in the open field soil, two mini-lysimeters were used to estimate infiltration and evapotranspiration (Figs. 6 and 7). A PVC plastic sewer pipe of 15 cm diameter was driven 30 cm into the soil profile in order to fill each lysimeter with undisturbed surface soil. Next, the soil core was removed and trimmed of all excess soil and roots. Finally, a plastic cap with a drain tube was affixed to the bottom of the core. The drain tube was connected to a plastic bottle which was housed in a second PVC pipe beside the lysimeter. Drainage effluent was collected following a few rainfall events and analyzed for nitrate, ammonium, and chloride. Details concerning water and soil sampling protocols and methods of analysis are given in Appendix 3.

Soil samples were taken in August, 1994 on a 10 m grid at four depth intervals (0 - 15, 15 - 30, 30 - 45, and 45 - 60 cm) across the entire field. During 1995 and 1996 fewer samples were taken and near existing instruments on the field. Sample locations are given in Fig. 6 for 1995 and 1996. Soil-water extracts were analyzed for nitrate, ammonium, and chloride.

Fig. 7. Schematic diagram of a mini-lysimeter installed at Kintore.



**2.3 Saturated Zone**

A total of 201 groundwater monitoring points were installed over the course of the field investigation. Multi-level wells were installed initially in two perpendicular transects across the middle of the field (Fig. 6). Fig. 8 shows well locations below ground surface along the main north-south transect. A 15 cm diameter auger hydraulic drilling rig was used to create bore holes for the multi-level monitoring wells. Each multi-level well
consisted of a 2.5 cm diameter PVC plastic centre stock with between 4 and 7 additional 1.25 cm diameter polypropylene monitoring wells fastened along the centre stock at progressively shallower depths (Fig. 9). The multi-levels were completed to an average depth of 7 m below the water table. Coarse-grained well sand was placed around each screen and bentonite plugs were placed between screens.

Screen lengths were 15 cm. The bentonite plug ensures that there is no direct hydraulic connection between screens. Other single- level monitoring wells were installed with a track-mounted, portable hydraulic rig or by physically driving a steel point into the ground with an electric jack hammer.

The position of the top of each well was level-surveyed with a Leica total station. The cartesian coordinates of each well are given in Appendix 4. Water levels were measured manually with a standard battery operated water-level tape on a monthly basis.

Fig. 8. Location of monitoring wells below ground surface along main north-south transect.

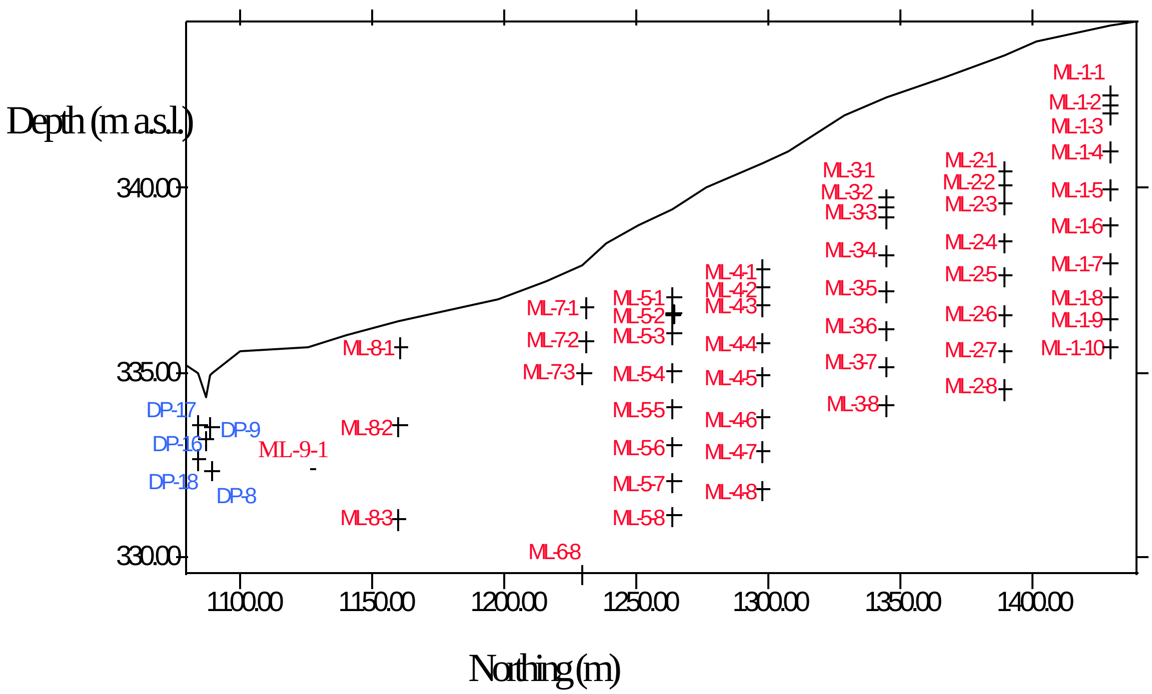
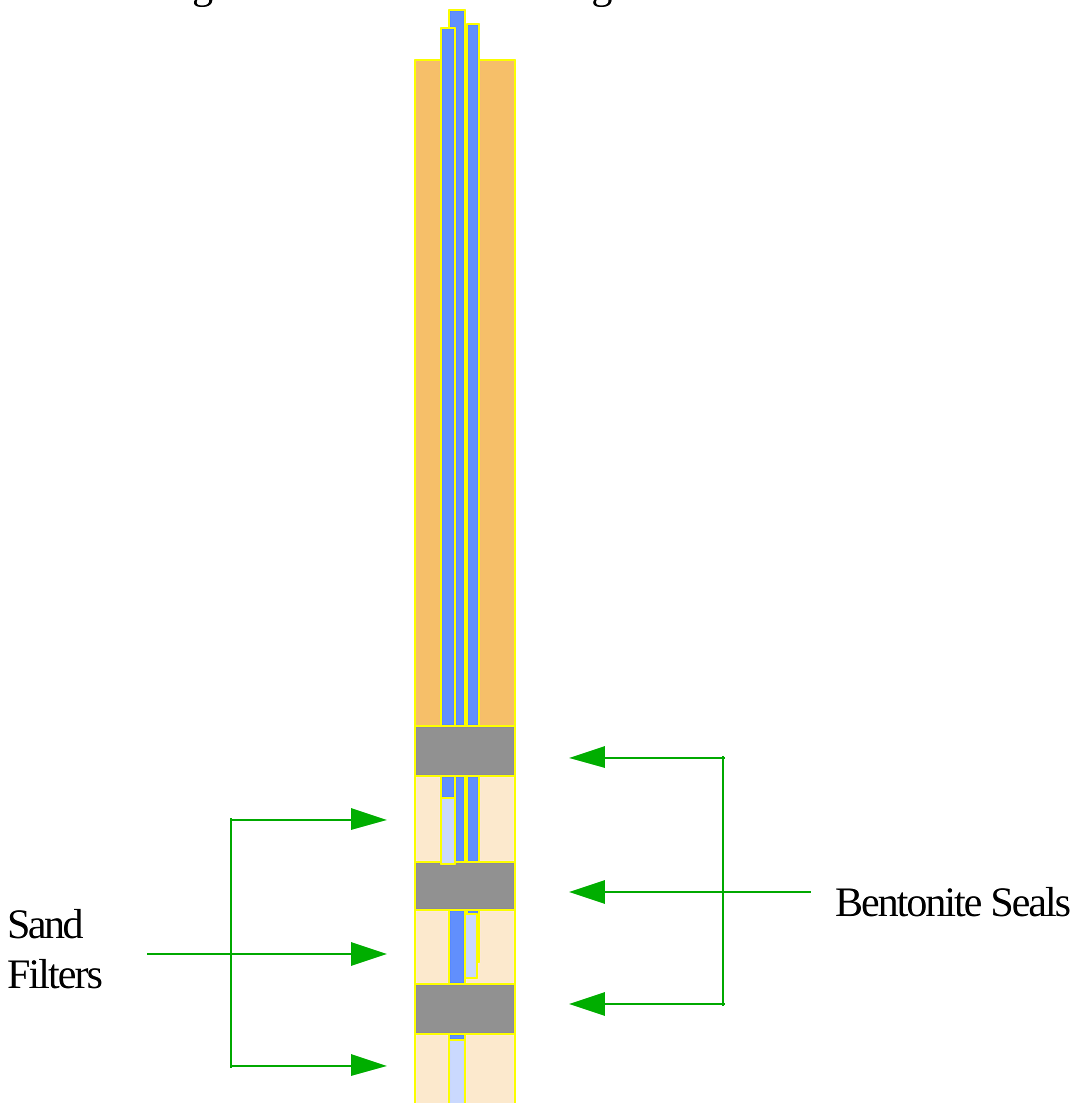


Fig. 9. Multi-level monitoring well.



The measurements of water level were converted to hydraulic head and contoured as plan views or vertical cross sections. Hydraulic head data was used to determine approximate directions of groundwater flow below the tile drains.

Following each episode of water level measurement, the wells were purged in preparation for groundwater sampling. A peristaltic pump was used to remove water from each well. The sampling protocol depended on the chemical species of interest (Appendix 3).

**2.4 Surface Waters and Tile Drains**

The water level in each tile drain and four points along the main and secondary drainage ditches were measured every 10 minutes with a float - potentiometer - datalogger system (Figs. 2 and 3). As well, water levels in overland flow (ie. runoff) monitoring stations were measured with the same system at two locations (Fig. 2). A low berm constructed about 10 years ago at the boundary between the field and uncultivated strip prevented most of the field runoff waters from entering the Logan drain. However, the runoff monitoring station near the southwest corner of the study field was located in a breach of the low berm. Runoff from the field into Logan drain was concentrated at this point. The runoff monitoring station near the northeast corner of the field was located at the base of the steepest slope on the study field.

Water levels measured at each station were converted to volume flux via stage-discharge relationships measured for each tile outlet and ditch point. A standard velocity-area stream gauging method (e.g. Dingman, 1994)was used to generate the stage -discharge curves. Water velocity was measured by personnel from UTRCA with a magnetic-particle velocity meter in each tile and at six-tenths the water depth and 10 cm increments across the ditch. Measurements were taken on a weekly basis and during major storm events. The product of water velocity and cross-sectional area gave an estimate of water discharge at each tile and surface water monitoring station.

Water samples for background analysis of chloride, nitrate, and ammonium were collected on a weekly basis from each tile outlet and seven points along the drainage ditches by UTRCA staff (Fig. 2). As well, seven Isco automatic water samplers were deployed to sample water during rainfall or snow-melt associated events (Fig. 2). For the most part, samples were taken at a two-hour interval and only samples from major events were saved for chemical analysis. In addition, the bulk electrical conductivity and the $O^{18}$ content over the course of a few precipitation events were measured for hydrograph separation analysis (c.f. Sklash and Farvolden, 1979).

## 3.0 Results

### 3.1 Water Balance

#### 3.1.1 Potential and Actual Evapotranspiration

Hourly summaries of climatic data collected at the MET station for each month in 1995 and 1996 are given in Appendix 5. Monthly total amounts of precipitation, and potential and actual evapotranspiration are given in Figs. 10 and 11 and a summary is presented in Table 1.

| | Precipitation (mm) | Actual Evapotranspiration (mm) | Surplus (mm) |
|---|---|---|---|
| 1995 | 1029.8 | 403.3 | 626.5 |
| 1996 | 1135.0 | 257.9 | 877.1 |

Table 1. Soil water budget for Kintore during 1995 and 1996.

Estimates of potential evapotranspiration for a well-watered alfalfra crop were calculated with the Penman-Monteith combination method using hourly averages of the measured climatic parameters (Monteith, 1965). Then an estimate of the actual evapotranspiration (AE) was computed using appropriate crop reduction coefficients from Jensen et al. (1989). Note that potential and actual evapotranspiration are nearly the same for 1995 (alfalfa crop) and that actual is much less than potential evapotranspiration for 1996 (corn crop).

We have also performed a similar comparison of other methods to estimate AE for the conditions at Kintore (Appendix A). All calculations of AE have been summarized in Appendix 6.

Fig. 10. Monthly total precipitation (PPE), potential evapotranspiration (PE), and actual evapotranspiration (AE) for the Kintore site, 1995.

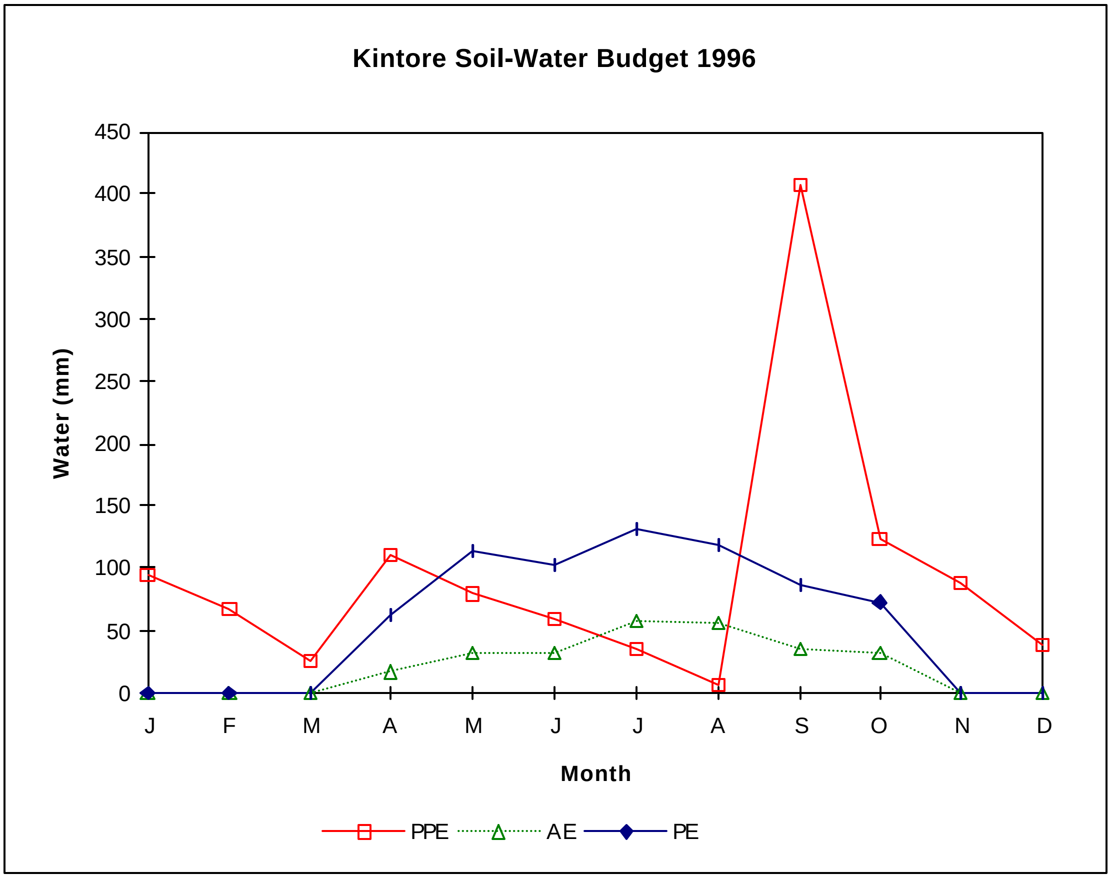
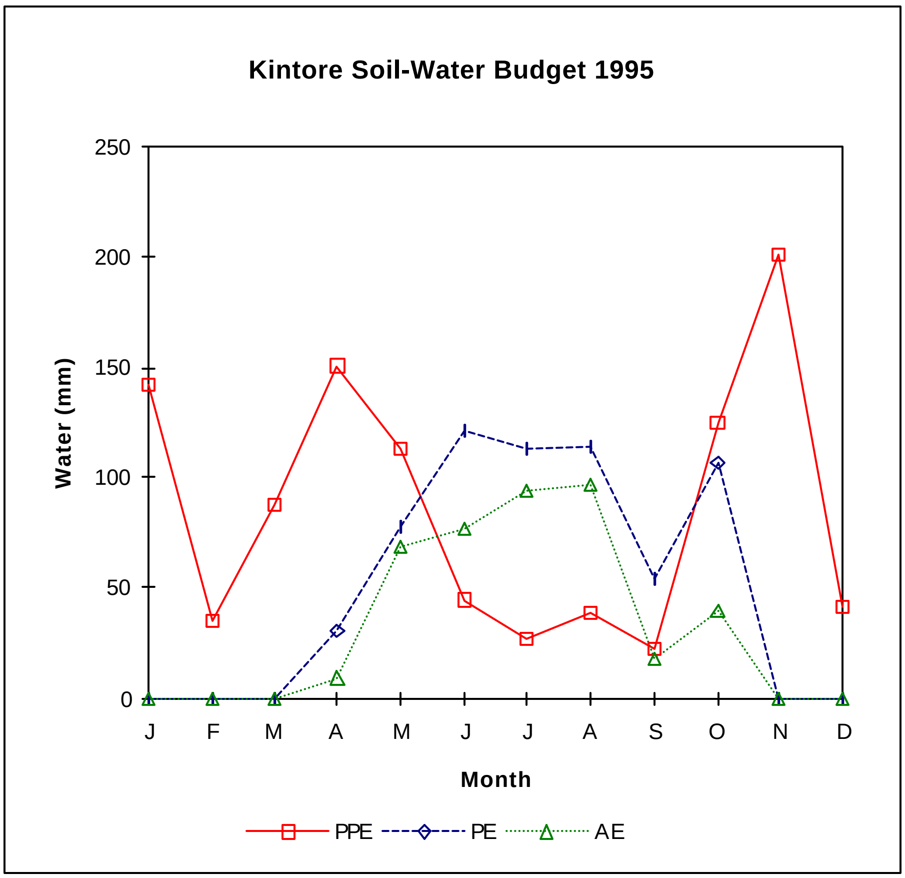


Fig. 11. Monthly total precipitation (PPE), potential evapotranspiration (PE), and actual evapotranspiration (AE) for the Kintore site, 1996.



### 3.1.2 Unsaturated Zone

The fluctuation in water content near the soil surface measured by TDR during the summers of 1995 and 1996 is given in Figs. 12 and 13. As anticipated, soil water content is generally lowest near surface in July and August; however, rainfall events temporarily restore water storage levels. Fig. 14 compares cumulative evapotranspiration calculated by the Penman-Monteith method with that calculated by average measurements of water content by all TDR probes during the summer of 1996. Agreement between the two methods is excellent until about Julian day 210 (July 29). There are two possible explanations for the discrepancy after day 210. It is possible that the corn roots extract water from below 0.40 m following day 210 or the soil is dry enough to limit the evapotranspiration and the Penman-Monteith results are too high. Tensiometer measurements of hydraulic head and gradient in the unsaturated for 1996 support the former hypothesis since measurements of the hydraulic gradient indicate that there was upward flow of water between days 215 and 255(Fig. 15). Therefore, the total AE calculated by the Penman-Monteith is considered to be accurate as given in Table 1.

Fig. 12. TDR measurements of water content during 1995. Note that A and B refer to measurements over the A and B soil horizons.

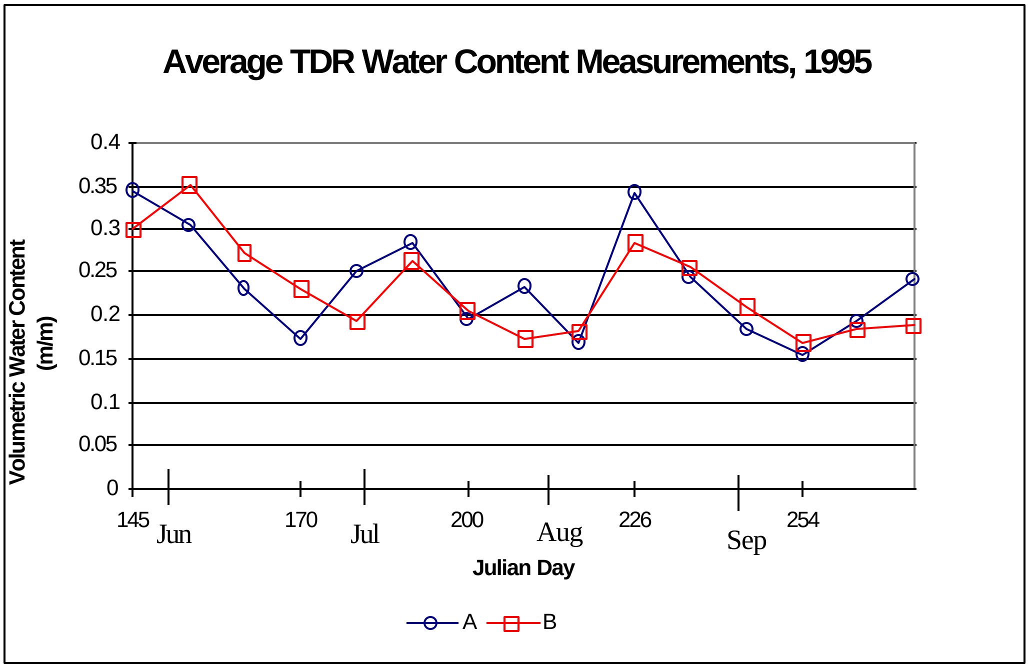


Fig. 13. TDR measurements of water content during 1996. Note that A and B refer to measurements over the A and B soil horizons.

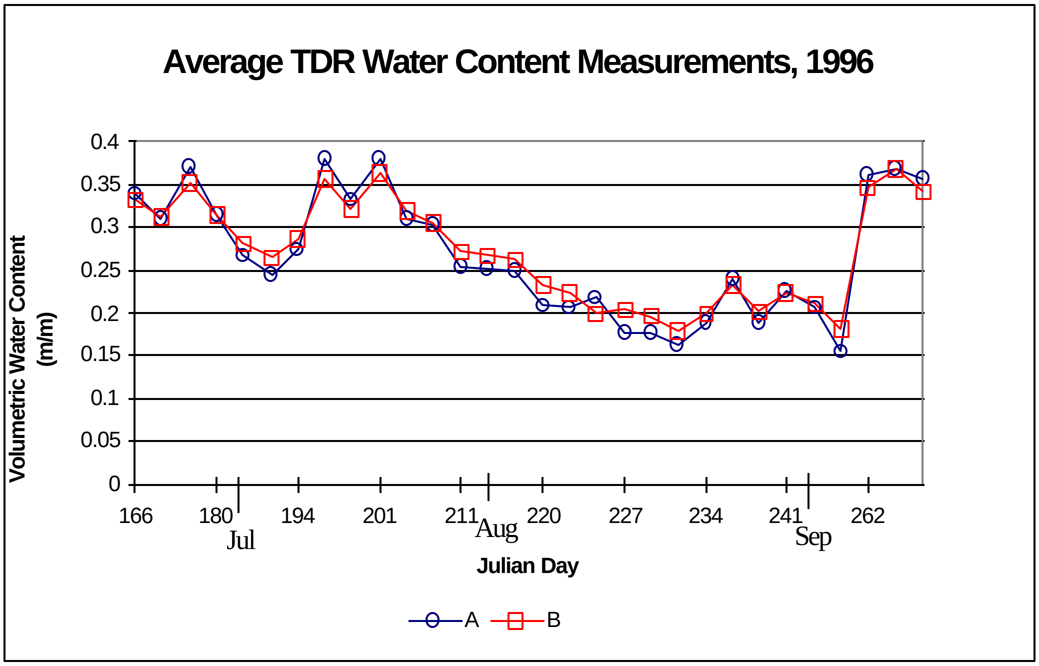


Fig. 14. A comparison of cumulative actual evapotranspiration calculated by the Penman-Monteith method and actual evapotranspiration calculated by TDR measurements of near-surface soil water contents during 1996.

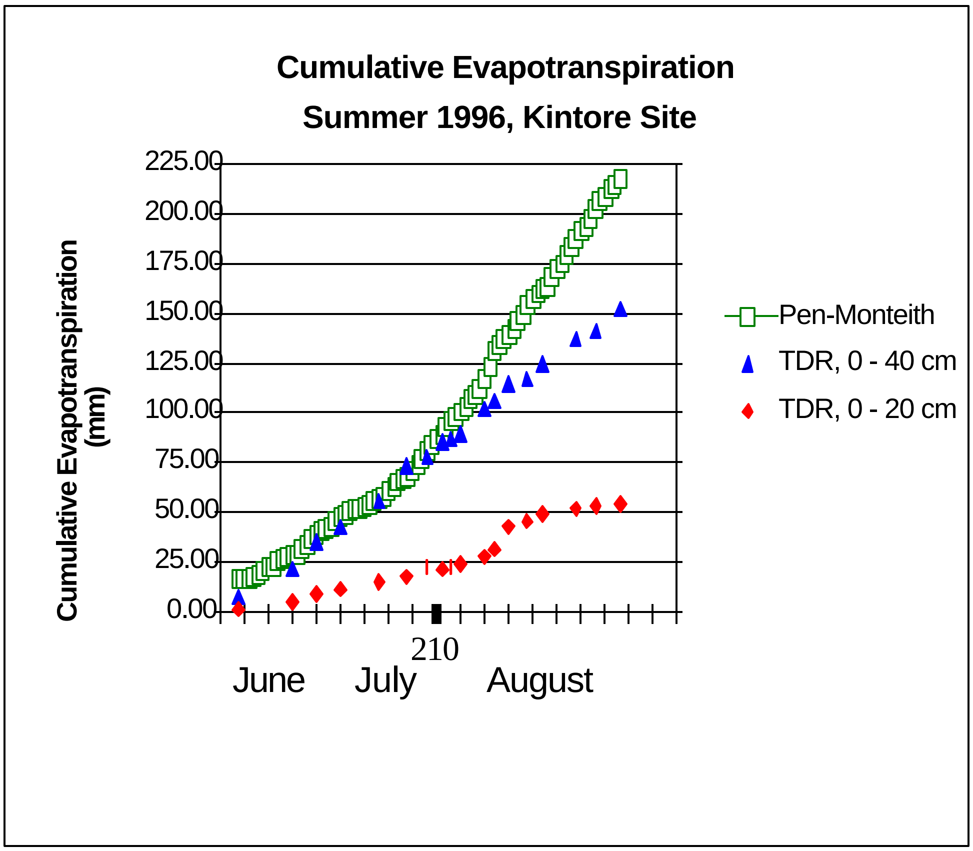
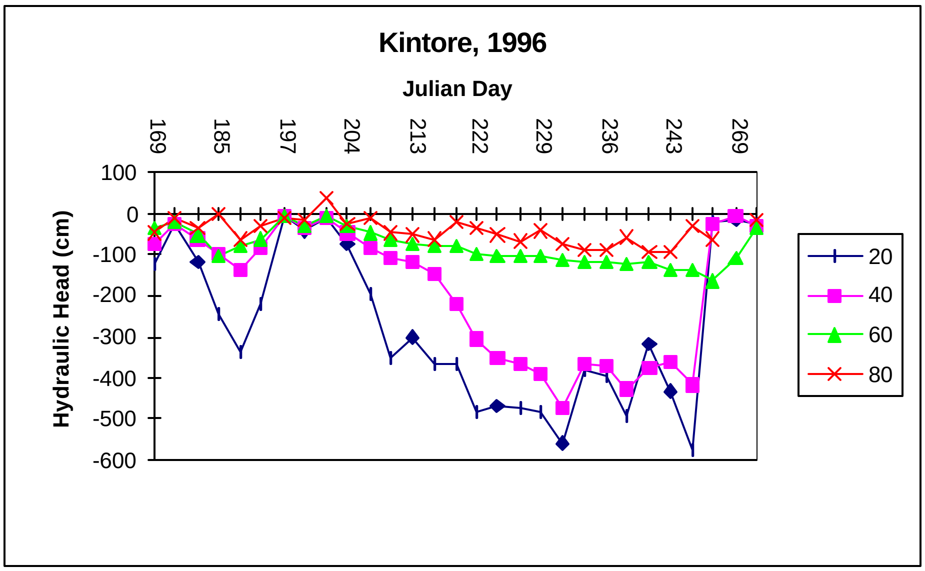


Fig. 15. Tensiometer measurements of hydraulic head and gradient during 1996.



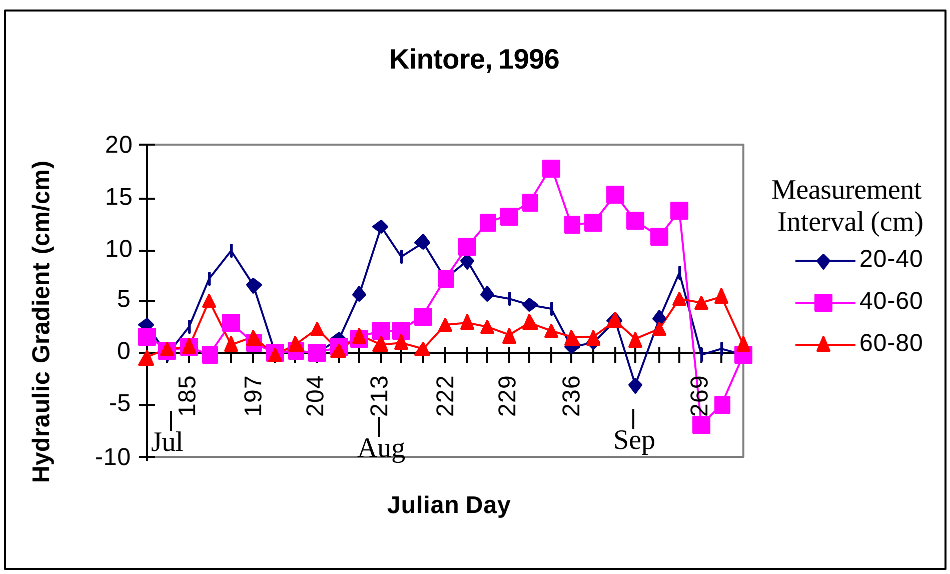



Fig. 16. Quantity of drainage effluent collected at lysimeter 2 during 1995.

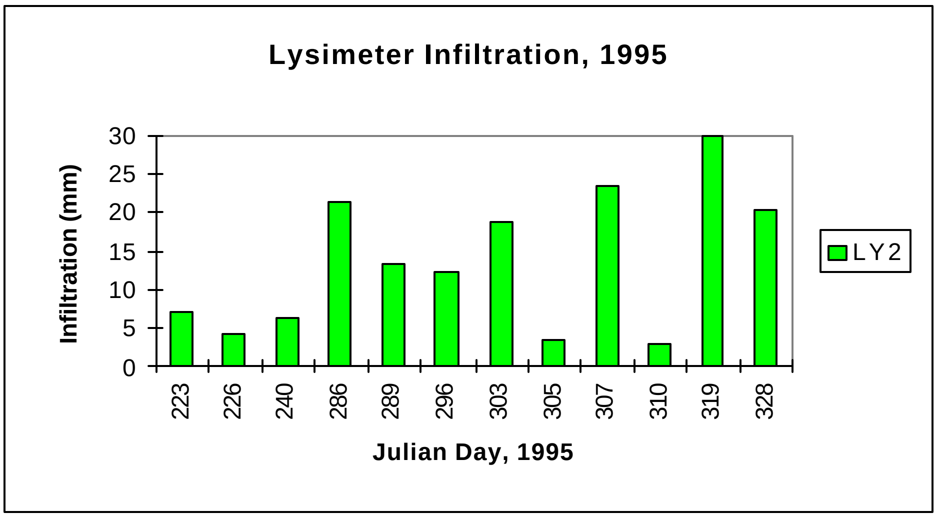
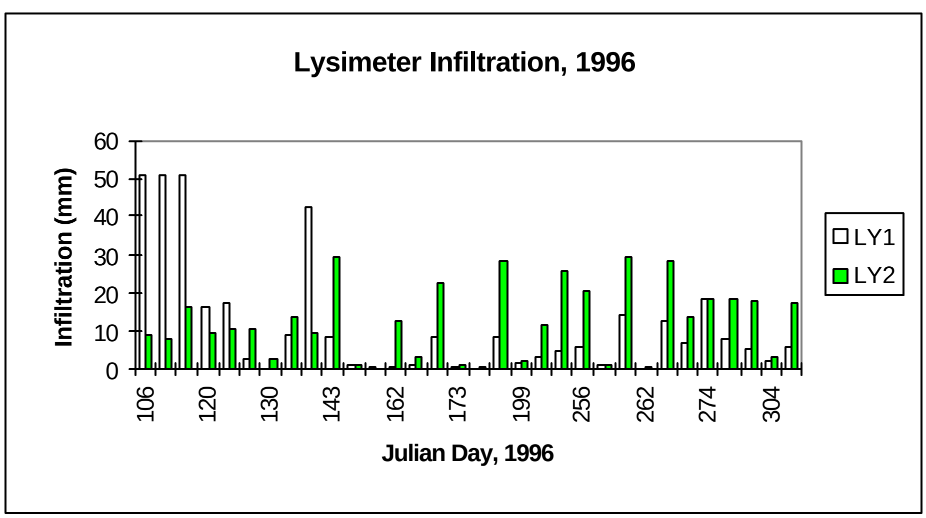



Fig. 17. Quantity of drainage effluent collected at lysimeters 1 and 2 during 1996.



Estimates of infiltration during 1995 (lysimeter 2 only) and 1996 based on the volume of drainage effluent collected by the lysimeters are shown in Figs. 16 and 17. Unfortunately, the lysimeters were not functional year-round due to freezing of water in the collection bottles. The average amounts of effluent collected at 30 cm depth by the lysimeter drains was 163 and 375 mm during 1995 and 1996, respectively. Note that these values of infiltration only represent 26 and 43 % of the surplus water given in Table 1. Potential explanations for this discrepancy include the fact that the collection bottles were frozen during the winter and the loss of soil structure inside the lysimeters. Loss of soil structure would reduce the potential for bypass flow between peds and give a lower overall amount of infiltration.

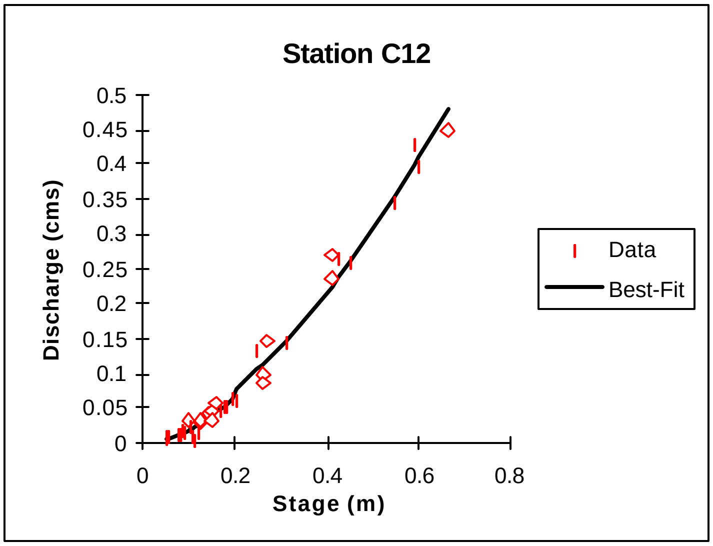
### 3.1.3 Tile Drain Discharge and Surface Runoff to Logan Drain

Examples of best-fit stage-discharge rating curves for surface water monitoring station C12 (see Figs. 2 and 3 for the locations of all surface water and tile drain monitoring stations) and tile drain monitoring station C11 are given in Fig. 18. The remaining rating curves for ditch and tile stations are given in Appendix 7. A two-parameter power formula was fit to the measurements of stage and discharge supplied by the UTRCA,

$$Q = aS^{b}$$

where Q ($m^3s^{-1}$) is discharge, S (m) is stage and a and b are the fitting parameters. For each surface water gauging station, separate curves for low flow and high flow conditions were

Fig. 18. Best-fit rating curves for ditch station C12 and tile station C11.



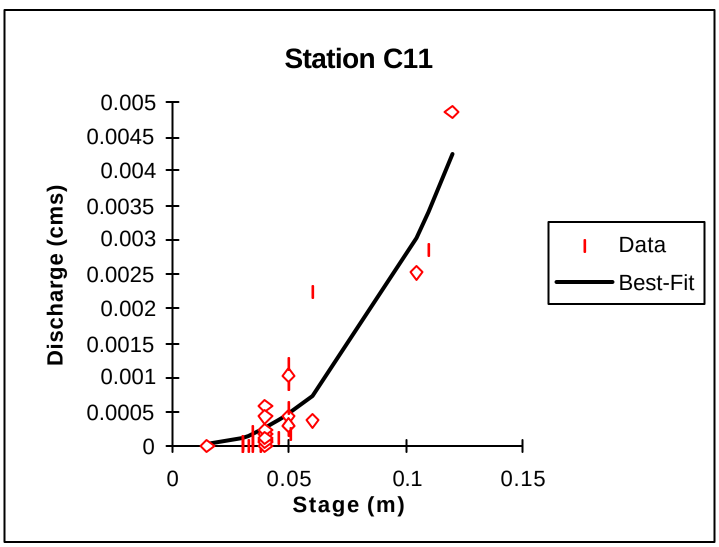



Fig. 19. Monthly total discharge measured at ditch stations C1 (upstream) and C12 (downstream).

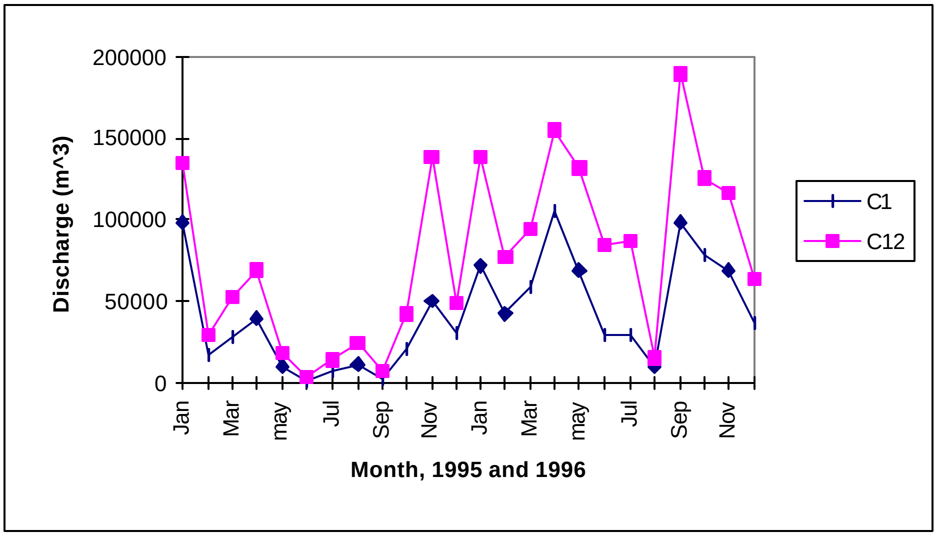
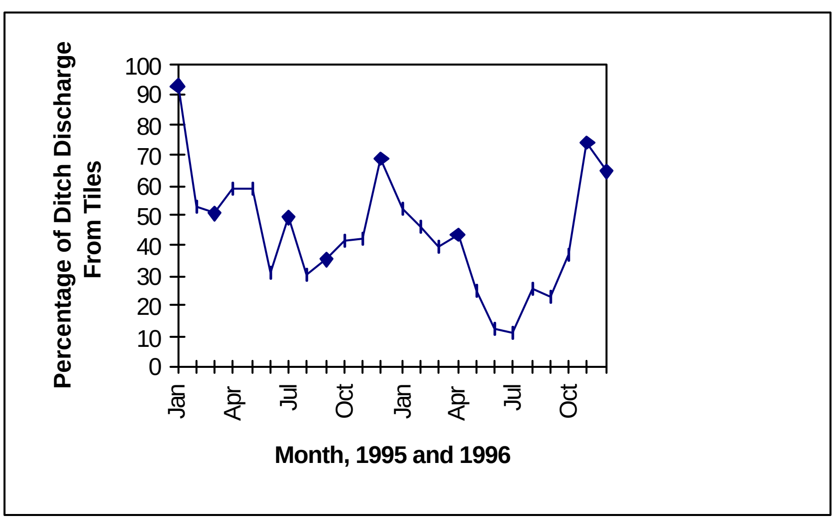



Fig. 20. Monthly summary of contribution of tiles and secondary ditch to increase in discharge between ditch stations C1 and C12.

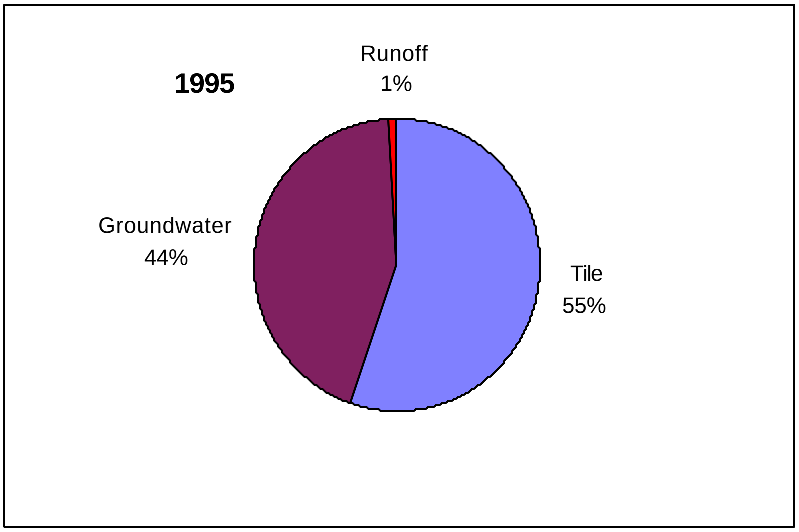


necessary. In general, the simple power formula appears to be provide an acceptable fit to the data.

The increase in ditch discharge between stations C1 and C12 (Fig. 6) is given in Fig. 19. Without exception the amount of discharge increases between stations C1 and C12. In Fig. 20, we present a monthly summary of the percentage contribution by all tile drains to the increase in ditch discharge between ditch stations C1 and C12. On average, tile drains and the secondary ditch contribute only about 45% of the increase in ditch discharge between C1 and C12. Hourly measurements of discharge at all monitoring stations is given in Appendix 8. Runoff measured at the breach in the berm near the southwest corner of the study field is a minor component (Fig. 21). The remainder of the water that contributes to the increase in discharge between C1 and C12 between high-flow events must be groundwater.

The amount of runoff measured at the breach in the berm must be included in the water balance equation as $Q_O$, the surface-water output from the study field. Note that $Q_I$, the surface-water input into the study field is minimal because most of the overland flow from the field to the north is diverted into a ditch which originates near the northwest corner of the study field. The ditch is situated between the study field and the northwestern woodlot and diverts any overland flow from the woodlot into the catchbasin shown on Fig. 3. Water from the catchbasin empties into the Logan ditch at C9 (Fig. 3). This quantity of water was measured using a float\datalogger system installed in the catchbasin. It is included as tile drain discharge.

Fig. 21. Relative contributions of tiles (including the secondary ditch), runoff measured at the breach near the southwest corner of the field, and groundwater to increase in ditch discharge between C1 and C12.



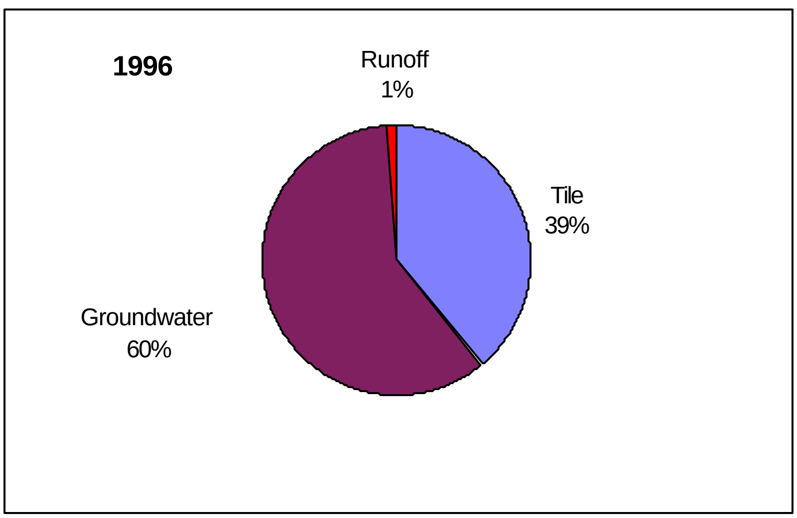



Fig. 22. Monthly summary of discharge from tiles draining the study field (monitoring locations are indicated on Fig. 6).

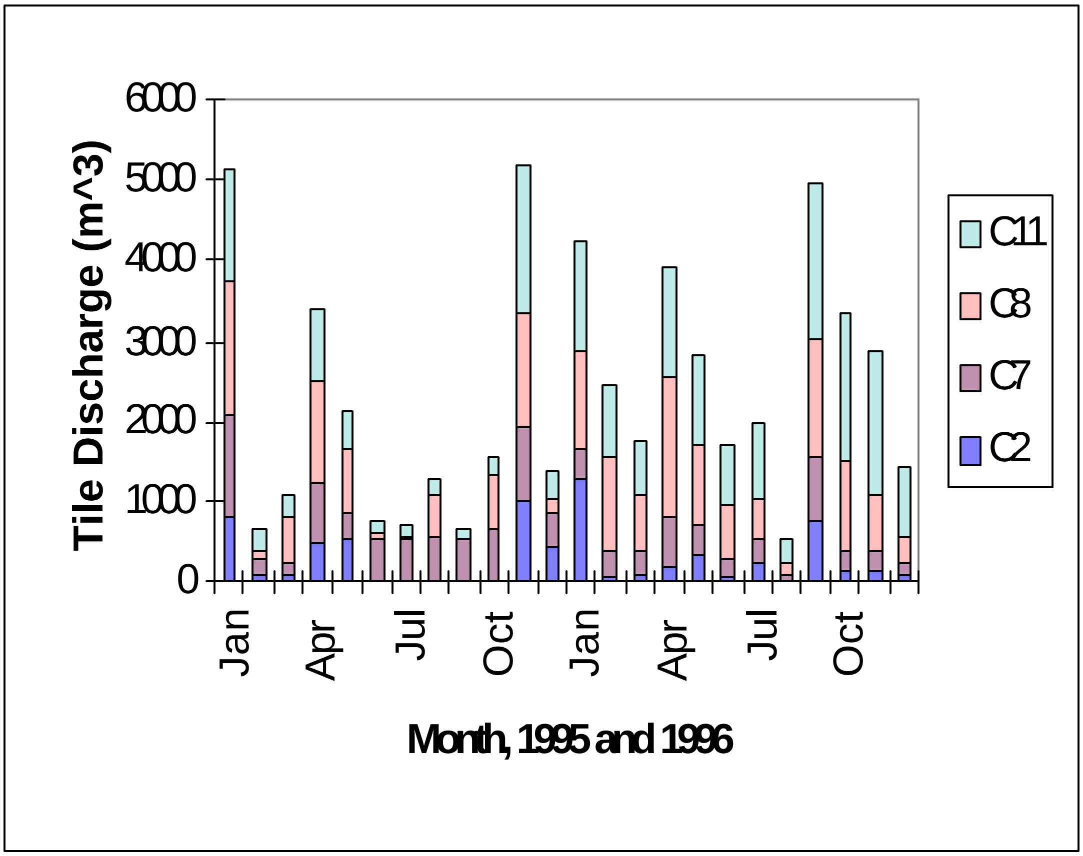


**Table 2.** Monthly summary of quantity of tile drainage in mm from each drainage area on the study field.

| Month | C2 | C7 | C8 | C11 |
|---|---|---|---|---|
| Jan, 1995 | 164 | 168 | 109 | 113 |
| Feb | 15 | 31 | 4 | 24 |
| Mar | 18 | 21 | 37 | 23 |
| Apr | 95 | 99 | 87 | 71 |
| May | 104 | 46 | 51 | 40 |
| Jun | 1 | 68 | 8 | 12 |
| Jul | 0 | 67 | 4 | 10 |
| Aug | 0 | 76 | 35 | 15 |
| Sep | 0 | 72 | 0 | 9 |
| Oct | 0 | 91 | 42 | 20 |
| Nov | 199 | 122 | 96 | 149 |
| Dec | 80 | 73 | 11 | 26 |
| **1995 Total** | **677** | **922** | **483** | **512** |
| Jan, 1996 | 257 | 51 | 80 | 109 |
| Feb | 10 | 42 | 80 | 73 |
| Mar | 18 | 39 | 48 | 51 |
| Apr | 38 | 83 | 115 | 111 |
| May | 71 | 49 | 66 | 89 |
| Jun | 8 | 29 | 46 | 60 |
| Jul | 49 | 39 | 35 | 74 |
| Aug | 0 | 15 | 7 | 26 |
| Sep | 155 | 104 | 97 | 158 |
| Oct | 26 | 36 | 74 | 149 |
| Nov | 19 | 34 | 47 | 143 |
| Dec | 19 | 18 | 23 | 65 |
| **1996 Total** | **681** | **539** | **716** | **1108** |

**Table 3.** Hydrological characteristics of each drainage area on the study field.

| | | C2 | C7 | C8 | C11 |
|---|---|---|---|---|---|
| Area (Ha) | | 0.5 | 0.75 | 1.5 | 1.25 |
| Slope Class and Percent | | moderate, 9 | very gentle, 3 | gentle, 5 | very gentle, 3 |
| Average Depth to Water Table (m) | Winter | 1 | .75 | .75 | .75 |
| | Spring | 1.25 | .75 | .75 | .5 |
| | Summer | 1.5 | 1.0 | 1.0 | .5 |
| | Fall | 1.25 | 1.0 | 1.5 | .75 |
| Average Hydraulic Gradient | Winter | .075 | -.4 | 0 | -.25 |
| | Spring | .05 | -.075 | .075 | -.1 |
| | Summer | -.05 | .175 | .175 | 0 |
| | Fall | -.05* | .3 | .45 | -.1 |
| Soil Texture | | loam | loam | silt loam | loam |

* A negative value indicates upward flow.

A monthly summary of the total discharge measured at tile drain monitoring stations which drain the study field is given in Fig. 22. Data from Fig. A22 is given in Table 2 as water depth equivalents for each tile drainage area. To account for differences in the measured amounts of tile drainage effluent, we give a summary of the drainage area, topography, average shallow hydraulic gradients, average depth to the water table, and soil texture at tile level for each drainage area in Table A3. The relationships between data in Tables 2 and 3 will be discussed in the overall summary presented at the end of the Kintore section following presentation of groundwater flow data. Discharge hydrographs measured at all monitoring stations during high-flow events afford the opportunity to estimate the contributions of event and pre-event water to the increase in discharge at each station. Event water refers to the proportion of the total discharge during an event that originates from the current rainfall or snowmelt event. Part of the event water is from overland flow (ie. runoff) and part is from interflow or shallow groundwater flow. Hydrographs are readily separated into event and pre-event waters using a mass balance approach (Sklash and Farvolden, 1979). All that is required are measurements of the concentrations of a nonreactive tracer in the rain or snow pack and the in the ditch or tile drain before and during the event.

In a separate study, the contributions of groundwater to streamflow were measured along a section of stream during baseflow conditions and during rainfall events (Cey et al., 1998a). Four techniques were used to estimate the contribution of groundwater to the stream along a 450 m reach (three during baseflow and one during stormflow conditions).

Under baseflow conditions, streamflow measurements using the velocity-area technique indicated the net groundwater flux to the stream during the summer months was ~10 mL/s/m. Hydrometric measurements (i.e. hydraulic gradient and hydraulic conductivity) taken using mini-piezometers installed in the sediments beneath the stream resulted in net groundwater flux estimates that were four to five times lower. The large spatial variability in hydraulic gradient and hydraulic conductivity measurements was likely a result of the geologic heterogeneity at the site. Seepage meters failed to provide reliable measurements of water flux into or out of the stream. Both the hydrometric and seepage meter estimates were considered unreliable because they did not account for the observed increases in streamflow.

Hydrograph separations were conducted using $^{18}O$ isotopes and electrical conductivity on two large rainfall events with different antecedent moisture conditions in the catchment. Both events showed that pre-event water (generally considered groundwater) dominated streamflow with 64-80% of the total stream discharge contributed by pre-event water. High water table conditions within the catchment resulted in greater stream discharge and a greater contribution of event water in the streamflow than that observed under low water table conditions for similar storm events. The results also showed that differences in riparian zone width, vegetation, and surface saturation conditions between the upper and lower catchment can influence the relative magnitude of streamflow response from the two catchment areas.

### 3.1.4 Regional Groundwater Flow and Storage

Measurements of hydraulic head at and below the water table were taken on a monthly basis throughout 1995 and 1996. Contours of the water table elevation for the summers and winters of 1995 and 1996 are given in Figs. 23 and 24. In general, the water table is 0.5 to 2.0 m below the ground surface of the study field. It is deepest in the upper part of the field and during the summer season. Shallow groundwater flow is generally parallel to the direction of the land-surface slope (Fig. 4). North-south trending cross-sections (see Fig. 8 for names of monitoring wells) of equipotential contours are given in Figs. 25 and 26. In general, shallow and deep groundwater flow is parallel to the direction of the slope. Equipotentials in the lower part of the field and near the creek indicate a significant component of upward groundwater flow (see also Table 3). This is consistent with artesian conditions in these areas. Indeed, groundwater was almost always flowing onto the land surface from a spring about 5 m north of monitoring well SP 14 (Fig. 6).

Slug tests were carried out on many monitoring wells to estimate the saturated hydraulic conductivity of geologic materials below the water table. Results are given in Fig. 17. Estimates of the average hydraulic conductivity combined with information from Figs. 25 and 26 can be used in a steady-state flow net analysis to estimate the amount of groundwater inflow and outflow below the field site.

Fig. 23. Contours of water table elevation (m a.s.l.) for summer and winter of 1995.

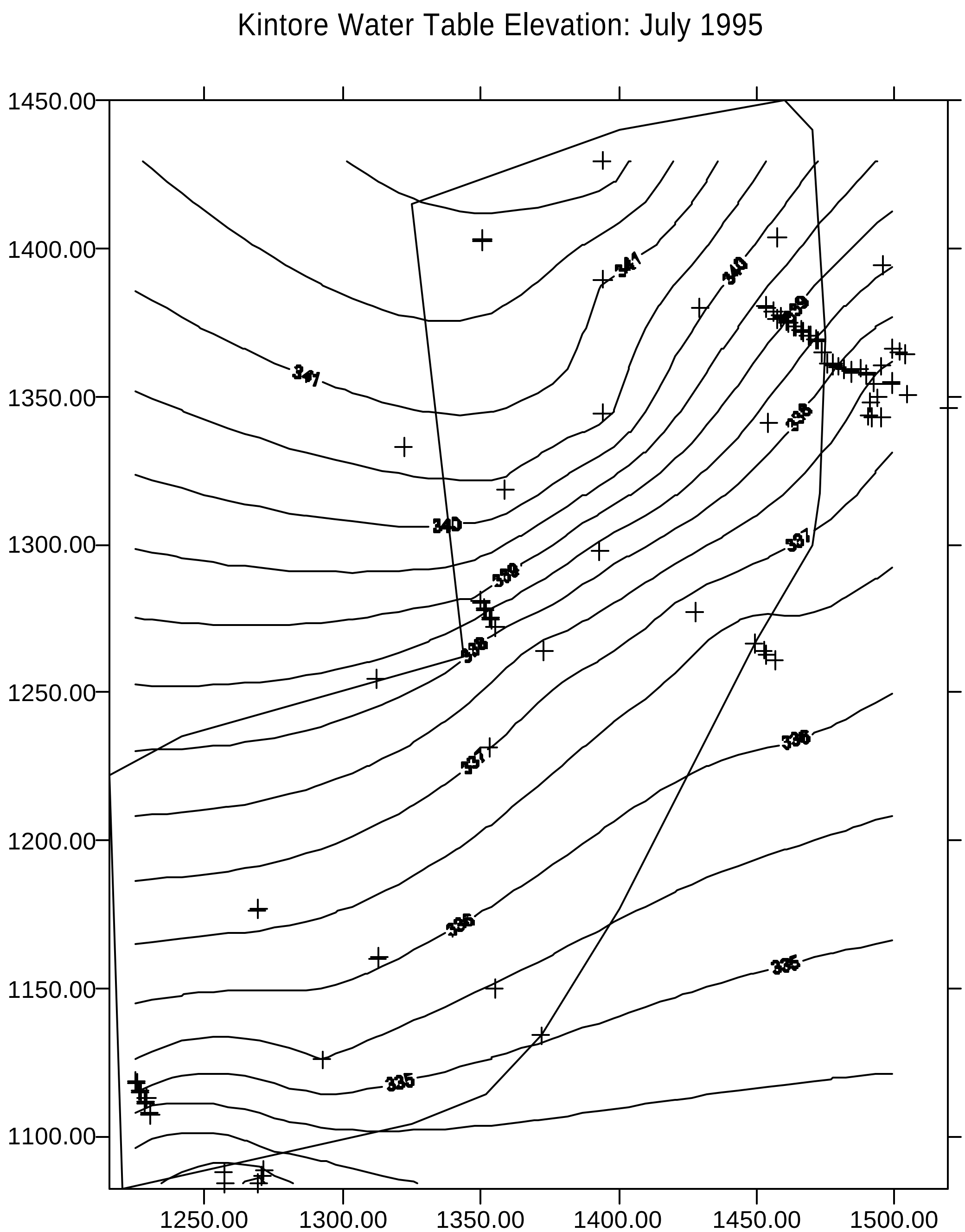


Fig. 23. (Continued)

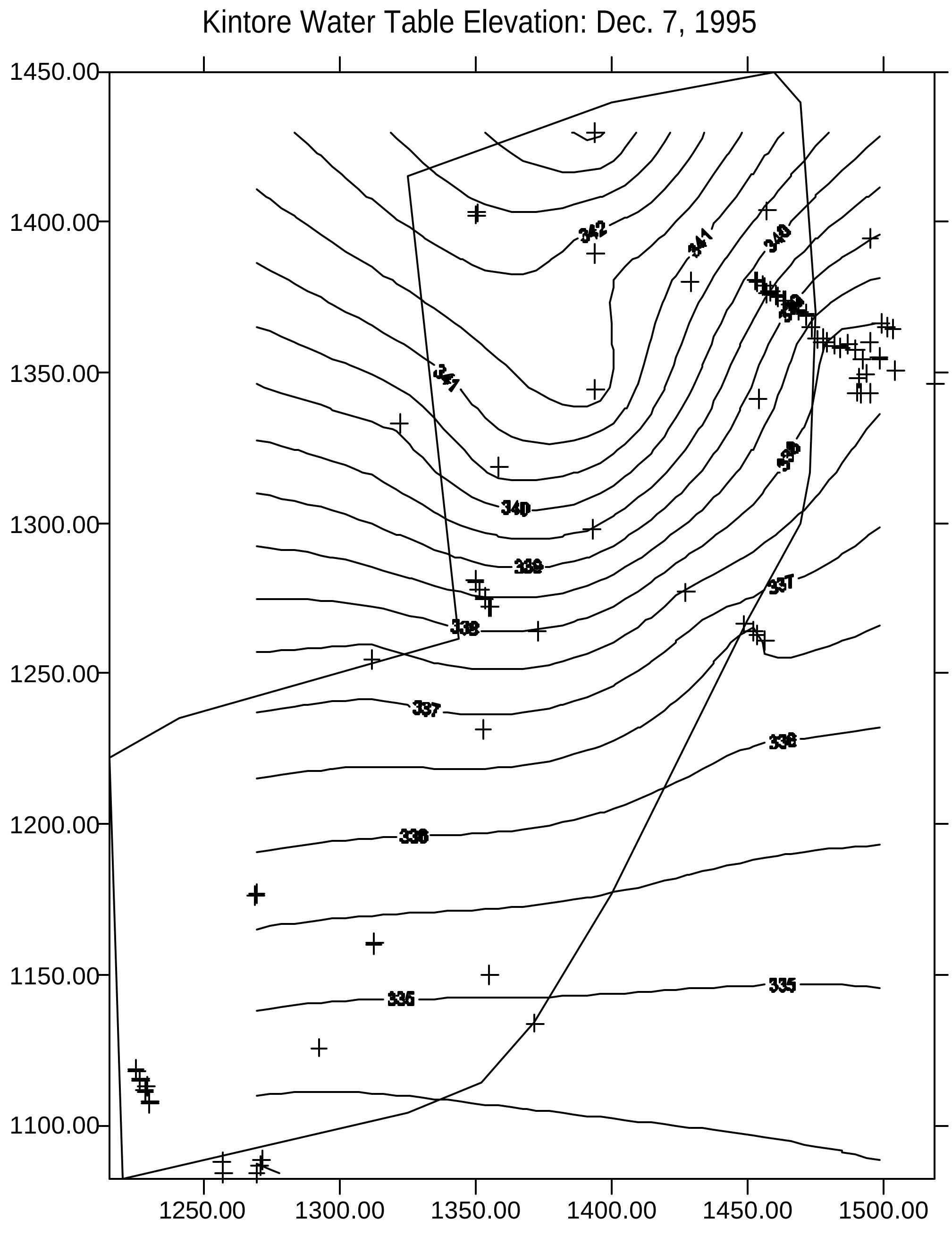


Fig. 24. Water table contours (m a.s.l.) for summer and winter of 1996.

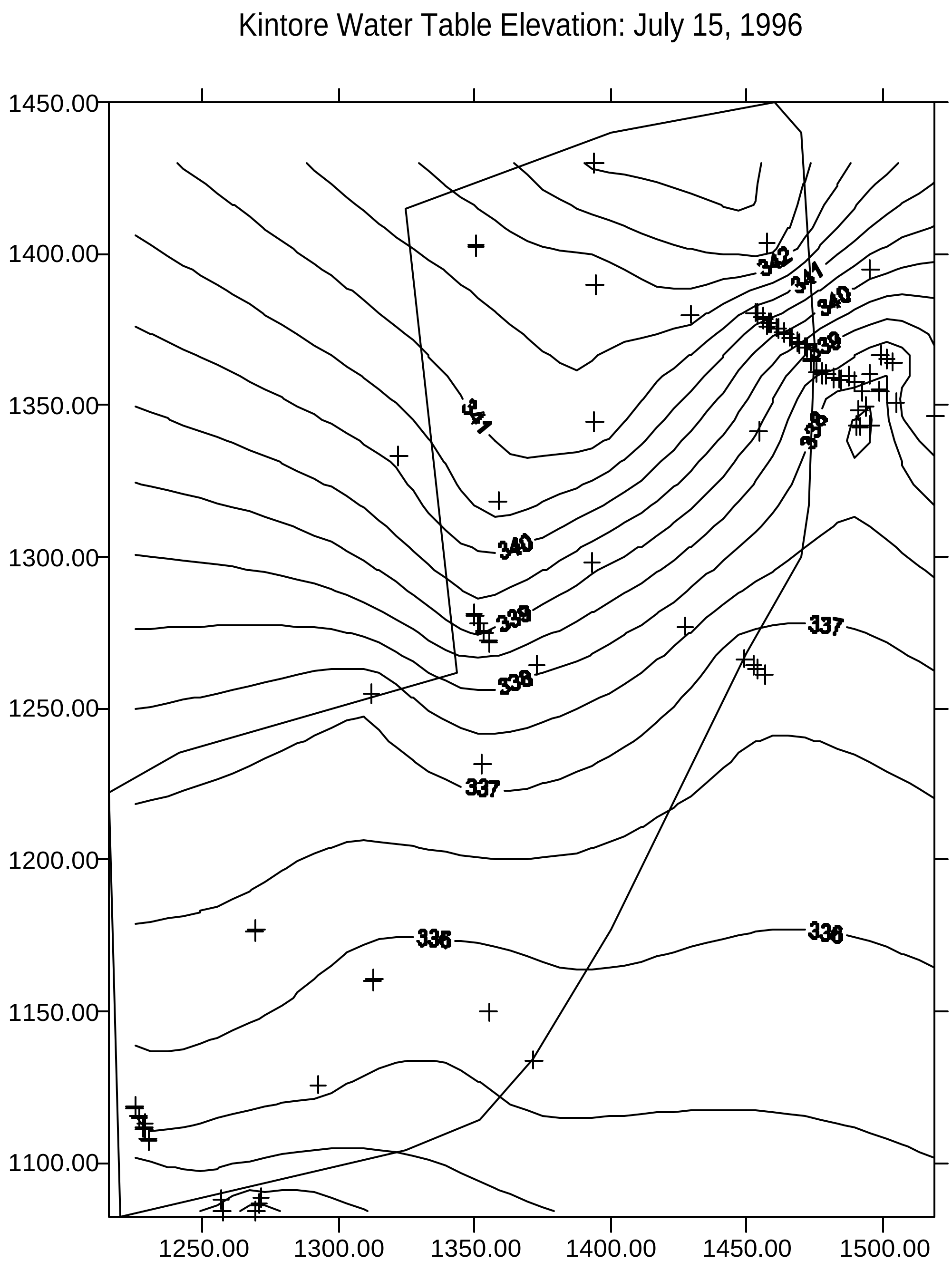


Fig. 24. (Continued)

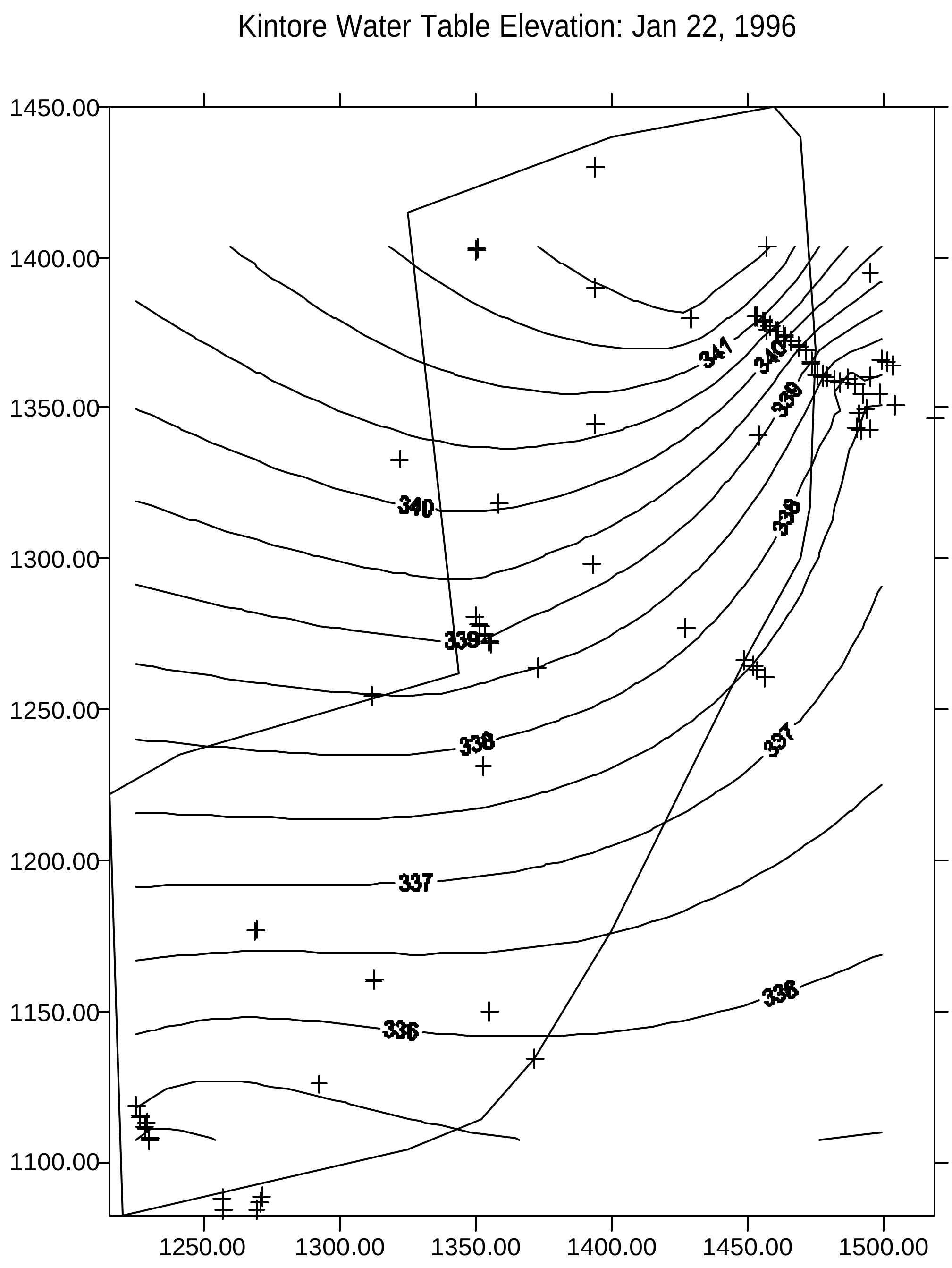


Fig. 25. Contours of equipotentials (m a.s.l.) during the summer and winter of 1995 for the main north-south cross-section shown in Fig. 8. Note that VE is the vertical exaggeration.

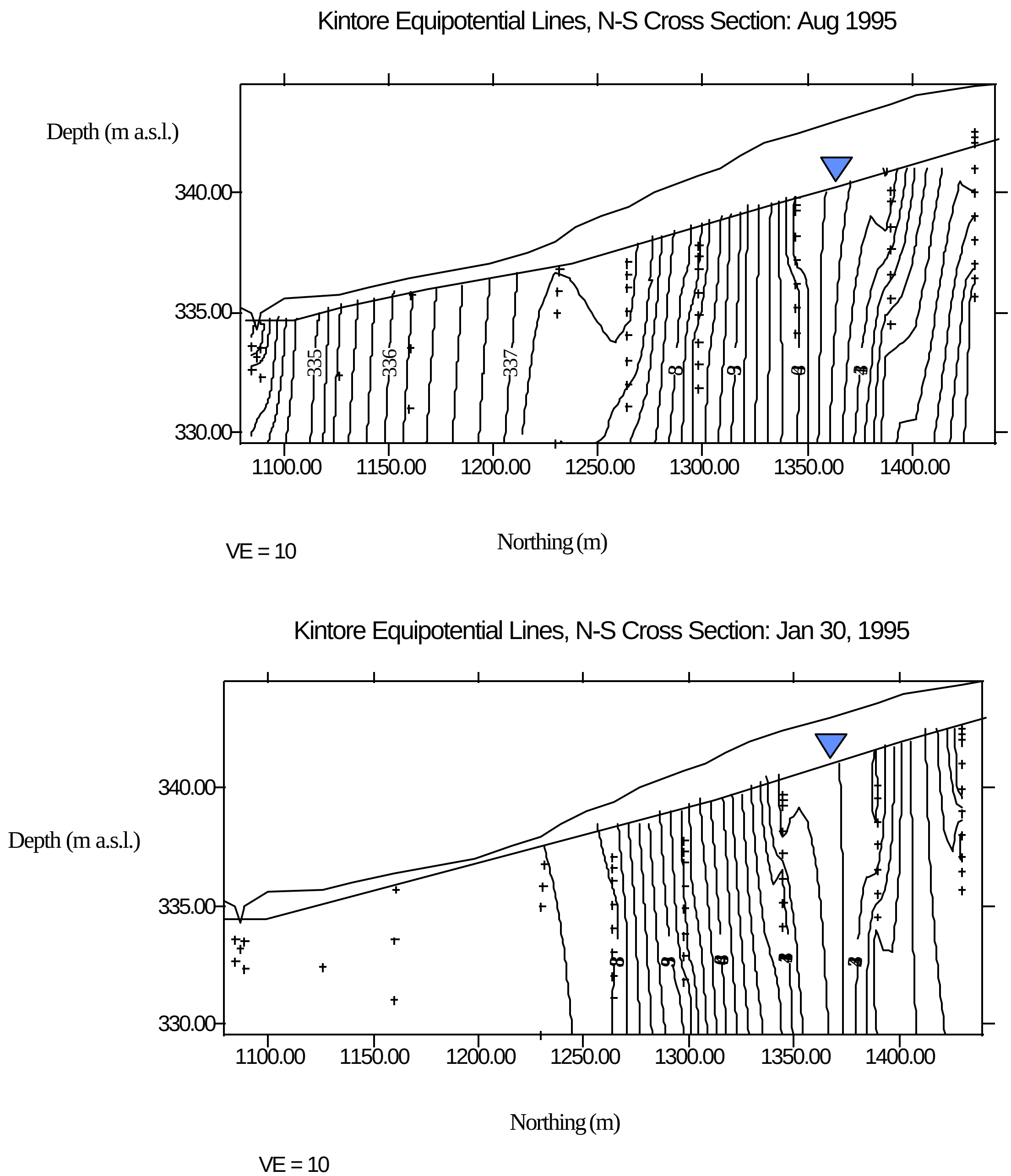
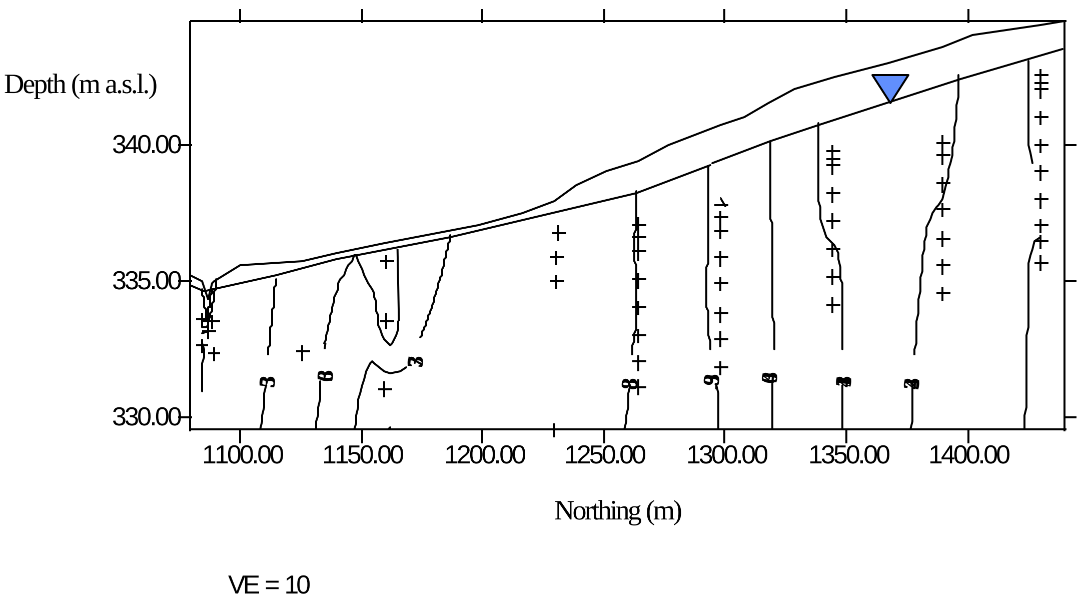


Fig. 26. Contours of equipotentials (m a.s.l.) during the summer and winter of 1996 for the main north-south cross-section shown in Fig. 8.

Kintore Equipotential Lines, N-S Cross Section: July15, 1996




Kintore Equipotential Lines, N-S Cross Section: Jan 18, 1996

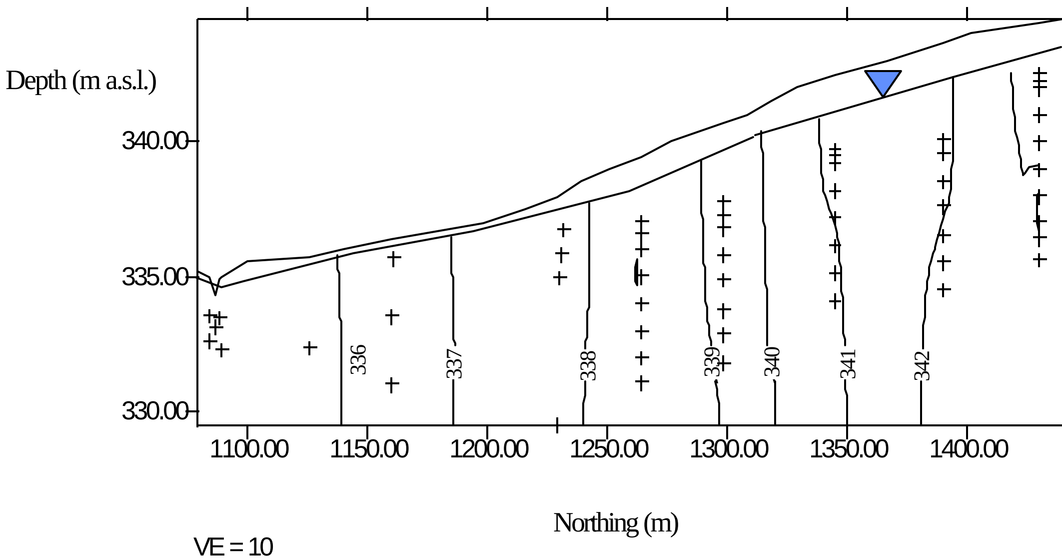
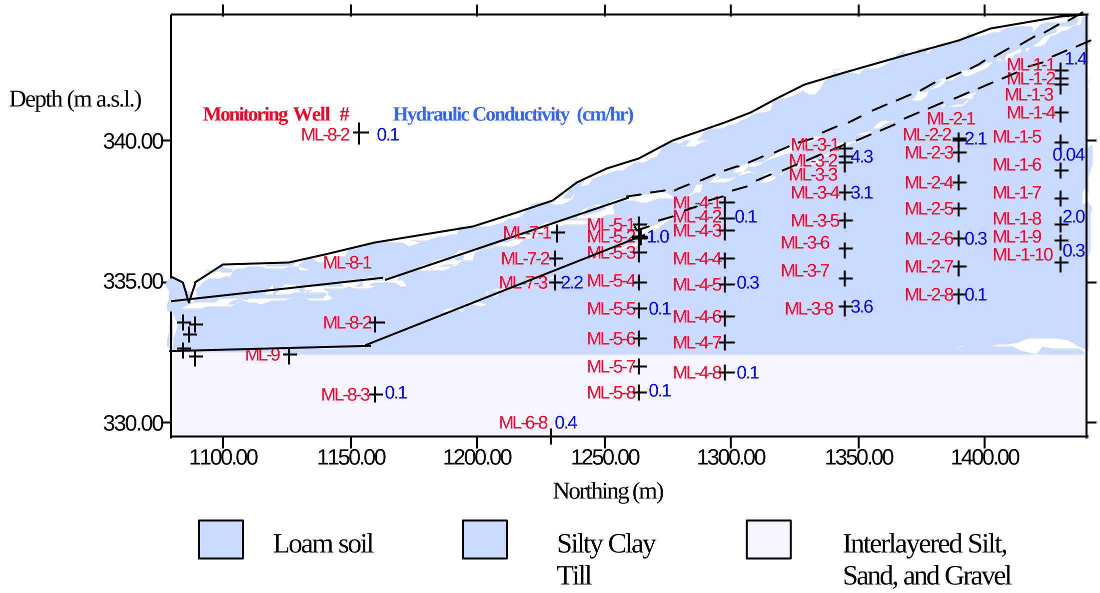


Fig. 27. Hydraulic conductivity measurements from slug test on wells along the main north-south cross section.



To estimate the amount of groundwater inflow ($G_I$ in equation 1) and groundwater outflow ($G_O$ in equation 1) below the study field, we use Darcy's Law and hydraulic gradients from Figs. 25 and 26. $G_I$ or $G_O$ (the volume of water per unit width of aquifer) are given by

$$K\ L\ \frac{i_s \% i_w}{2}$$

where K is the hydraulic conductivity (we use average values based on Fig. 27), L is the thickness of the aquifer at the north and south boundaries of Fig. 27 for calculation of $G_I$ and $G_O$, respectively, and $i_s$ and $i_w$ are the hydraulic gradients measured during the summer and winter. Note that parameter values are different for $G_I$ and $G_O$ and are based on information in Figs. 25, 26, and 27. Calculations of $G_I$ and $G_O$ are summarized in Table 4.

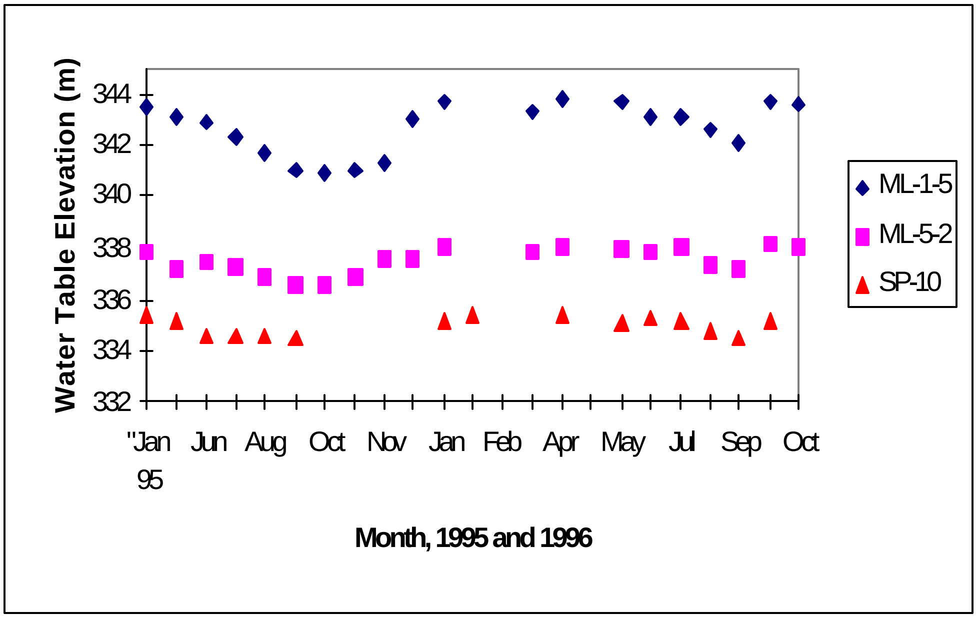
**Table 4.** Calculation of groundwater inflow and outflow for 1995 and 1996.

| Year | K (m/year) | L(m) | $i_s$ | $i_w$ | $G_I$ ($m^2$/year) |
|---|---|---|---|---|---|
| 1995 | 82 | 13.3 | 0.026 | 0.03 | 30.5 |
| 1996 | 82 | 13.3 | 0.025 | 0.025 | 27.3 |

| Year | K (m/year) | L(m) | $i_s$ | $i_w$ | $G_O$ ($m^2$/year) |
|---|---|---|---|---|---|
| 1995 | 79 | 5 | 0.028 | 0.032 | 11.9 |
| 1996 | 79 | 5 | 0.042 | 0.02 | 12.2 |

The change in the amounts of surface water and groundwater storage during 1995 and 1996 are the only two terms in the water balance equation (equation 1 on page 1) yet to be determined. Since there are no standing bodies of water within the study area, we assume that the change in surface water storage ()$S_s$ of equation 1) on an annual basis is trivial. However, we need to determine if there is a change in groundwater storage ()$S_G$ of equation 1) over the course of the investigation. We estimate )$S_G$ for each year by determining the water volume difference between water table elevations at the beginning and end of each year. Average values of specific yield needed in the calculation of )$S_G$ , were obtained from soil core measurements (Appendix 1). We have selected three monitoring wells at different slope positions for the calculation of )$S_G$. Fig. 28 shows the elevation of the water table throughout 1995 and 1996 for monitoring wells ML -1 - 5, ML - 5 - 5, and SP 10 (see Figs. 6 and 8 for well locations). In Table 5 we present a summary of calculations of )$S_G$ for 1995 and 1996. So, it is apparent that the upper area gains about an average of 23 mm and the lower area loses about 6 mm from storage during 1995. Likewise, during 1996 the upper area gained 8 mm and the lower area lost 21 mm from groundwater storage.

**Fig. 28.** Elevation of the water table measured at three monitoring wells during 1995 and 1996.



| 1995 | Jan 95 | Jan 96 | )h** | Specific Yield | )$S_G$*** |
|---|---|---|---|---|---|
| ML-1-5 | 343.556* | 343.676 | 0.12 | 0.16 | 0.019 |
| ML-5-2 | 337.9 | 338.07 | 0.17 | 0.16 | 0.027 |
| SP 10 | 335.388 | 335.348 | -0.04 | 0.14 | -0.006 |

*** elevation of the water table in m a.s.l.**

**** overall change in water table elevation during 1995 in m.**

***** change in groundwater storage in m.**

| 1996 | Jan 96 | Oct 96 | )h** | Specific Yield | )$S_G$*** |
|---|---|---|---|---|---|
| ML-1-5 | 343.676 | 343.686 | 0.01 | 0.16 | 0.0016 |
| ML-5-2 | 338.07 | 338.16 | 0.09 | 0.16 | 0.014 |
| SP 10 | 335.348 | 335.198 | -0.15 | 0.14 | -0.021 |

*** elevation of the water table in m a.s.l.**

**** overall change in water table elevation during 1996 in m.**

***** change in groundwater storage in m.**

**Table 5.** Calculation of the change in groundwater storage at three positions in the study field during 1995 and 1996.

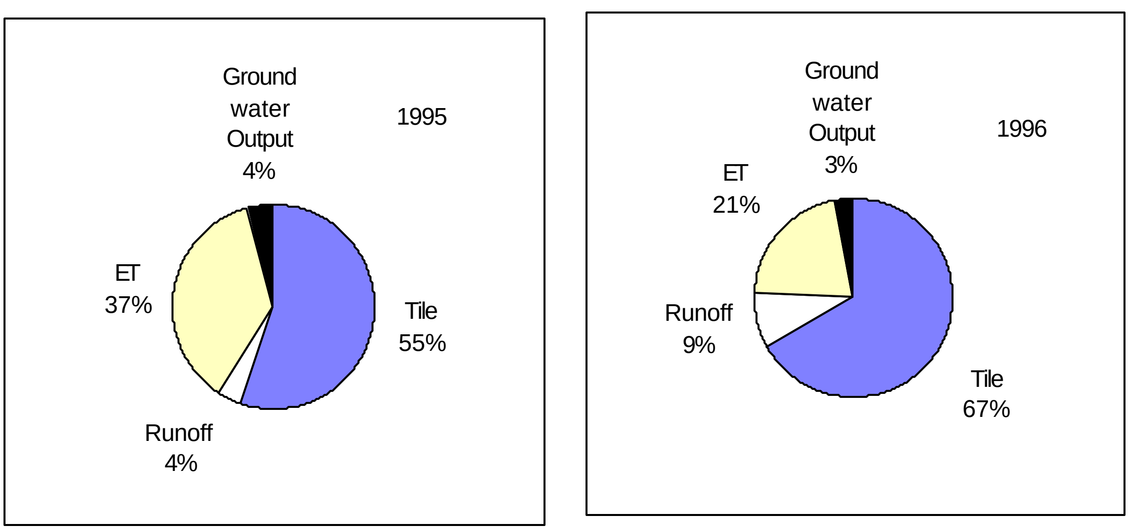
## 4.0 Summary of Water Balance

A summary of the water balance for 1995 and 1996 is given in Table 6. Each of the output components are given in Fig 29 as a percentage of the total inputs.

| | P | $G_I$ | $G_O$ | ET | $Q_I$ | $Q_o$ | )$S_S$ | )$S_G$ | $T_o$ | Dif |
|---|---|---|---|---|---|---|---|---|---|---|
| 1995 | 1030 | 102 | 40 | 403 | 0 | 46 | 0 | 14 | 598 | 59 |
| 1996 | 1135 | 91 | 41 | 257 | 0 | 105 | 0 | -5 | 801 | 17 |

**Table 6.** Summary of the water balance for Kintore site, 1995 and 1996. Units are mm.

**Fig. 29.** Percentage that each output component comprises of the total water input during 1995 and 1996.



**Part 2 Nitrogen and Chloride Mass Balances**

**1.0 Fertilizer and Precipitation Inputs**

Fertilizer inputs were mainly in the form of liquid swine manure applied twice in 1995 and once in 1996. As well, a solution of 28% nitrogen was applied in the spring of 1996. The quantities of chloride and nitrogen applied with fertilizer and also in precipitation are summarized in Tables 7 and 8.

| Fertilizer | Date (day #) | Kg N /Ha | Kg Cl /Ha |
|---|---|---|---|
| Spring 1995 | Apr. 17 (107) | 3.3 (manure) | 8.3 (manure) |
| Fall 1995 | Sept. 18 (261) | 2.9 (manure) | 7.3 (manure) |
| Spring 1996 | May 8 (129) | 24.5 (manure)<br>101 (28% N) | 14.5 (manure) |

**Table 7.** Inputs of nitrogen and chloride by fertilizer applications.

| Precipitation | Kg N/Ha | Kg Cl/Ha |
|---|---|---|
| 1995 | 15 | 15 |
| 1996 | 13 | 9 |

**Table 8.** Inputs of nitrogen and chloride through precipitation.

**2.0 Soil Storage and Harvest Outputs**

The results of N and Cl soil extraction analyses are given in Appendix 8. A detailed sampling of soil between 0 - 60 cm depth on a 10 m grid in August, 1994 revealed no obvious trends in the distribution of N or Cl. Therefore, following this initial intensive sampling, samples were taken during 1995 and 1996 at fewer locations (Fig. 6). The main objectives of soil sampling were to estimate changes in the amount of nitrogen and chloride storage of near-surface soils on an annual basis. Soil N and Cl results for 1995 and 1996 are summarized in Table 9. Complete results are given in Appendix 9.

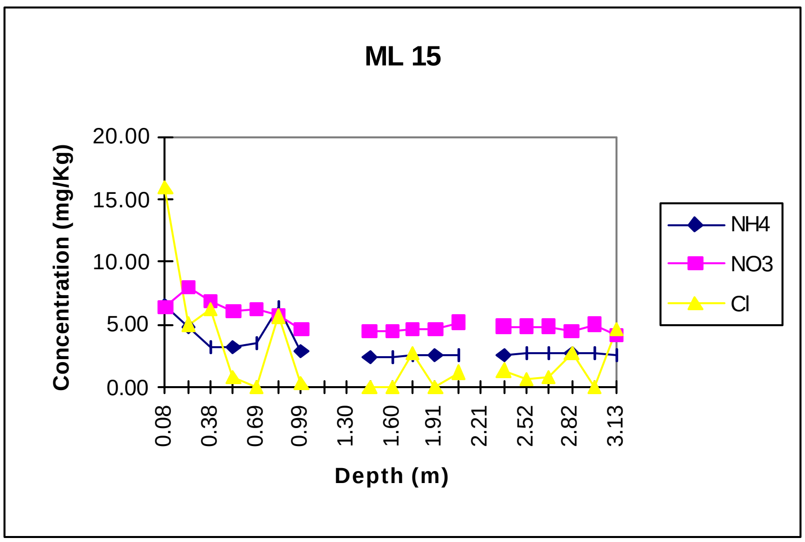
| Sampling Time | Kg $NO_3$/Ha | Kg $NH_4$/Ha | Kg Cl/Ha |
|---|---|---|---|
| Apr. 1995 | 7 | 9 | 56 |
| Nov. 1995 | 27 | 13 | 62 |
| Apr.1996 | 21 | 10 | 41 |
| Dec. 1996 | 22 | 6 | 31 |

Table 9. Average amounts of nitrogen and chloride for 1995 and 1996 in soils.

A number of deep soil cores were taken during the winter of 1996 in conjunction with monitoring well installation on the field. The cores were divided into 15 cm increments and extractions for N and Cl were performed. N and Cl versus depth are given in Fig. 30 for monitoring wells ML 20 and 15, located in the lower and upper parts of the study field, respectively (Fig. 6). Results for other cores are given in Appendix 8.

During 1995, three cuts of alfalfa removed 79, 71, and 68 Kg N /Ha. The increase in soil nitrogen levels between Apr. 1995 and Nov. 1995, shown in Table 9, is due to an addition of manure in mid September. The corn crop grown in 1996 was harvested by a combine with a GPS and yield monitor system. Corn yield over the field

**Fig. 30.** Concentration (mg/Kg) of nitrate, ammonium, and chloride in soil samples measured in Feb. 1996 at monitoring wells ML 15 and 20.



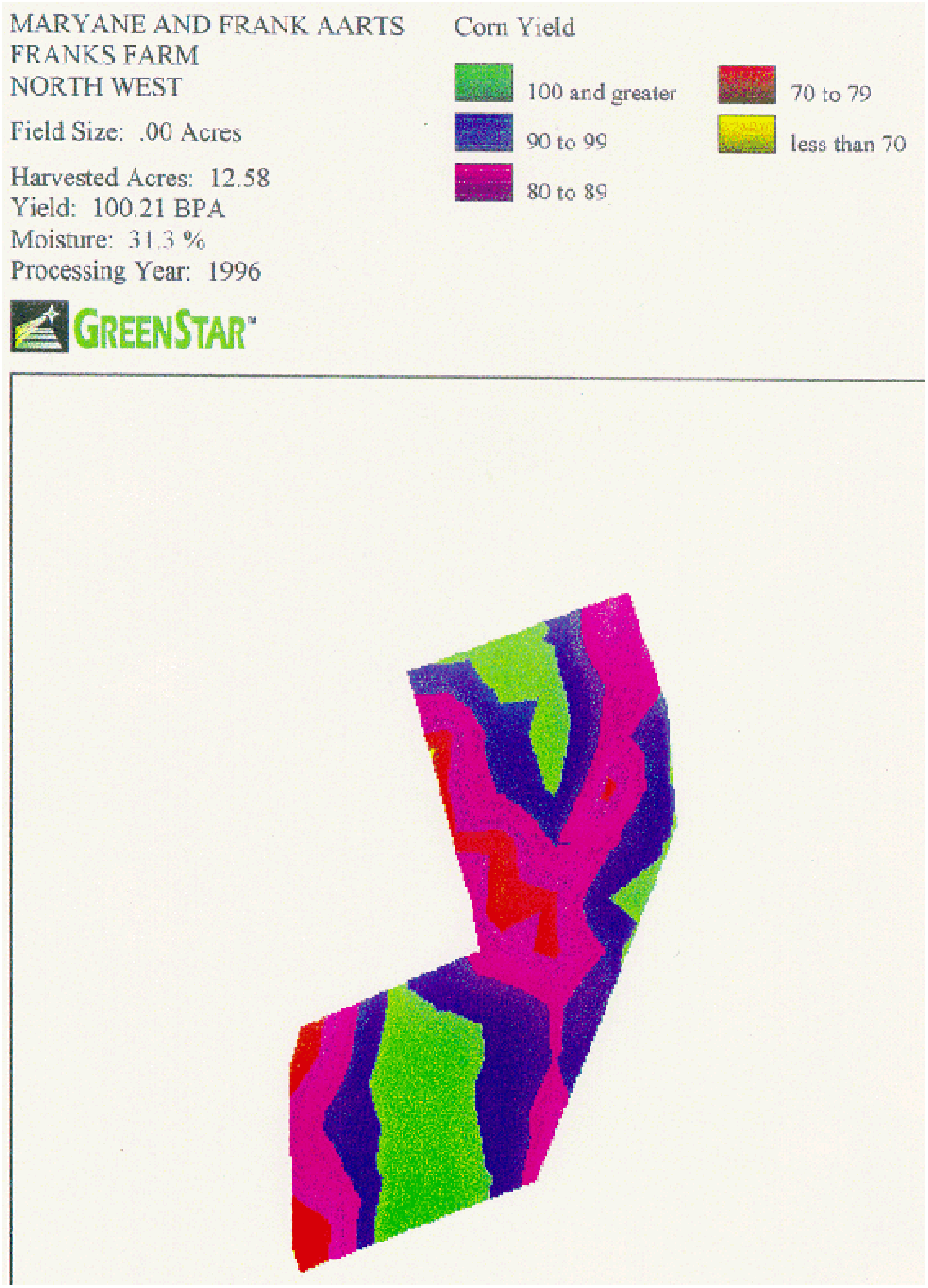



Figure 31. Contours of corn yield for Kintore, 1996

area is shown in Fig. 31. The total harvested area was 5.1 Ha. The total dry corn yield (31.3% moisture) was 27085 Kg. Laboratory analysis found 0.66% of N in the kernels. Therefore, the average output of N was 35 Kg N/Ha.

**3.0 Surface Runoff and Tile Drain Outputs**

Throughout 1995 and 1996 there was an increase in nitrate concentration in the downstream direction of the Logan drain based on water samples collected weekly by UTRCA staff at stations C1, C3, C5, C10, and C12 (Fig. 32). However, there were essentially no trends in ammonium and chloride concentration in the downstream direction (Fig. 32). The concentration of ammonium is much less than nitrate along the entire length of the ditch.

Tile drains and the secondary ditch contributed a significant amount of nitrogen and chloride to the Logan drain. Background concentrations of nitrate, chloride, and ammonium based on weekly grab samples taken throughout 1995 and 1996 from each tile outlet that only drains the study field are given in Fig. 33. Similarly, concentrations of the same species from the tiles that drain other fields and from the secondary ditch are shown in Fig. 34. All results are listed in Appendix 10.

The concentrations of nitrate, ammonium, and chloride in samples taken by the Isco automatic water samplers during major snowmelt or rainfall events are given in Figs. 35 (samples from Logan drain) and 36 (samples from tile outlets which drain the study field). Included on Fig. 36 are the manure application dates. Results from all Iscos are listed in Appendix 11.

**Fig. 32.** Nitrate, ammonium, and chloride concentrations at Logan drain stations C1, C3, C5,C10, and C12.

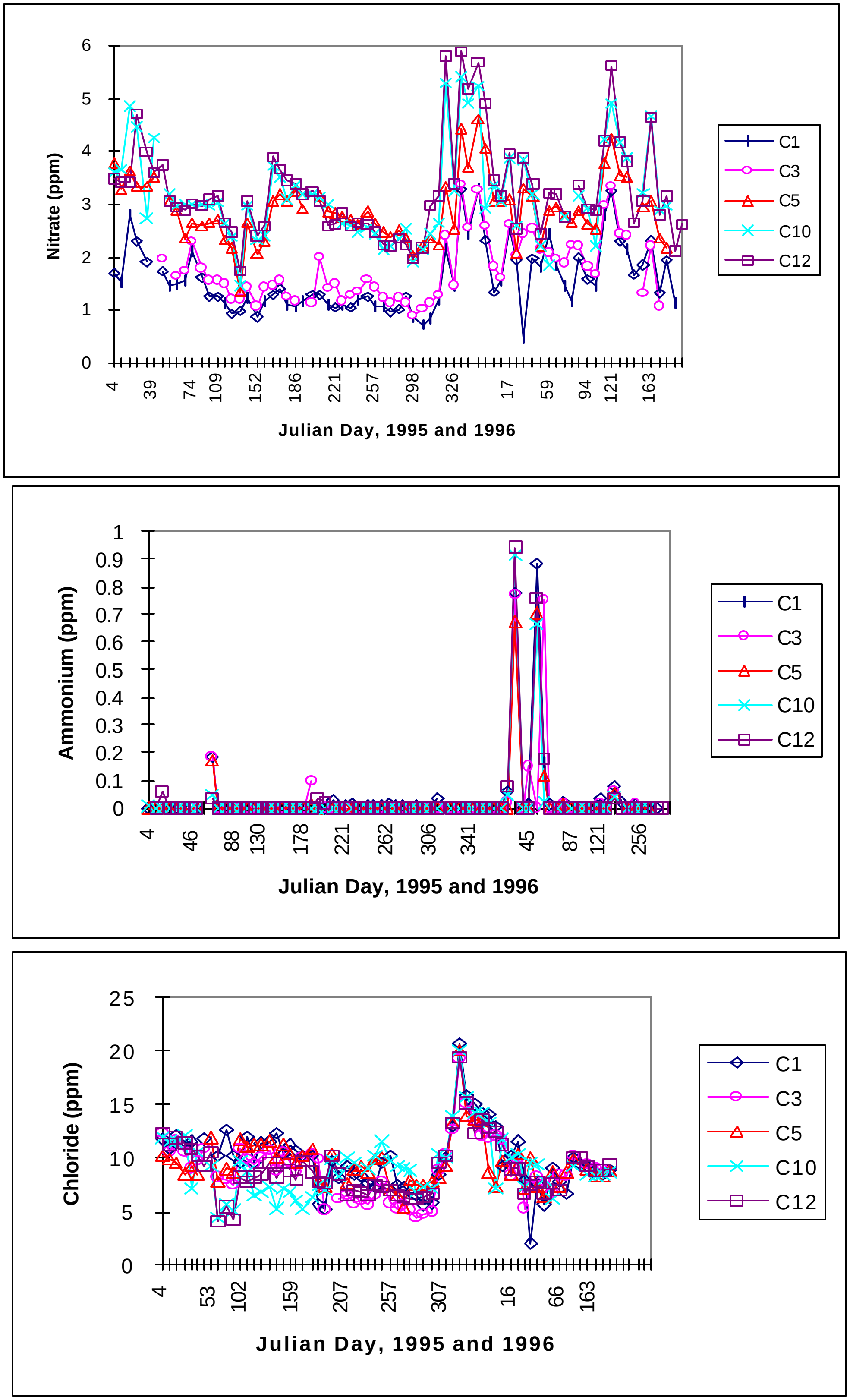


Fig. 33. Variation in the concentration of nitrate, ammonium, and chloride in tile outlets which drain the study field.

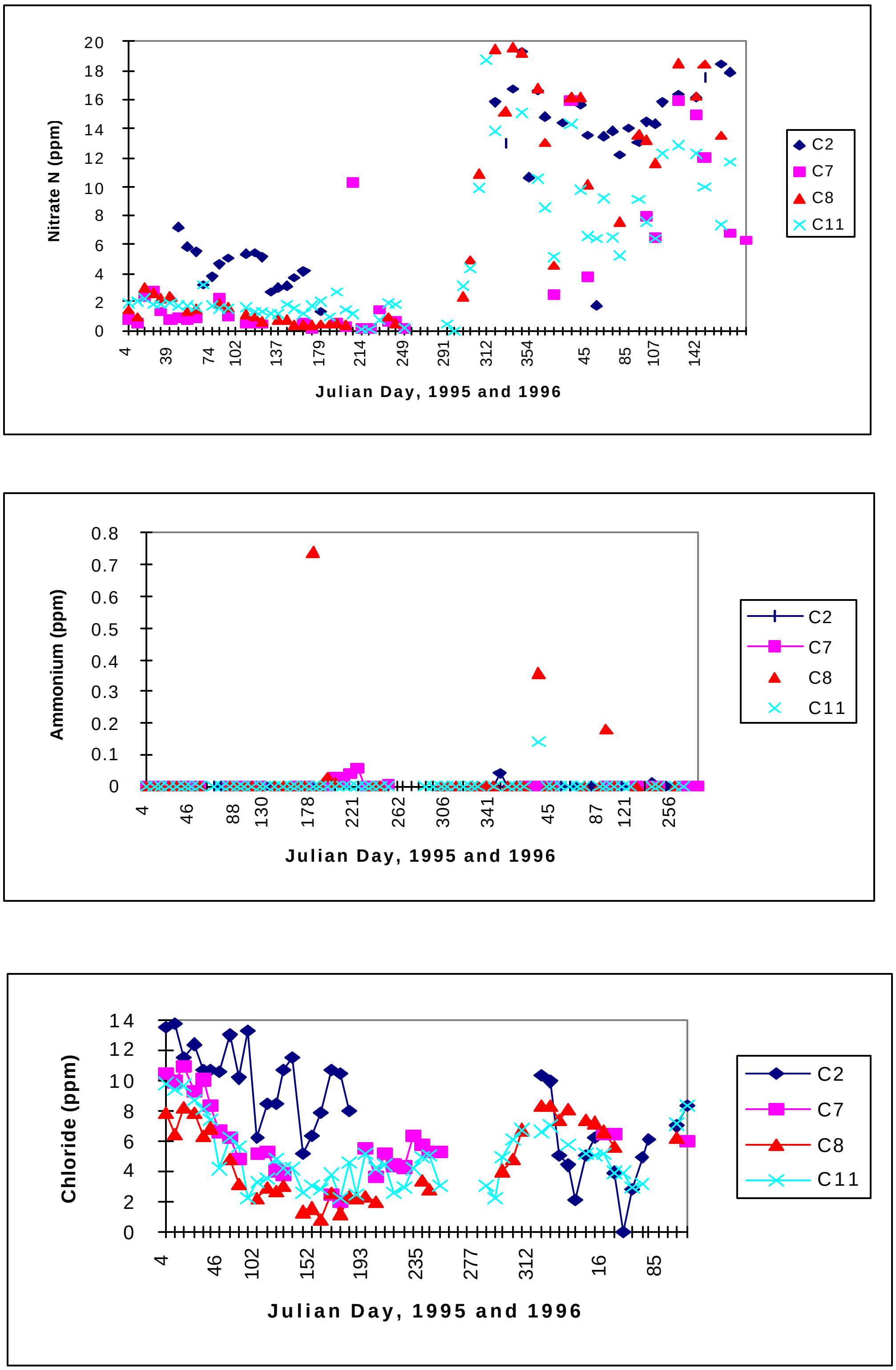


Fig. 34. Variation in concentration of nitrate, ammonium, and chloride from tile outlets which do not drain the study field and the secondary ditch.

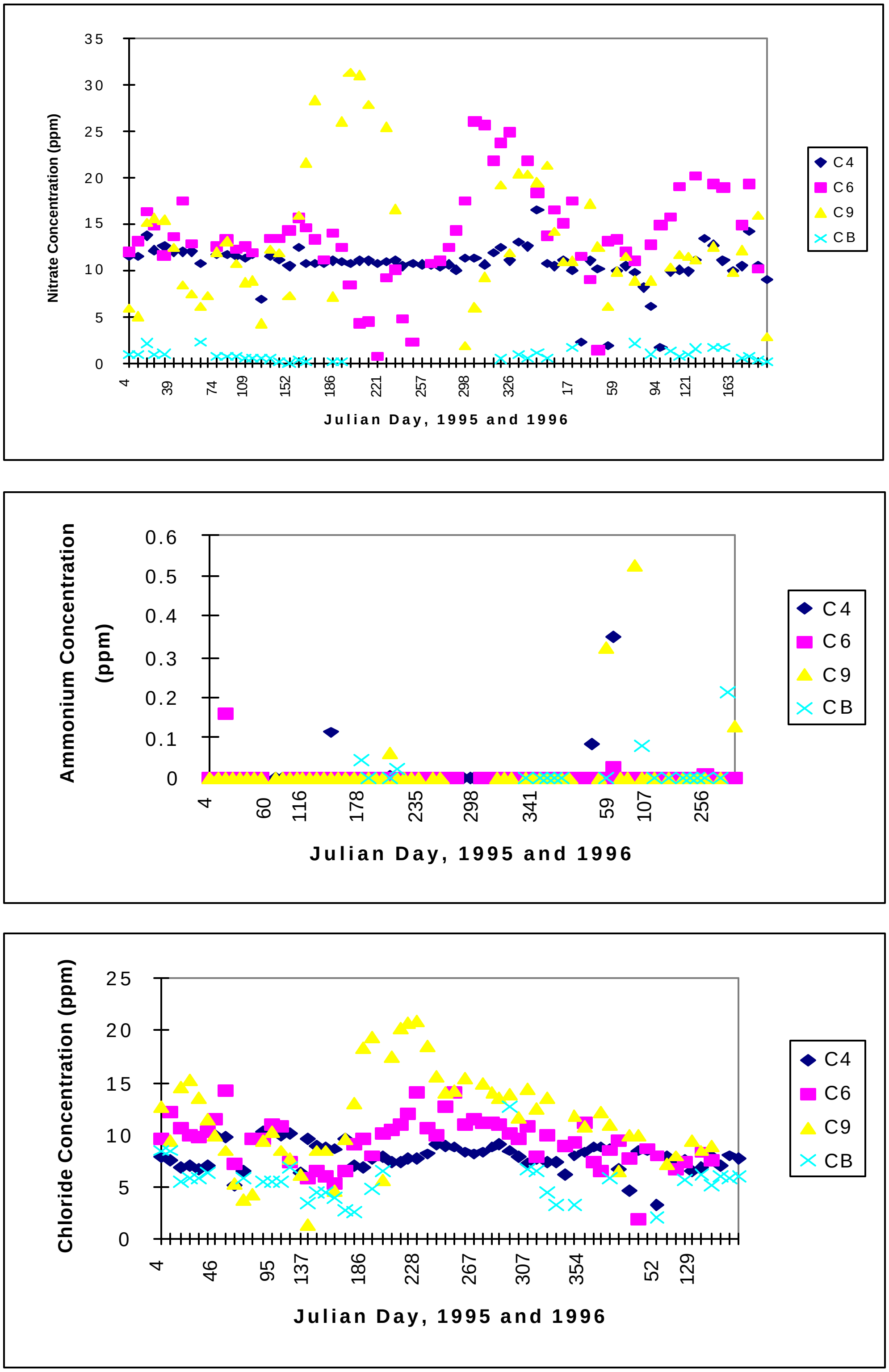


Fig. 35. Concentrations of nitrate, ammonium, and chloride in samples taken by Iscos at C1 and C12.

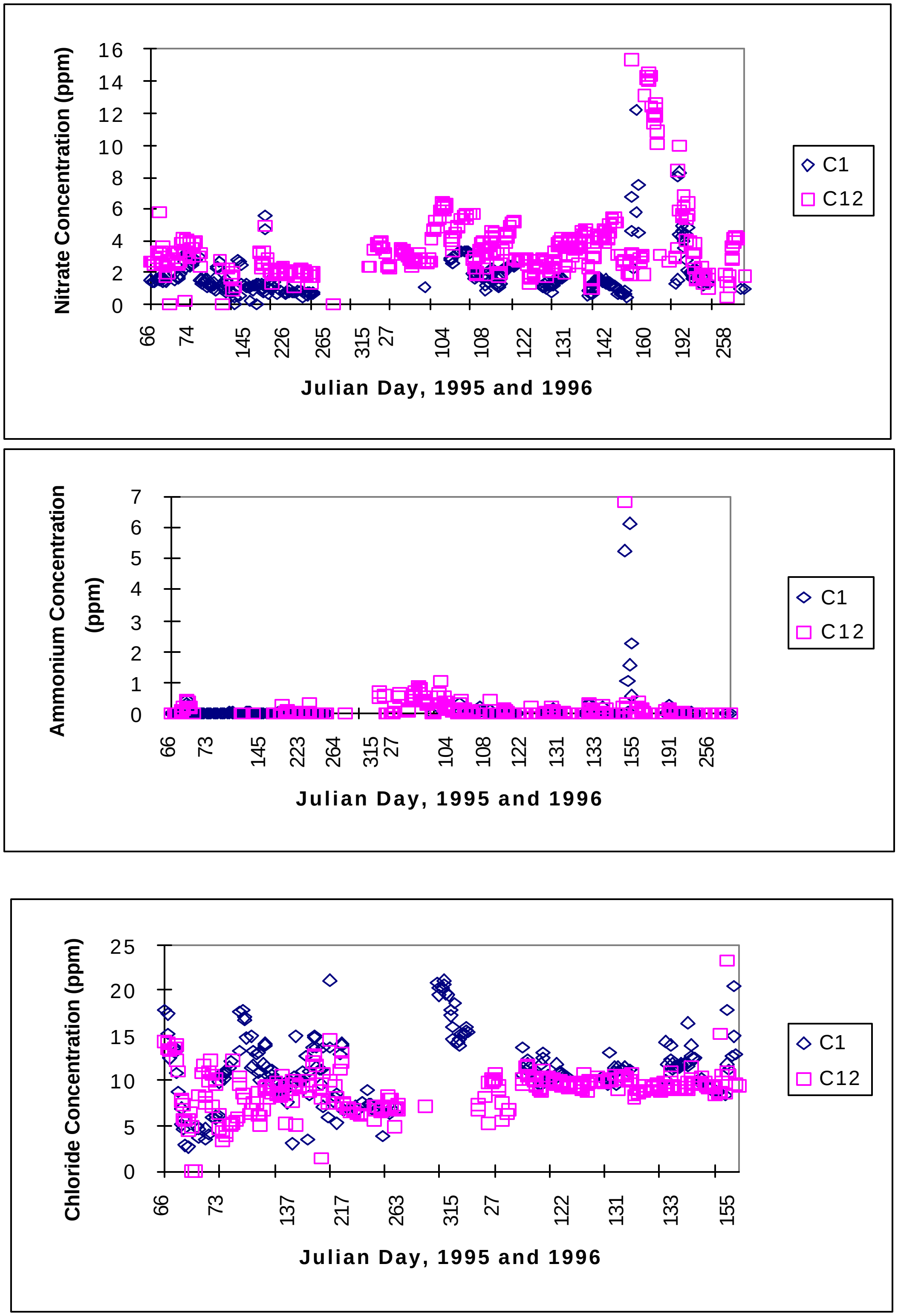


Fig. 36. Concentrations of nitrate, ammonium, and chloride in samples taken by Iscos at tile outlets which drain the study field. Arrows indicate manure application dates.

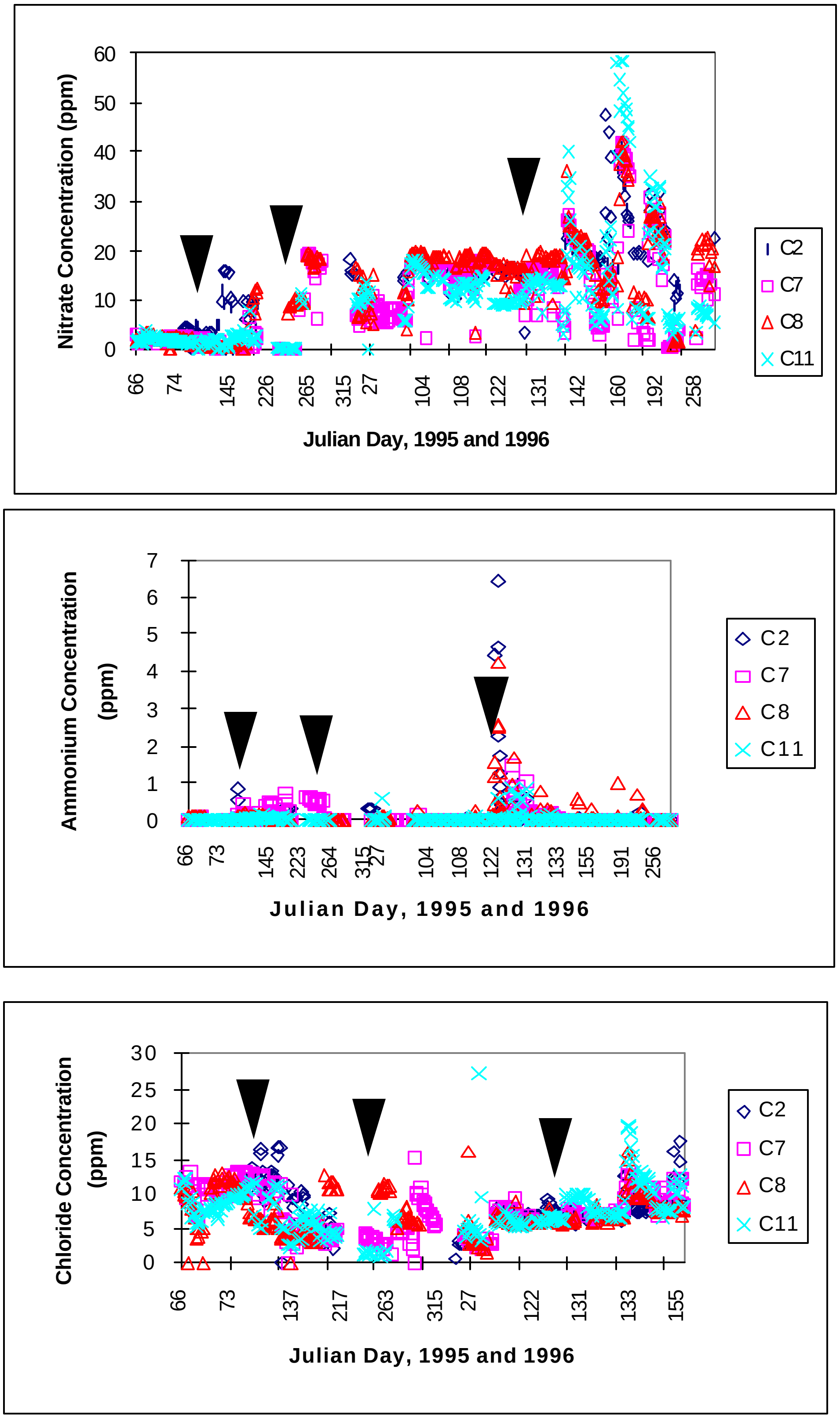
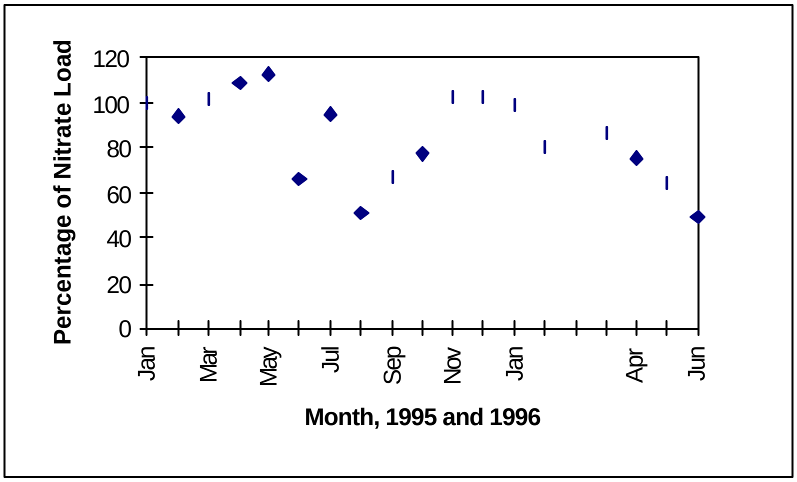
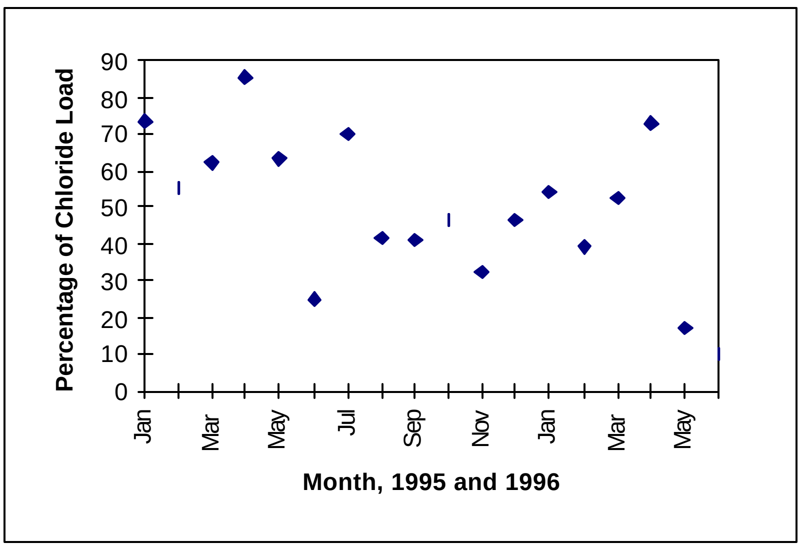


Fig. 37. Percentage of increase in nitrate and chloride load between ditch stations C1 and C12 from all tile outlets and the secondary ditch during 1995 and 1996.



In Fig. 37, we summarize the contribution of all tile drains and the secondary ditch to the overall increase in nitrate and chloride loadings between ditch monitoring stations C1 and C12 during 1995 and 1996. Calculation of load was based on the product of the mean monthly concentration from the weekly grab and Isco samplers and the total monthly discharge. On average, the tile drains and the secondary ditch contribute about 84% of the increase in nitrate load and only 47% of the increase in chloride load. Table 10 gives the total output of nitrogen (based on nitrate only) and chloride from each tile drainage area on the study field.

| Nitrate | C2 | C7 | C8 | C11 |
|---|---|---|---|---|
| 1995 | 59 | 43 | 24 | 29 |
| 1996 | 63 | 30 | 62 | 55 |

Table 10a. Output of nitrate in Kg N/Ha from each tile drainage area. Note that 1996 data only includes up to the end of June.

| Chloride | C2 | C7 | C8 | C11 |
|---|---|---|---|---|
| 1995 | 50 | 52 | 24 | 30 |
| 1996 | 22 | 18 | 26 | 23 |

Table 10b. Output of chloride in Kg Cl/Ha from each tile drainage area. Note that 1996 data only includes up to the end of June.

Surface runoff from the study field into Logan drain measured at the breach in the berm near the southwest corner of the field was much less than the total volume of tile drainage effluent (Fig. 21). Similarly, the amount of nitrogen and chloride which entered the drain through the breach was lower than the contribution from tiles. The total amounts were 1.6

and 3.5 Kg N/Ha and  and 3.5 and 7.8  Kg Cl/Ha  for 1995 and 1996 (up to the end of June), respectively.

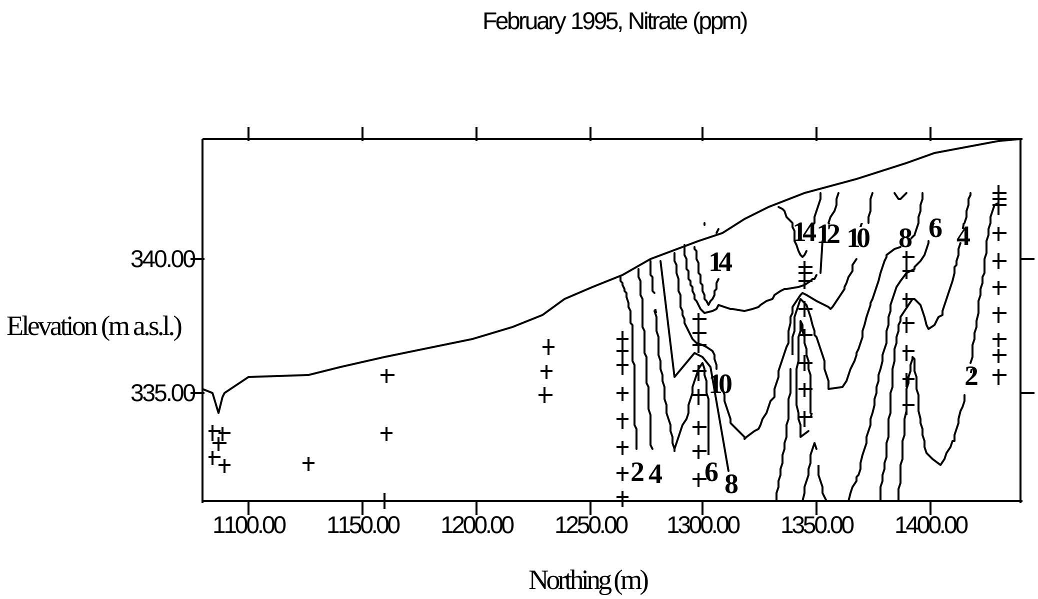
**4.0 Groundwater Input and Output of N and Cl**

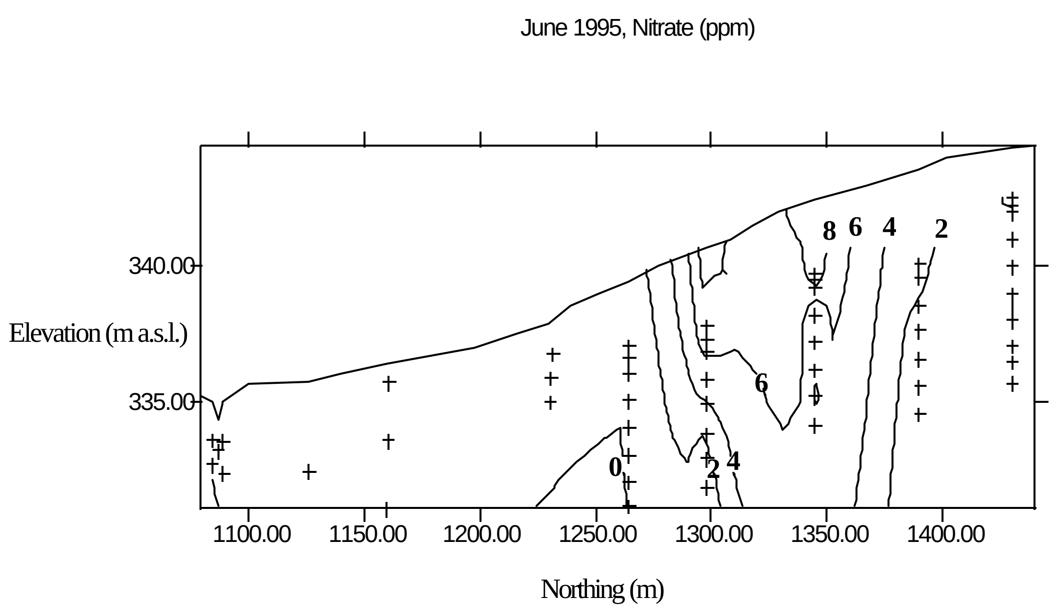
Nitrate was by far the dominant nitrogen species found in groundwater.  The variation in nitrate concentration with depth during 1995 and 1996 along the north-south trending cross-section shown in Fig. 8 is given in Fig. 38.  As well, the distribution of chloride at the same sampling times is given in Fig. 39.  All other results of nitrate, chloride, and ammonium analyses of groundwater samples are given in Appendix 12.

The highest concentrations of nitrate were found consistently in the upper slope position.  As well, in the upper slope area, nitrate concentration decreased with depth for many  sampling dates.  There appears to be slightly higher concentrations and more widespread occurrence of nitrate during the winter periods.  For the most part, the distribution pattern of chloride was very similar to nitrate.  A minor exception is that groundwater near the ditch in the lower part of the field still retains a small amount of chloride.

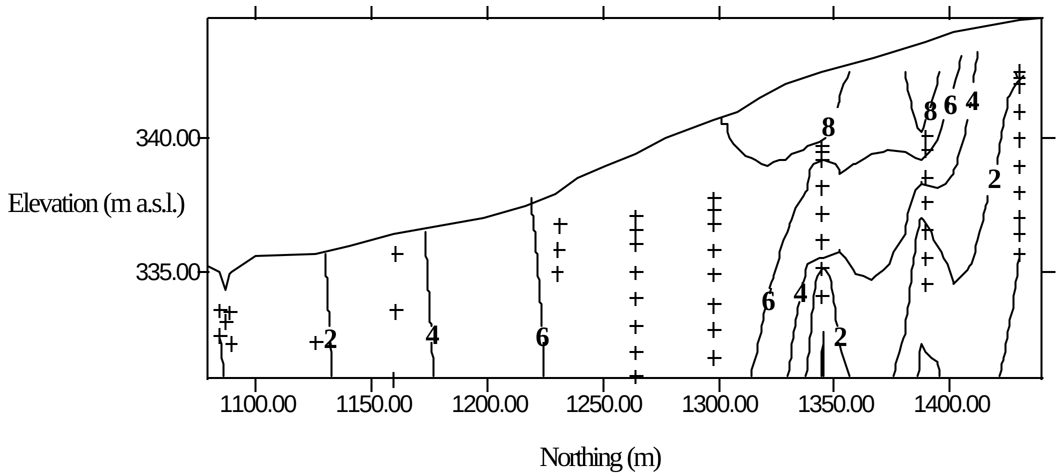
In order to understand the distribution and fate of nitrogen below the water table, we document the geochemical conditions that control nitrification-denitrification. Measurements of pH, Eh, and dissolved oxygen were performed in the field about once a month during 1995 and 1996.  The pH of groundwater was between  7.3 and 8.3 for all monitoring wells. There were no obvious

Fig. 38. Seasonal changes in nitrate concentration in groundwater along north - south cross section.



January 1996, Nitrate (ppm)




June 1996, Nitrate (ppm)

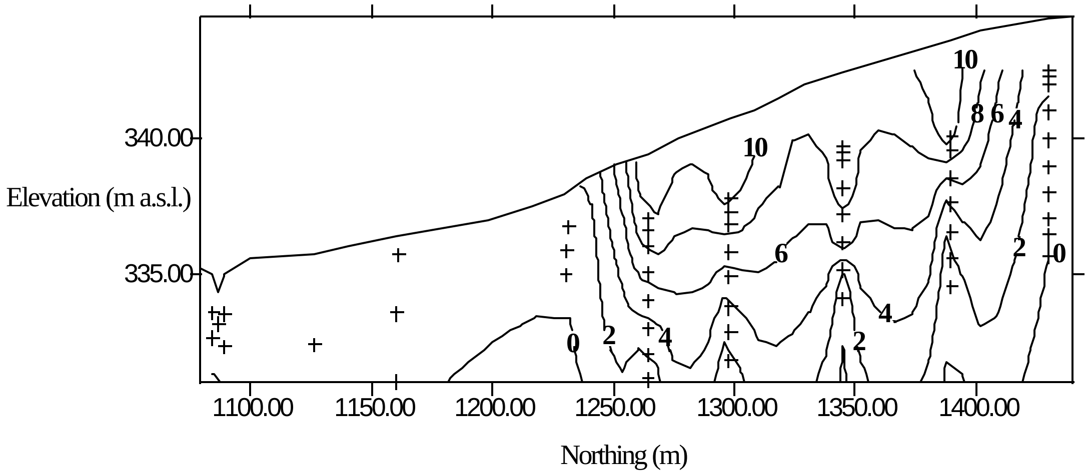
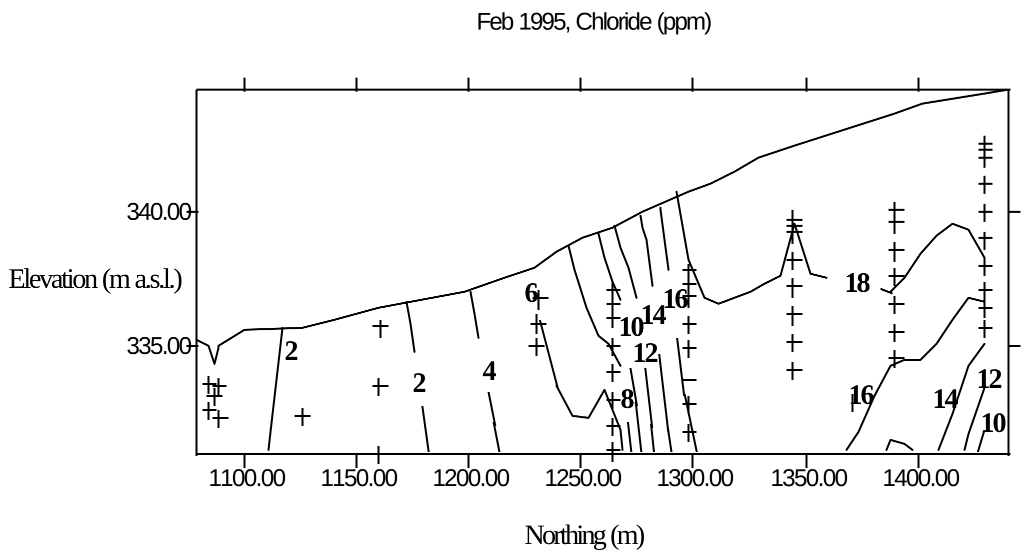
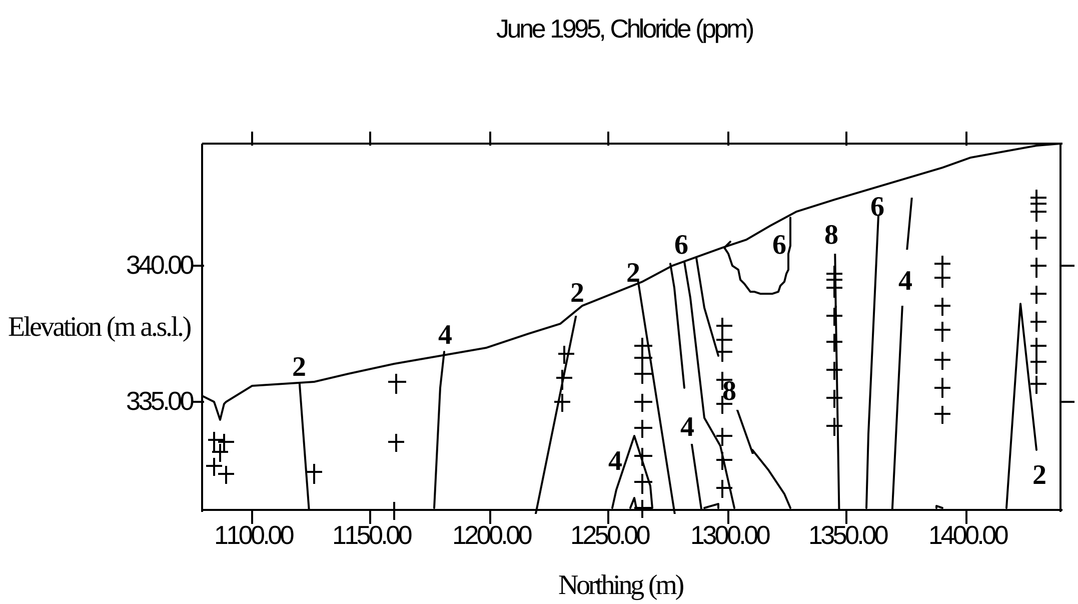
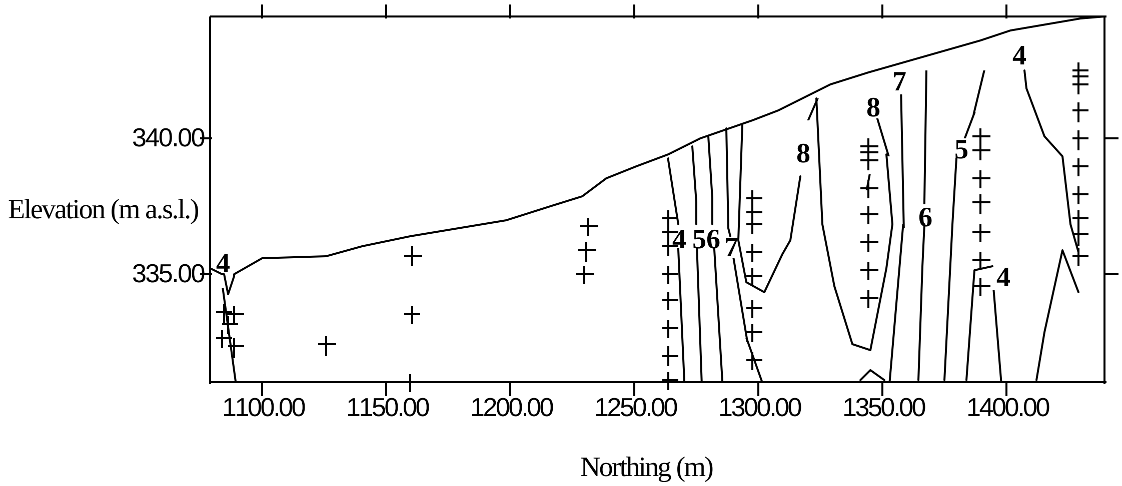


Fig. 39. Seasonal changes in chloride concentration in groundwater along north - south cross section.



Jan 1996, Chloride (ppm)




June 1996, Chloride (ppm)

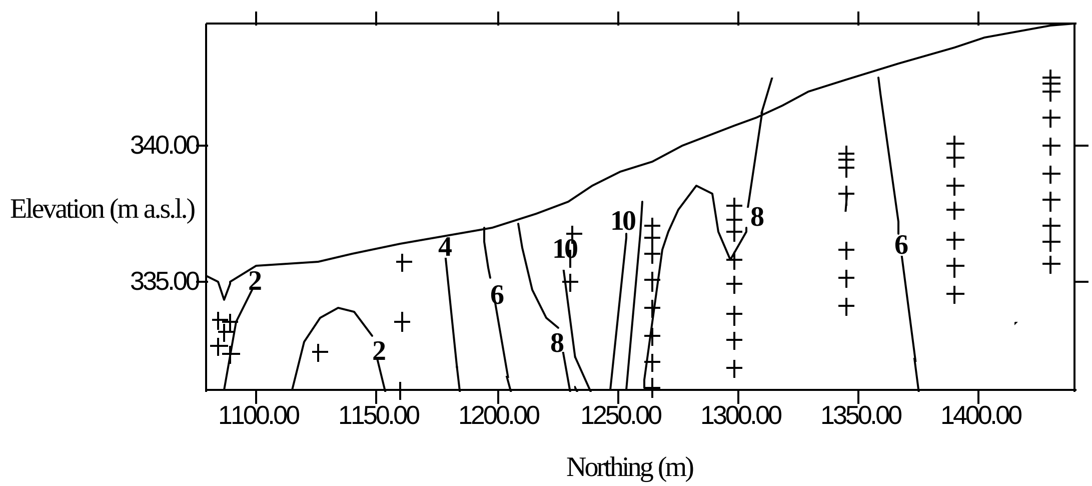
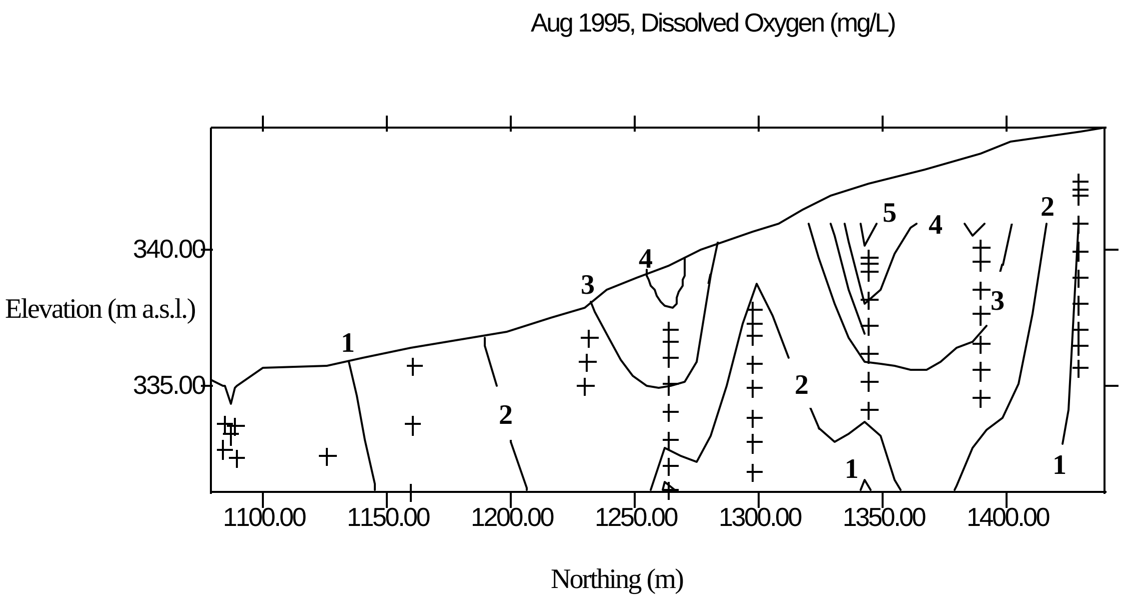
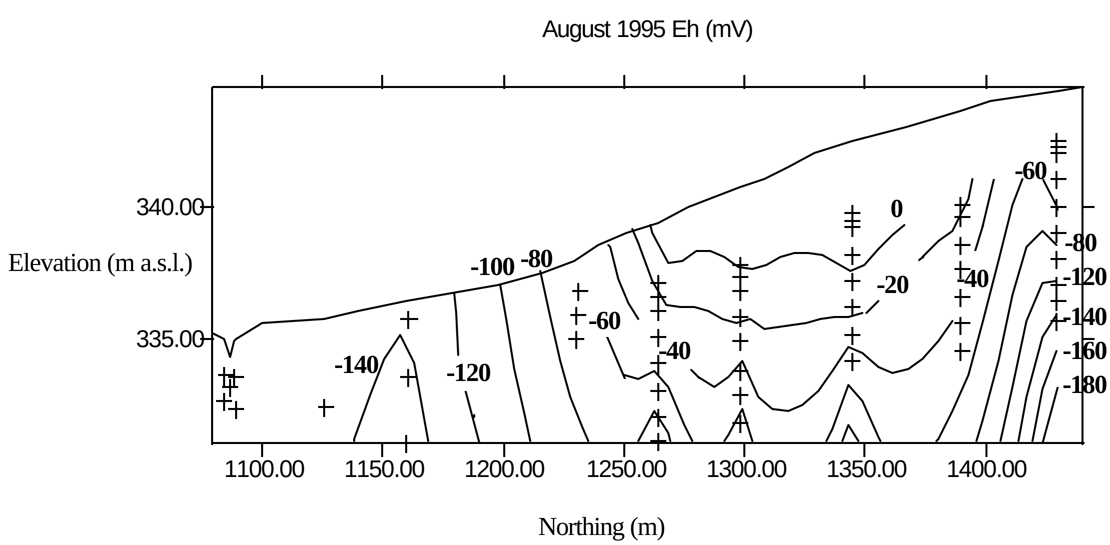


Fig. 40. Distribution of dissolved oxygen and Eh values measured in monitoring wells along the main north - south cross section during August, 1995.



spatial or temporal trends in the distribution of pH values. However, there were distinct spatial trends in Eh and dissolved oxygen values of groundwater samples. Fig. 40 shows values of Eh and dissolved oxygen for groundwater samples taken along the main north - south cross section through the study field during August, 1995. Both Eh and dissolved oxygen decrease with depth and with proximity to the ditch. All remaining geochemical measurements taken from Kintore monitoring wells at different sampling times are given in Appendix 10.

Examination of Figs. 38 and 40 reveals a positive correlation between groundwater zones that contain low values of nitrate and dissolved oxygen. This trend is consistent with loss of nitrate by the denitrification process which requires reducing conditions (Gillham and Cherry, 1978). The redoxcline for denitrification to occur is normally considered to be at or below 2 mg/L of dissolved oxygen. In this case, the 2 mg/L contour ranges in depth from essentially the ground surface in the lower part of the field to over 10 m below ground surface in the central part of the upper area of the field (Fig. 40). The range in depths of the redoxcline in this study are consistent with redoxcline depths reported by Robertson et al. (1996) for other locations in southern Ontario.

Since denitrification has the potential to have a very positive impact on groundwater quality, we undertook a more detailed investigation of the fate and distribution of nitrate beneath the uncultivated strip. A transect of wells in the northeastern part of the study field beginning at ML 2 and ending at DP 26 on the eastern side of the Logan drain (Fig 6) were sampled intensively during the spring of 1996. Details of this study are given in Cey et al. (1998b). The main conclusions were that due to groundwater geochemical conditions and the downward deflection of groundwater flow near the contact between the study field and the uncultivated strip, groundwater nitrate concentrations were reduced to non-detect below the uncultivated strip. These results reinforce the idea that uncultivated buffer areas not only protect surface waters from contaminants in overland flow, but may serve to remediate high-nitrate groundwater flowing from cultivated areas (Hill, 1990). But, the unique discovery of this study is that enhanced infiltration of water with low nitrate concentrations through the uncultivated strip deflects groundwater from the field downward where denitrification occurs under reducing conditions. Contaminated groundwater from the field does not actually flow through the shallow, organic-rich sediments beneath the uncultivated strip.

In order to complete the field mass balance of nitrogen and chloride, we must estimate the quantity of groundwater inputs and outputs for both species. As discussed above, much of the nitrate in groundwater is lost through denitrification at depth or beneath the uncultivated strip. However, groundwater must still contribute a small amount of nitrate to the Logan drain because surface runoff and tile drain inputs do not account for all of the increase in nitrate load between ditch stations C1 and C12 (Fig. 37). Indeed, significant concentrations of nitrate were found in monitoring wells MP 2 and MP 4 which were installed in the ditch bed (see Appendix 2b, Fig. 3 for the locations of these wells in the ditch bed). Nitrate concentrations in all MP wells installed in the ditch bed are shown in Fig. 41. Note that the width of the uncultivated strip is quite narrow (about 5 m) near wells MP 2 and MP 4 resulting in lower excess infiltration to deflect high nitrate groundwater. Also, there may be a minimum travel distance or residence time necessary for denitrification.

Based on the information given in Fig. 37, groundwater must contribute a greater proportion of chloride than nitrate to the Logan drain. As shown in Fig. 39, concentrations of chloride in wells near the ditch were between 2 to 4 ppm. The concentration of chloride in MP wells installed in the ditch bed was much higher as shown in Fig. 42. The discrepancy in chloride concentrations between groundwater samples taken from drive points (DP wells 8, 9, 16, 17, and 18 near the ditch in Figs. 6 and 39) and MP wells (Fig. 42) is difficult to explain. One possible explanation is that shallow groundwater not detected in the deeper wells converges and discharges at the creek. Unfortunately MP wells were only sampled two times and only during 1996.

The average concentrations of nitrate and chloride found in groundwater samples from MP wells are 1.8 and 9.0 ppm. It was noted above that tile drains contributed proportionally more

Fig. 41. Concentration of nitrate in groundwater samples taken from MP wells installed in the ditch bed.

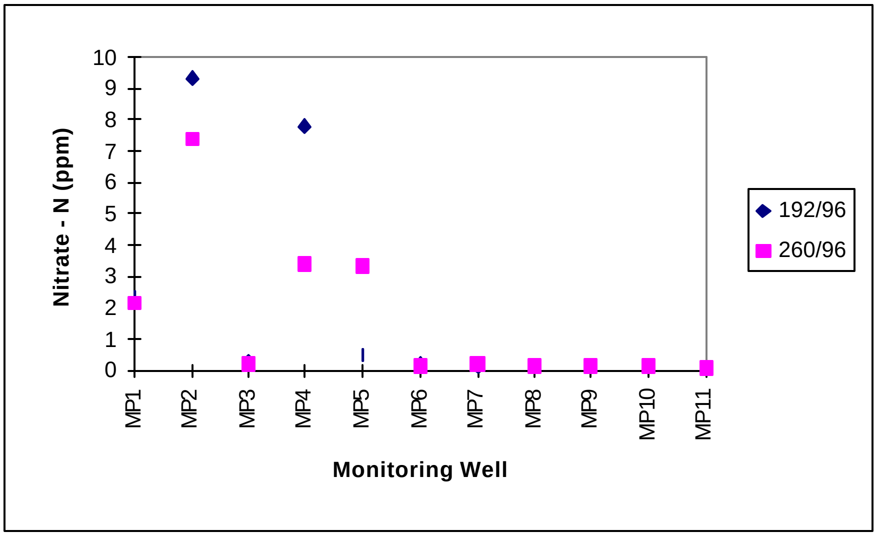
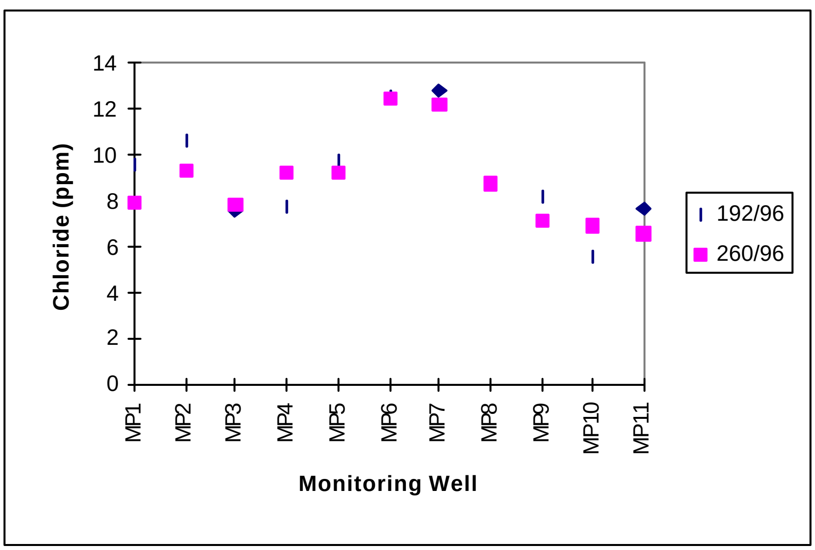



Fig. 42. Concentration of chloride in groundwater samples taken from MP wells installed in the ditch bed.



nitrate than chloride to the increase in loading between ditch stations C1 and C12 (Fig. 37). Therefore, since groundwater is the only other major source of water for the ditch, it must contribute proportionally more chloride than nitrate to the increase in loads between C1 and C12. The average concentrations of nitrate and chloride in groundwater can be calculated from the remaining increase in loads between C1 and C12 after subtracting the contribution due to the tiles and the secondary ditch. The concentrations of nitrate and chloride in groundwater based on this mass balance calculation are 1.2 and 9.8 ppm, respectively. These values are very close to the average concentrations of nitrate and chloride measured in groundwater samples taken from MP wells installed in the base of the ditch.

A summary of groundwater input and output of nitrate and chloride to and from the study field is given in Table 11. The amount of each species was calculated by the product of flow volumes and average annual concentrations at groundwater input and output boundaries.

| | $G_I$ N | $G_O$ N | $G_I$ Cl | $G_O$ Cl |
|---|---|---|---|---|
| 1995 | 8 | 0.4 | 24 | 0.4 |
| 1996 | 7.3 | 0.41 | 14.6 | 0.62 |

Table 11. Groundwater inputs ($G_I$) and outputs ($G_O$) of nitrate and chloride expressed as Kg N or Cl/Ha.

## 5.0 Summary of Nitrogen and Chloride Mass Balances

Table 12 documents the inputs and outputs of nitrogen and chloride from the Kintore site during 1995 and 1996.

| N | Inputs | | | Outputs | | | | | |
|---|---|---|---|---|---|---|---|---|---|
| | Fert. | PPE | $G_I$ | $G_o$ | Tile | Runoff | Crop | )$S_S$ | Unaccounted |
| 1995 | 6 | 15 | 8 | 0.4 | 34 | 2 | 218* | -24 | -187 |
| 1996 | 124 | 13 | 7 | 0.4 | 54 | 4 | 35 | 3 | 54 |

| Cl | Inputs | | | Outputs | | | | | |
|---|---|---|---|---|---|---|---|---|---|
| | Fert. | PPE | $G_I$ | $G_o$ | Tile | Runoff | Crop | )$S_S$ | Unaccounted |
| 1995 | 15 | 15 | 24 | 0.4 | 34 | 4 | 0 | -6 | 10 |
| 1996 | 14 | 9 | 15 | 0.6 | 23 | 8 | 0 | 10 | 16 |

Table 12. Summary of nitrogen and chloride balances for 1995 and 1996.

Unaccounted values are the differences between input and output not accounted for in the change in storage. A negative unaccounted value represents an unaccounted surplus in the amount of output and vice-versa for a positive unaccounted value. Interpretation of results of the nitrogen balance for 1995 is complicated by the presence of alfalfa. If we assume that the alfalfa does not utilize any nitrogen from the soil, then the crop output value of 218.0 Kg N/Ha can be left out of the mass balance calculation. If we subtract all outputs of N from the total inputs, the remainder is actually -7 Kg N/Ha. This negative value combined with the increase in soil storage of 24 Kg N/ha gives an overall contribution of 31 Kg N/Ha to the soil profile by the alfalfa crop. Note that the amounts of chloride unaccounted for in the mass balance measurements are much less than the unaccounted nitrogen values. Nitrogen is much more reactive in the environment and hence it is more difficult to measure a nitrogen balance.  Also bear in my mind that we did not measure gaseous losses of nitrogen either through direct flux of ammonia to the atmosphere from applied manure or losses through denitrification.  In a separate project completed at the Kintore study site, it was demonstrated that denitrification is a significant process in the attenuation of nitrate-laden groundwater from the field entering the riparian zone (Cey et al., 1998b).

**Part 3 Herbicides in Tiles and Groundwater Below Tiles**

Herbicides applied in the fall of 1995 to kill the alfalfa crop were 2-4D and Roundup.  The former is strongly adsorbed to soil particles and Roundup is technically difficult to detect in water samples (personal communication, Prof. C. Hall, University of Guelph, 1995).  Therefore no herbicide analyses were performed in 1995. Of the five herbicides applied in the spring of 1996, we selected atrazine for an investigation of herbicide leaching below the root zone.  The main objective of the analysis was to determine if atrazine followed the same general transport paths as nitrate and chloride from the root zone to the Logan drain.

Atrazine residue analyses were performed at the Dept. of Environmental Biology, University of Guelph.  The limit of detection was 0.1 ppb.  The maximum concentration detected was 36 ppb from a sample taken at tile outlet C2 on day 160, about 12 days following application.  The average amount found in samples (total 67) taken from tile drains was 5 ppb. The highest concentration detected in groundwater was 4 ppb from a sample taken on day 156 from monitoring well ML 7 - 2. The average concentration in groundwater samples (total 10) was 2 ppb. Overall, of the 100 analyses performed, 46% contained more than 1 ppb of atrazine, which is the current maximum limit for total pesticide concentration in drinking water for many European countries. Results are listed in Appendix 13.

# Part 4 Biological Effects on Logan Drain

## 1.0 Introduction

Agricultural activities have been identified as major contributors to environmental stress. Those which lead to the pollution of watercourses include clearing of the land, sub-surface tile drainage, channelization, tilling, planting, grazing and the application of fertilizers, pesticides and herbicides. Ground water and overland flow convey organic and inorganic sediments and the applied chemicals from the disturbed soil into the watercourses draining agricultural catchments. Macroinvertebrates offer an excellent way to examine biological aspects of water quality. Rosenberg and Resh (1996) summarized the characteristics which lend macroinvertebrates, and aquatic insects in particular, to biomonitoring of aquatic ecosystems: (1) They are ubiquitous; (2) The large number of species exhibit a range of responses; (3) They are relatively sedentary; (4) They can have long life cycles.

Many assessments of water pollution have focused on alterations in community diversity, loss of sensitive species, or increases in the number of tolerant species as a means to evaluate biotic effects. The focus of the present study was to examine ecosystem structure and function by estimating the magnitude and pathways of energy flow at the species level both upstream and downstream of the control site. An attempt was made to relate the differences found in the benthic communities to the agricultural activities on the study site.

Agricultural inputs known to impact benthic communities include suspended sediments, nutrients, organic wastes, pesticides and herbicides (Dance and Hynes 1980). These are briefly discussed below in order to emphasize the complexity of land use responses in aquatic ecosystems.

**1.2 Sediments**

An increased rate of erosion and the subsequent sediment loading of lotic environments is one of the physical impacts imposed on aquatic ecosystems. Alterations to the land through sub-surface tile drainage, channelization, construction of ditches and land clearing can create high sediment loading during initial construction and increased delivery rates in the long term. Tilling, planting and the trampling of riverbanks by grazing animals loosens the soil and may also increase soil erosion. Suspended solids may interfere with invertebrate respiration, abrade the exoskeleton or hinder food collection of filter feeders. Reduced light penetration may limit algal and macrophyte growth creating secondary effects which result in depletion of habitat and high quality autochthonous food sources. There have been many attempts to determine the effects of sedimentation on lotic benthic population (Minshall and Minshall 1977; Resh 1977; Williams 1980; Lenat et al. 1981; Erman and Erman 1984). The analysis is complicated by the effects of mean particle size, size heterogeneity and particle surface roughness. The general trend of these studies is that as mean particle size increases, a corresponding increase is observed in overall numbers of invertebrates, number of taxa, number of Ephemeroptera and secondary production of Trichoptera. Rabeni and Minshall (1977) noted that densities of Chironomidae increased, while densities of Ephemeroptera, Trichoptera and Plecoptera decreased following the application of a layer of silt to the substrate. A similar dominance of Chironomidae and reduction of *Baetis*, *Simulium*, *Cheumatopsyche* and *Stenelmis* was observed by Berkman and Rabeni (1986) in areas of high sedimentation. Sedimentation can result in the loss or creation of habitat for specific organisms by reducing exposed rock surfaces and smothering periphyton populations. In cases where depth is reduced, air temperatures may exert a greater influence on water temperature fluctuations. Some organisms benefit from sedimentation while many others do not.

**1.3 Pesticides and Herbicides**

Ground and surface water from land treated with pesticides can be highly toxic to non-target aquatic organisms. Dosdall and Lehmkuhl (1989) observed catastrophic drift up to 107 km downstream from the point of methoxychlor treatment in the North Saskatchewan River. The methoxychlor drift response was dependent on species, distance from the injection site, and the time after injection. Initial catastrophic drift and high mortality are typically followed by a reversion phase and recolonization through drift from upstream reaches, migration from the hyporheos, and egg hatching (Wallace and Hynes 1981). Wallace et al. (1991) studied the effect of the insecticide methoxychlor on streams over a 5-year period. The depleted macroinvertebrate population coincided with a two-order-of-magnitude decrease of fine particulate organic matter (FPOM) exported downstream over the 3-year treatment and 1-year recovery period. The secondary impact of the reduced organic matter available downstream was not determined. The toxicity of pesticides are probably highly variable and a function of invertebrate morphology, lifestage, size, habitat and sex.

Herbicides have lower acute toxicity to animals than pesticides and the effects on aquatic communities at low levels can be difficult to detect (Cooper 1993). Some are toxic to algae and by reducing primary production may limit habitat, food availability and dissolved oxygen concentrations.

### 1.4 Nutrients

Inorganic and organic forms of nitrogen and phosphorous may appear in rivers and streams from the application of commercial fertilizers and animal wastes. These important nutrients may enhance algal and macrophytic growth, alter the balance between autochthonous and allochthonous production, and accelerate eutrophication. Excessive algal growth can alter benthic species composition because of physical changes in habitat or altered daily fluctuations in dissolved oxygen concentrations. In extreme cases, the decay of plant biomass or large amounts of organic matter from animal wastes may lead to the proliferation of aerobic decomposers and reduced oxygen levels. Stream characteristics such as flow rate, canopy, order, depth and substrate type can also influence primary production levels.

**1.5 Temperature**

Agricultural activities may also impact water temperatures. As mentioned earlier, sedimentation may reduce water depths and increase the diel, seasonal and annual water temperature fluctuations. Clearing vegetation from the riparian zone will increase irradiance and primary production in light limited cases, and raise temperatures thus decreasing the capacity to dissolve oxygen (Brazier et al. 1973; Murphy et al. 1981; Rosenfeld and Roff 1991; Bilby and Bisson 1992). Even small changes in temperatures can affect growth, metabolism, reproduction, emergence and the distribution of aquatic insects. Vannote and Sweeney (1980) concluded that each of the species they studied had an optimal temperature, above or below which larval tissue growth was suppressed more than adult tissue development resulting in smaller adults and reduced fecundity. The importance of water temperature is emphasized in the work of Benke and Parsons (1990) and Benke and Jacobi (1994). They developed relationships predicting daily growth rates for black flies and mayflies, in blackwater streams on the Georgia Coastal

Plain, in which daily water temperature was the only independent variable.

Increased temperatures are not necessarily detrimental to aquatic ecosystems, however. Bilby and Bisson (1992) compared allochthonous and autochthonous contributions to the trophic support of fish populations in clear-cut and old-growth forested streams. They concluded that the higher fish production in the clear-cut site, despite a two-fold larger combined allochthonous and autochthonous input in the old-growth site, reflected the greater production of autochthonous algae which is more nutritious than allochthonous material.

## 2.0 Research Design

The field site is located at Lot 19 Concession 12 in East Nissouri Township, Oxford County near Kintore, Ontario. The 57 hectare cash crop and hog farm is operated by Frank Aarts. A 5 hectare field was selected for detailed study (Fig. 1). The objective of the investigation was to assess the health of the aquatic ecosystem in the Logan Drain by comparing the benthic communities above and below the study field.

The Site 1 extends upstream 75 metres from the point at which the stream emerges from the wooded area (Site 1; Fig. 1). Site 5 encompasses 20 metres of the stream within the wooded area just below the field (Site 5; Fig. 1). Sites 2, 3 and 4 are located just below major inflows from a secondary ditch and several tile openings. Data collected from 1994 showed that water entering Logan Drain from the secondary ditch and tile drains C6 to C9 had higher nitrogen concentrations than the drain itself. Intensive sampling to estimate secondary production and assess the overall impact of agricultural activities on the fauna was conducted at Sites 1 and 5. Intermediate sites were sampled less frequently to clarify the rate of change of community structure through the study reach.

The drain channel is 1 to 2 metres wide, the substrate consists of gravel up to boulder size and the banks are lined with grasses and shrubs. Sites 1, 2 and 5 have forested canopies while the remaining two are open and subject to elevated irradiance. Alfalfa was grown on the north side of Logan Drain in 1995 and corn to the south. Both sides were planted to corn in 1996.

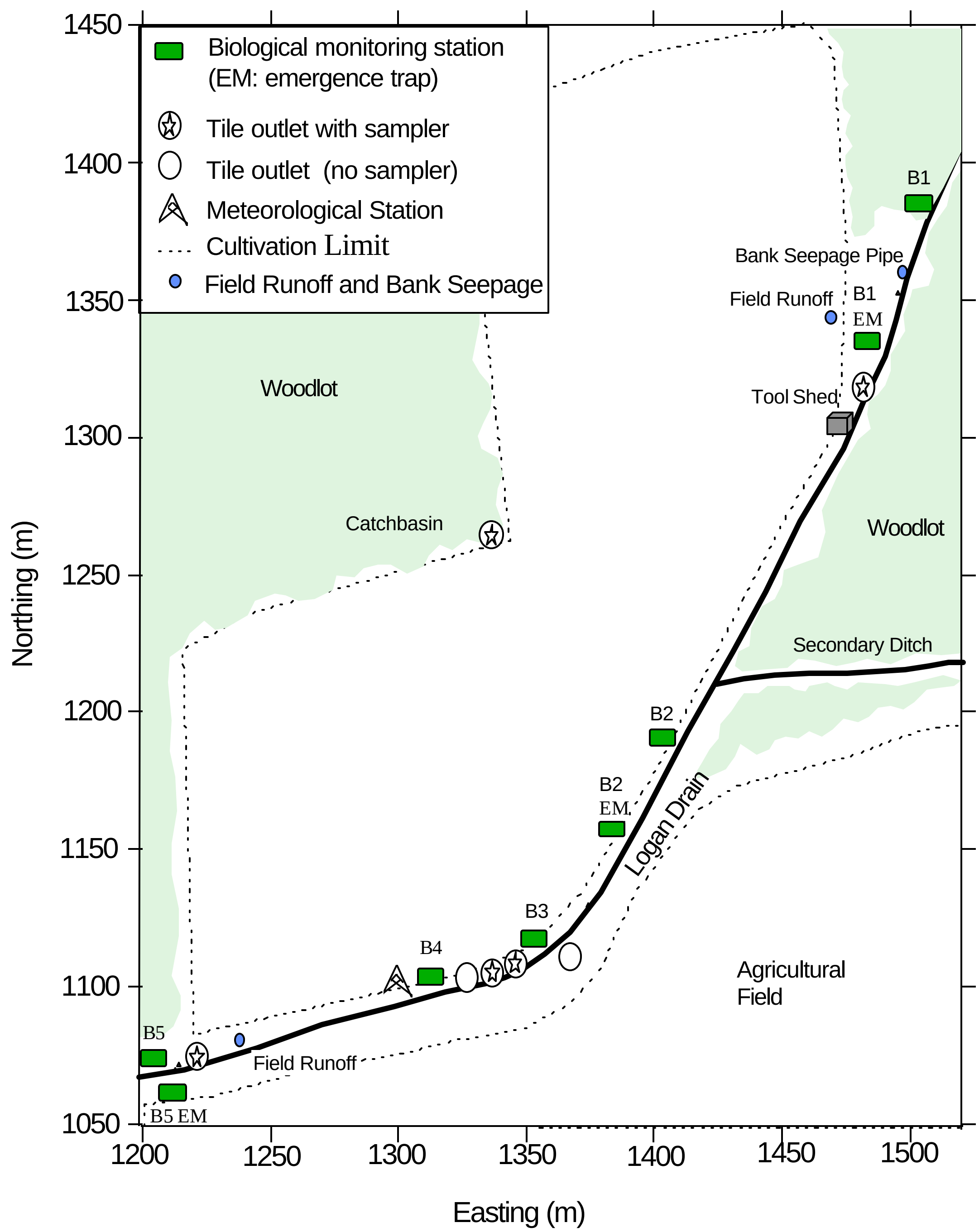



Figure 1. Biological monitoring stations in Logan Drain, Kintore, Ontario. Benthic and emergence sampling stations shown.

**3.0 Materials and Methods**

This study used biweekly, stratified random Surber sampling of the two main sites to measure invertebrate density, biomass and secondary production in the Logan Drain. The sampling began 25 May 1995, became monthly between November 1995 and April 1996, and returned to biweekly until October 1996. Five replicates were collected from both the upstream and downstream sites on each sample date using 10% formalin to preserve the samples in the field. In order to reduce the physical impact of sampling in this small stream, all work was conducted from planks spanning the drain, using a mini-Surber (0.01 $m^2$) sampling device with a mesh size of 0.200 mm. The substrate was disturbed to a depth of 5 cm. Three floating emergence traps were emptied weekly from 19 April 1995 until 23 October 1995 to reinforce the cohort lifecycle data obtained from the benthic samples and aid in identification of the aquatic insect species. The upstream (Site 1em), midstream (Site 2em), and downstream (Site 5em) emergence trap locations are shown on Figure 1. Sites 2, 3 and 4 were sampled monthly from 26 April 1996 to 27 September 1996 using the same methods.

**3.1 Density**

In the laboratory, each Surber sample was rinsed with water and agitated to suspend the preserved animals. The supernatant was then poured through a 0.100-mm sieve to separate the animals from the debris and the process repeated until no further animals were observed in the supernatant. The animals from each of the 390 samples were sorted to the ordinal level (Diptera were sorted to family) at 6x magnification and preserved in vials of 70% ethanol. All animals collected between 25 May 1995 and 09 May 1996 were identified, counted, maximum sclerotized head dimension recorded (+/- 0.02 mm) and then stored in 70% ethanol until further processing.

### 3.2 Biomass

The maximum sclerotized head dimension was head width for all animals collected except for Ceratopogonidae. These small midge larvae have elongated heads making head length the logical dimension to correlate with weight.

Linear regressions relating dry mass to head dimension for each of the common taxa were developed using animals from the stored samples. Head dimensions were measured from over 30 specimens for each species, these individuals were then oven dried for 48 hours at 60EC and dry mass obtained using a Cahn C-31 microbalance. The animals used covered the complete range of head dimensions found for each taxa. In the case of

Chironomidae, animals in the first two instars were weighed in groups of five to minimize measurement errors. Equations were then developed using linear regressions of the natural logarithm of dry mass on the natural logarithm of head dimension. These equations were of the form:

$$\text{Ln (dry mass)} = a + b^{*}\text{Ln (head dimension)}$$

where: a and b are regression coefficients for each taxa. Larvae of Tipulidae and Tabanidae do not have external sclerotized body parts; because their numbers were relatively small, every animal was weighed individually. Dry mass was not adjusted to allow for storage in both formalin and ethanol (Mason 1983).

### 3.3 Secondary Production

Secondary production is a measure of two important population parameters, individual growth and population survivorship. They are combined to estimate “that amount of tissue elaborated per unit time per unit area, regardless of its fate” (Waters and Crawford 1973). Production estimates also aid in quantifying energy and material transfer through the different trophic levels of a community.

While cohort methods could have been used to estimate secondary production for the univoltine taxa, they are not applicable to the more complex lifestyles of the multivoltine Chironomidae or Ceratopogonidae. To ensure uniform results, the size-frequency method (Hamilton 1969, Benke 1984) was utilized for all taxa. Instrumental to this method is an accurate estimate of the mean cohort production interval (CPI) for each taxon. For most of the invertebrates found in the Logan Drain, the CPI could be estimated from the field data. A combination of field data and literature values of growth rates (Mackey 1977) was necessary to determine the CPI of the Chironomidae and Ceratopogonidae.

### 3.4 Biotic Indices

Two biotic indices were calculated to evaluate the impact of the control site on the benthic fauna of Logan Drain. The Hilsenhoff Biotic Index (BI) reflects organic and nutrient pollution (Hilsenhoff 1987, Hilsenhoff 1988a) which may result in reduced dissolved oxygen levels and stressed invertebrates. The BI is based on the number of each invertebrate taxon found at a site weighted by that taxon’s sensitivity to pollution. This index was calculated for samples collected from May 1995 to May 1996, in which the fauna were identified to species level. An alternative index developed by Hilsenhoff (Hilsenhoff 1988b) is designed for rapid field assessment (FBI) and the animals need only be identified to the family level. This index was used to evaluate the communities at the five sites sampled in the summer of 1996. With both of these methods, a higher index score indicates greater organic pollution.

The EPT richness index is the number of Ephemeroptera, Plecoptera and Trichoptera (all considered to be sensitive to pollution) taxa found at a site on a given date. Reduced numbers are an indication of stress. The EPT richness index was used to evaluate the Drain habitat at the upstream and downstream sites between May 1995 and May 1996.

### 3.5 Abiotic Data

Temperature was recorded weekly at each sample site from 22 June 1995 through 09 November 1995, monthly from 07 December 1995 through 26 April 1996 and then biweekly until 10 October 1996. Substrate was quantified as a function of exposed surface area for specific sediment sizes (fines <4 mm, small gravel 4-25 mm, medium gravel 25-50 mm, large gravel 50-75 mm, small cobble 75-150 mm, medium cobble 150-225 mm, large cobble 225-300 mm, small boulder 300-600, large boulder >600) at each of the study sites. Abiotic data gathered by the Earth Sciences Department at the University of Waterloo included concentrations of nitrate-nitrogen, ammonium-nitrogen, total soluble nitrogen, dissolved oxygen and organic carbon as well as pH, electrical conductivity and continuous water temperature recordings at the tool shed and the meteorological station. The Upper Thames Conservation Authority developed stage-discharge curves at two locations along the drain.

## 4.0 Results

### 4.1 Abiotic

Mean values of the relevant water chemistry data are summarized in Table 1. Both nitrogen concentrations and temperature changed along the course of the drain through the study site. Nitrate nitrogen concentrations at the downstream site were approximately two times greater than at Site 1.

Temperature differences were also detected between some of the biological sampling sites. Temperatures at the two main wooded sites were not significantly different ($t = -1.000$, $df = 39$, $p = 0.323$), nor were those between the wooded Site 2 and the downstream site ($t = -0.329$, $df = 40$, $p = 0.744$). The cleared Site 3 was significantly different from Sites 1 and 2 (Site 1 $t = -3.246$, $df = 37$, $p = 0.002$; Site 2 $t = -2.372$, $df = 37$, $p = 0.023$) and the cleared Site 4 was significantly different from each of the three wooded sites (Site 1 $t = -3.076$, $df = 14$, $p = 0.008$; Site 2 $t = -2.736$, $df = 14$, $p = 0\ 016$; Site 5 $t = -2.824$, $df = 14$, $p = 0.014$). Mean temperatures increased downstream from Site 1 until reaching the southwest woodlot, where they began to fall.

The substrate changed progressively from an even distribution of boulders, cobble, gravel and fines at Site 1 to predominantly gravel and fines at Site 4 (Fig. 2). Some recovery was evident at the downstream site with a shift back to cobble and some exposed boulders.

| Site | Mean Temp. (EC) | Mean Dissolved Oxygen (mg/l) | Mean pH | Mean Electrical Conductivity (Fsiemens/cm) | Mean DOC (mg/l) | Mean $NO_3^-$ Nitrogen (mg/l) | Mean $NH_4$ Nitrogen (mg/l) | Mean Total Soluble Nitrogen (mg/l) |
|---|---|---|---|---|---|---|---|---|
| 1 | 11.6 (15) | 10.06 (2) | 8.07 (2) | 455 (2) | 3.30 (21) | 1.54 (77) | 0.04 (78) | 2.22 (10) |
| 2 | 12.0 (15) | 10.16 (2) | 8.07 (2) | 458 (2) | 1.74 (19) | 2.91 (78) | 0.03 (78) | 5.12 (10) |
| 3 | 12.6 (15) | na | na | na | na | na | na | na |
| 4 | 12.6 (15) | 10.04 (2) | 7.94 (2) | 459 (2) | 2.24 (20) | 3.05 (77) | 0.04 (78) | 3.51 (10) |
| 5 | 12.1 (15) | 9.78 (2) | 7.99 (2) | 461 (2) | 2.15 (21) | 3.19 (80) | 0.05 (79) | 3.51 (10) |

(na appears when data is not available at a particular site)

Table 1. Water chemistry data at the five sample sites in Logan Drain, Kintore. Values in brackets are the number of measurements included in the mean.

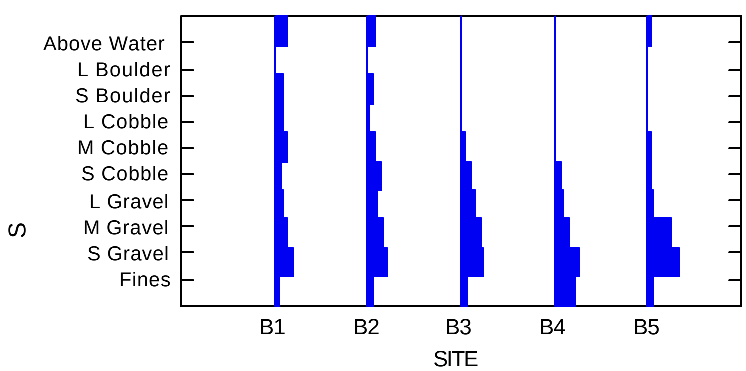



Figure 2. Percent of substrate type by surface area at each site.

**4.2 Density and Biomass (May 1995 to May 1996)**

Mean annual density and biomass for the main groups of invertebrates at both the upstream and the downstream sites are presented in Table 2. Relative to Site 1, the fauna at Site 2 included fewer Ephemeroptera, Plecoptera, Trichoptera and Oligochaetae but larger numbers of Coleoptera and Diptera. Individual species within these groups did not always follow this trend. Mean annual biomass mirrored the densities with the exception of the Chironomidae in which biomass was slightly lower downstream. Mean individual biomass was lower in each of the major groups at the downstream site.

The density of Diptera was roughly equal to that of all other groups combined at both sites. The Chironomidae, the largest family in this group, were represented by 47 taxa. Of the 13 most common chironomid taxa, all but 4 were more common downstream. The mean biomass per animal was greater upstream for every chironomid species and for the family overall as well. This led to a drop in mean annual biomass downstream of 12% in spite of the 15% increase in mean annual density. The densities of the other Diptera (Ceratopogonidae, Tipulidae and Tabanidae) also increased downstream, while mean biomass per animal decreased and biomass remained stable.

| | Mean Annual Density ($N/m^2$) | | | Mean Annual Biomass ($mg/m^2$) | | | Annual Production ($mg/m^2$) | | | Mean Individual Biomass (mg/individual) | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Taxon | Site 1 | Site 5 | % Change | Site 1 | Site 5 | % Change | Site 1 | Site 5 | % Change | Site 1 | Site 5 | % Change |
| **Diptera** | 24718 | 30468 | 23 | 3014 | 2960 | -2 | 20454 | 20958 | 2 | 0.127 | 0.104 | -18 |
| Chironomidae | 19374 | 22184 | 15 | 381 | 336 | -12 | 6434 | 6988 | 9 | 0.020 | 0.015 | -25 |
| Tipulidae | 3091 | 4153 | 34 | 2565 | 2531 | -1 | 13568 | 13383 | -1 | 0.830 | 0.609 | -27 |
| Tabanidae | 800 | 1620 | 103 | 57 | 82 | 44 | 300 | 431 | 44 | 0.071 | 0.051 | -28 |
| Ceratopogonidae | 376 | 497 | 32 | 11 | 11 | 0 | 152 | 156 | 3 | 0.029 | 0.022 | -24 |
| Simulidae | 1077 | 2014 | 87 | na | na | na | na | na | na | na | na | na |
| **Coleoptera** | 9464 | 9950 | 5 | 536 | 575 | 7 | 1064 | 1197 | 13 | 0.057 | 0.058 | 2 |
| **Ephemeroptera** | 4224 | 3781 | -10 | 446 | 333 | -25 | 4649 | 3450 | -26 | 0.106 | 0.088 | -17 |
| **Plecoptera** | 914 | 852 | -7 | 48 | 25 | -48 | 353 | 227 | -36 | 0.053 | 0.029 | -45 |
| **Trichoptera** | 8886 | 5117 | -42 | 4247 | 1852 | -56 | 21069 | 8827 | -58 | 0.478 | 0.362 | -24 |
| **Oligochaetae** | 6653 | 3720 | -44 | na | na | na | na | na | na | na | na | na |
| Naididae | 4585 | 804 | -82 | na | na | na | na | na | na | na | na | na |
| Tubificidae | 1904 | 2700 | 42 | na | na | na | na | na | na | na | na | na |
| **Total** | 54859 | 53888 | -2 | 8291 | 5745 | -31 | 47589 | 34659 | -27 | na | na | na |

(na - not determined for this taxa)

Table 2. Mean annual density, mean annual biomass, annual production and mean individual biomass for the major groups of macroinvertebrates at both sites in Logan Drain, Kintore. Data collected from May 1995 to May 1996.

Nine species of Coleoptera were found at the two sites. The riffle beetle *Optioservus fastiditus* made up more than 99% of the beetle population and was found in similar numbers and biomass at both sites. Mean annual density and biomass increased by 5% and 7%, respectively, downstream and the mean biomass per animal remained constant.

Four taxa made up the Ephemeroptera population at both sites (Table 3). *Baetis brunneicolor* was the most common mayfly at the upstream site and its decline in abundance by 49% accounted for the reduced mayfly presence downstream. *Baetis macdunnoughi*, a smaller and more tolerant species, increased in abundance by 114% downstream. An overall reduction in Mayfly density of 10% was accompanied by a decrease in biomass by 25%. This was a result of the mean biomass per mayfly dropping from 0.106 mg upstream to 0.088 mg downstream.

Of the two species of Plecoptera found in this section of the Logan Drain, *Allocapnia vivipara* dominated in abundance and in biomass at both sites. Density and biomass at the downstream site were 7% and 48% lower, respectively. The more pronounced change in biomass was due to the mean biomass per animal dropping from 0.053 mg to 0.029 mg.

Seven species of Trichoptera were common to both sites and sufficiently abundant to analyze. Mean annual biomass was higher upstream for each of these taxa. Densities were also greater upstream for each with the exception of *Cheumatopsyche oxa* (a tolerant hydropsychidae) and *Lepidostoma costalis* which were 15% and 2% higher downstream, respectively. As with the orders Ephemeroptera and Plecoptera, the mean biomass per animal decreased from 0.478 mg upstream to 0.362 mg downstream.

| Functional Feeding Group | Taxon | Mean Annual Density (N/m$^2$) Site 1 | Site 5 | % Change | Mean Annual Biomass (mg/m$^2$) Site 1 | Site 5 | % Change | Annual Production (mg/m$^2$) Site 1 | Site 5 | % Change |
|---|---|---|---|---|---|---|---|---|---|---|
| **Shredders** | **Tipulidae** | | | | | | | | | |
| shredder, collector (G) | *Tipula ssp* | 1948 | 1356 | -30 | 2542 | 2412 | -5 | 13449 | 12758 | -5 |
| | **Plecoptera** | | | | | | | | | |
| shredder-detritivore | *Allocapnia vivipara* | 876 | 794 | -9 | 45.6 | 19.5 | -57 | 344 | 201 | -42 |
| shredder-detritivore | *Amphinemura delosa* | 38 | 58 | 53 | 2.3 | 5.0 | 117 | 9 | 26.0 | 188 |
| | **Trichoptera** | | | | | | | | | |
| shredder | *Lepidostoma costalis* | 46 | 48 | 2 | 11.3 | 10.3 | -9 | 41 | 41 | 0 |
| | **Total Shredders** | **2908** | **2256** | **-22** | **2601** | **2447** | **-6** | **13843** | **13026** | **-6** |
| **Collectors (F)** | **Trichoptera** | | | | | | | | | |
| collector (F), predator | *Ceratopsyche slossonae* | 5194 | 2148 | -59 | 2929 | 981.4 | -66 | 15024 | 4647 | -69 |
| collector (F), predator | *Cheumatopsyche oxa* | 1612 | 1849 | 15 | 519.6 | 378.5 | -27 | 2527 | 1613 | -36 |
| collector (F), predator | *Diplectrona modesta* | 367 | 70 | -81 | 215.6 | 55.7 | -74 | 673 | 134 | -80 |
| collector (F) | *Chimarra obscura* | 593 | 74 | -88 | 137.0 | 40.0 | -71 | 423 | 211 | -50 |
| collector (F) | *Dolophilodes distinctus* | 174 | 110 | -64 | 51.7 | 45.9 | -11 | 328 | 280 | -15 |
| | **Chironomidae** | | | | | | | | | |
| collectors (F and G) | *Micropsectra* Group | 1300 | 1689 | 30 | 13.6 | 14.6 | 7 | 447 | 552 | 23 |
| collector (F) | *Rheotanytarsus exiguus* | 1056 | 2581 | 144 | 14.3 | 25.9 | 81 | 218 | 535 | 145 |
| | **Total Collectors (F)** | **10296** | **8521** | **-17** | **3881** | **1542** | **-60** | **19640** | **7972** | **-59** |
| **Collectors (G)** | **Ephemeroptera** | | | | | | | | | |
| collector (G) | *Baetis brunneicolor* | 3134 | 1612 | -49 | 214.6 | 171.2 | -20 | 3485 | 2100 | -40 |
| collector (G) | *Baetis macdunnoughi* | 863 | 1844 | 114 | 32.5 | 54.0 | 66 | 398 | 723 | 82 |
| collector (G) | *Paraleptophlebia debilis* | 46 | 89 | 93 | 2.4 | 5.3 | 121 | 36 | 71 | 97 |
| | **Tipulidae** | | | | | | | | | |
| collector (G) | *Antocha sp* | 327 | 63 | -81 | 2 | 0.5 | -75 | 10 | 3 | -70 |
| | **Chironomidae** | | | | | | | | | |
| collector (G) | *Polypedilum ssp* | 3265 | 3184 | 34 | 39.8 | 52.2 | 31 | 179 | 258 | 44 |
| collector (G) | *Parametriocnemus lundbeckii* | 4154 | 4317 | 4 | 49.3 | 46.0 | -7 | 837 | 884 | 6 |
| collector (G) | *Orthocladius* Group | 1355 | 1818 | 34 | 39.8 | 52.2 | 31 | 179 | 258 | 44 |
| collector (G) | *Eukiefferiella claripennis* | 501 | 722 | 44 | 7.2 | 7.9 | 10 | 75 | 101 | 35 |
| collector (G) | *Tvetenia paucunca* | 4675 | 5345 | 14 | 41.1 | 39.7 | -3 | 894 | 965 | 8 |
| collector (G) | *Parakiefferiella sp* | 661 | 699 | 6 | 4.6 | 4.6 | 0 | 78 | 112 | 44 |
| collector (G) | *Thienemanniella sp* | 594 | 320 | -46 | 4.3 | 2.3 | -47 | 97 | 82 | -15 |
| collector (G) | *Corynoneura sp* | 258 | 330 | 28 | 0.7 | 1.0 | 43 | 20 | 14 | -30 |
| collector (G) | *Paracricotopus sp* | 97 | 72 | -26 | 1.0 | 0.6 | -40 | 17 | 7 | -59 |
| collector (G) | *Diamesa nivoriunda* | 106 | 179 | 69 | 35.7 | 31.9 | -11 | 295 | 351 | 19 |
| | **Total Collectors (G)** | **20036** | **20594** | **3** | **574** | **509** | **-11** | **8743** | **8235** | **-6** |
| **Scrapers** | **Trichoptera** | | | | | | | | | |
| scraper | *Neophylax nacatus* | 900 | 818 | -9 | 382.9 | 339.7 | -11 | 2053 | 1901 | -7 |
| | **Coleoptera** | | | | | | | | | |
| scraper, collector (G) | *Optioservus fastiditus* | 9464 | 9950 | 5 | 535.7 | 575.4 | 7 | 1064 | 1197 | 13 |
| | **Ephemeroptera** | | | | | | | | | |
| scraper, collector (G) | *Stenonema vicarium* | 181 | 236 | 30 | 196.9 | 102.0 | -48 | 730 | 556 | -24 |
| | **Total Scrapers** | **10545** | **11004** | **4** | **1116** | **1017** | **-9** | **3847** | **3654** | **-5** |
| **Predators** | **Chironomidae** | | | | | | | | | |
| predator | *Trissopelopia ogemawi* | 1352 | 928 | -31 | 30.7 | 17.7 | -42 | 955 | 563 | -41 |
| | **Tabanidae** | | | | | | | | | |
| predator | *Chrysops sp* | 800 | 1620 | 103 | 57 | 81.5 | 43 | 300 | 431 | 44 |
| predators | **Ceratopogonidae** | 376 | 497 | 32 | 11.0 | 11.3 | 3 | 152 | 156 | 3 |
| | **Tipulidae** | | | | | | | | | |
| predator | *Dicranota sp* | 816 | 2734 | 235 | 21 | 118 | 462 | 109 | 622 | 471 |
| | **Total Predators** | **3344** | **5779** | **73** | **120** | **229** | **91** | **1516** | **1772** | **17** |

Table 3. Mean annual density, biomass and production for the common taxa.

Although biomass was not determined for the Oligochaetae, densities dropped by 44% at the downstream site. An 82% decline in the Naididae accounted for the change. A 42% increase in the Tubificidae population was also observed downstream.

Both density and biomass of the shredders and filter-feeders fell downstream. The scrapers and gathering collectors maintained stable densities but had a lower mean annual biomass at Site 2. The predators were the only functional feeding group to substantially increase in density and biomass at the downstream site.

**4.3 Density and Biomass (May to October 1996)**

Comparison between the upstream, downstream and the three sites internal to the control area, was limited to the ordinal level for all groups except the Diptera which were sorted to family. Even at this coarse level of analysis, several trends were obvious. Densities of both Diptera and Coleoptera increased downstream (Table 4). In particular, the chironomid population increased by 118%, from 62% to 75% of the total invertebrate density.

**4.4 Secondary Production**

Overall secondary production was 47.59 g/m$^2$ at Site 1 (upstream) and 34.66 g/m$^2$ at Site 5 (downstream). This change was solely attributable to the lower production of the 3 pollution sensitive orders (Ephemeroptera, Plecoptera and Trichoptera); production by the tolerant animals (Chironomidae, Ceratopogonidae, Tipulidae, Tabanidae and Coleoptera) remained stable or increased downstream. When the sites are compared by functional groups it becomes apparent that only amongst the filtering collectors did production fall substantially downstream. In the other functional

| | Mean Summer Density ($N/m^2$) | | | | |
|---|---|---|---|---|---|
| Taxon | Site 1 | Site 2 | Site 3 | Site 4 | Site 5 |
| **Diptera** | 2089 | 3363 | 3578 | 3949 | 4356 |
| Chironomidae | 1963 | 3242 | 3410 | 3861 | 4275 |
| Tipulidae | 15 | 24 | 31 | 20 | 18 |
| Tabanidae | 1 | 3 | 10 | 3 | 5 |
| Ceratopogonidae | 31 | 29 | 37 | 33 | 27 |
| Simulidae | 79 | 65 | 90 | 32 | 31 |
| **Coleoptera** | 467 | 509 | 780 | 850 | 665 |
| **Ephemeroptera** | 336 | 226 | 453 | 171 | 285 |
| **Plecoptera** | 67 | 49 | 37 | 62 | 152 |
| **Trichoptera** | 250 | 296 | 363 | 237 | 267 |
| **Total** | 3209 | 4443 | 5211 | 5269 | 5725 |

Table 4. Mean summer density of the major groups of macroinvertebrates found at the five sites in Logan Drain, Kintore. Data collected by monthly samples from April 1996 to September 1996.

groups, reduced production from the intolerant taxa was offset by increases from the tolerant.

Upstream, two filter feeding hydropsychid caddisflies, C*eratopsyche slossonae* and *Cheumatopsyche oxa,* contributed 15.02 $g/m^2$ and 2.53 $g/m^2$, respectively, and over 89% of the filter feeder production. These two taxa also produced the most biomass of the filter feeding collectors at the downstream site with 4.65 $g/m^2$ and 1.61 $g/m^2$, respectively. The large drop in their production was not replaced by other taxa in their functional group.

Amongst the shredders, the stonefly *Allocapnia vivipara* had the largest percent drop between sites. It had little impact on total production by this functional group as the cranefly shredder *Tipula ssp* produced 13.45 $g/m^2$ upstream and 12.76 $g/m^2$ downstream and dominated both sites.

The collector-gatherers were the next largest contributors to overall production. Three of the four mayfly species are collector-gatherers. *Baetis brunneicolor* accounted for the slight overall reduction in this functional group but most of the Chironomidae are in this group as well and increased production by *Polypedilum ssp*, *Parametriocnemus lundbeckii*, the *Orthocladius* Group, *Tvetenia paucunca* and *Diamesa nivoriunda* helped maintain similar production levels to those upstream.

Biomass produced by scrapers was similar at both sites. The predators were the only functional group to have greater production downstream.

**4.5 Biotic Indices**

The EPT richness index, BI and FBI are shown in Table 5. EPT shows a reduced mean number of Ephemeroptera, Plecoptera and Trichoptera taxa present downstream on each sample date, from 10.1 to 9.2. The BI increased from upstream to downstream for both the 1995-1996 sample year with taxa identified to species and the 1996 sample year with taxa identified to order and family only. The FBI indicates that the downstream populations live in an environment of greater organic enrichment than Site 1.

| | Biotic Indices | | |
|---|---|---|---|
| Site | EPT richness index | BI (1995-1996) | FBI rapid assessment (1996) |
| 1 | 10.1 | 4.63 | 4.90 |
| 2 | na | na | 5.01 |
| 3 | na | na | 4.90 |
| 4 | na | na | 5.01 |
| 5 | 9.2 | 4.78 | 4.97 |

(na appears when insufficient data is available to calculate the indice)

Table 5. Summary of three biotic indices calculated for the five sites studied in Logan Drain.

## 5.0 Discussion

A substantial shift in the community structure and function was detected by each of the analytical methods employed throughout this study.

### 5.1 Nutrients, Temperature and Herbicides

In small headwater streams, characterized by narrow, shallow channels with a more or less complete canopy, allochthonous inputs of detritus are expected to exceed autochthonous production in the stream itself (Vannote et al. 1980). As stream order increases and shading by riparian vegetation is reduced, primary production by algae and macrophytes begin to dictate the supply of organic material to higher trophic levels. At Kintore, both the upstream and downstream sites are heavily shaded and appear to be light limited. The elevated nitrogen concentrations didn’t lead to any visible increase in standing crop of the algal mats or macrophytes downstream with the exception of a brief bloom of filamentous algae at the mouth of the secondary ditch in the early summer of 1995. In addition, density, biomass and secondary production by scrapers, which rely on attached diatom production, remained constant between sites indicating little change in their food supply and no noticeable impact from the higher nitrogen concentrations.

Water temperature affects the period and rate of growth, reproduction and emergence in aquatic invertebrates. Lifecycles were established for all of the common taxa found at Sites 1 and 2 in order to estimate secondary production. Most of the Diptera have many generations per year, the exact number difficult to determine. The remaining taxa were clearly either univoltine or bivoltine. There was no evidence that any of these animals had altered lifecycles from one site to another: recruitment, emergence periods and the number of generations per year appeared identical. Drain temperatures were similar between these two main sites. If the detailed lifecycle studies had also been completed at Sites 3 and 4, they may have revealed changes due to the significantly higher temperatures recorded there.

**5.2 Sediments**

Shredders convert large organic particles, such as deciduous leaves, to fine particles and feces which are, in turn, consumed by filter feeders or collectors. Clearing of farmland should result in fewer allochthonous inputs, greater irradiance and enhanced autochthonous production. This should reduce the density of shredders, leading to lower concentrations of fine suspended organic particles downstream. The greater production of shredders at our Site 1 is consistent with this prediction. The decline in secondary production downstream was almost entirely due to the impaired success of the dominant filter feeding collectors, notably *Ceratopsyche slossonae* and *Cheumatopsyche oxa.* However, it is unlikely that reduced shredder activity led to the far greater reduction of 11.67 g/m$^2$ in filter feeder production at this site.

Mean individual biomass decreased downstream for all of the major groups of animals studied. Identical weight to head width regressions were used at both sites, indicating that either the animals in each instar were smaller, or that fewer individuals reached the late instars downstream. There were no differences between sites in bodysize of *C. slossonae* animals in the same instar (e.g. Table 6), but differences in density became greater in later instars. Increased mortality and/or drift of the late instars of *C. slossonae* and *C. oxa* appears to have led to the reductions of 19% and 36% in mean individual biomass at Site 5 relative to Site 1. However, the smaller mean size does not fully explain the lower production by *C. slossonae* downstream.

Comparison of the numbers in each instar at each site indicate that recruitment of first instar individuals was 44% lower downstream (Table 6) and this is primarily responsible for the diminished production by *C. slossonae*.

Lower recruitment downstream was not confined to *C. slossonae*; three of the four other filter feeding caddisflies were also much less abundant downstream. Lower recruitment can result from a poor emergence in the last generation, reduced oviposition by females, greater mortality of eggs, and increased mortality in the first instar. The adult stage of *Ceratopsyche ssp*. can live up to 15 days, and the close proximity of the two sites ensure that there will be some mixing between adult populations. The density of previous generations is not likely to be responsible for the poor recruitment at one site versus another.

Female caddisflies return to the water to lay their eggs on solid submerged surfaces. Willis et al. (1992) noted that the eggs of *C. slossonae* were found only on clean, moss-free rock surfaces. The downstream site in the Logan Drain has fewer exposed boulders on which females can crawl into the water from and attach their eggs. The larger percentage of fines eliminates oviposition sites on the smaller cobble and gravel substrates as well.

| Ceratopsyche slossonae | Mean Annual Density ($N/m^2$) | | | Mean Individual Biomass (mg/individual) | | |
|---|---|---|---|---|---|---|
| Instar | Site 1 | Site 5 | % Change | Site 1 | Site 5 | % Change |
| 1 | 958 | 537 | -44 | 0.009 | 0.009 | 0 |
| 2 | 1277 | 605 | -53 | 0.035 | 0.038 | 9 |
| 3 | 1287 | 474 | -63 | 0.148 | 0.157 | 6 |
| 4 | 1019 | 297 | -71 | 0.668 | 0.718 | 7 |
| 5 | 653 | 235 | -64 | 3.072 | 2.829 | -8 |

Table 6. Mean annual density and mean individual biomass of *Ceratopsych slossonae* upstream and downstream of the control site in Logan Drain.

Willis et al. (1992) found egg mortality to be negligible while first instar mortality approached 93% as a result of cannibalism and predation. A slight change in first instar mortality could dramatically impact production by this species. Mortality in later instars was low in their study and in Logan Drain.

It has been suggested that each taxon of net-spinning caddisflies partitions a specific fraction of the drifting stream seston (Wallace et al. 1977), the niche occupied being dictated by the net mesh size, structure and current requirements. Fine meshed nets are constructed in slow currents found on the sides or undersides of rocks while nets with coarse mesh sizes are built in faster water capable of transporting larger particles. Mesh sizes and typical current requirements vary according to the species and instar of the caddisflies. Late instar animals manufacture coarser nets than early instars of the same species and require faster currents. Net mesh sizes range from 1x6 $\mu$m for *Dolophilodes distinctus* to 176x298 $\mu$m for *C. slossonae* (Wallace et al. 1977, Fuller and Mackay 1979) amongst the Trichoptera at our study sites. In order for these insects to occupy the same riffle, a wide range of current velocities is required. In addition to providing a solid surface to which eggs and nets can be anchored, substrate type and heterogeneity dictate the availability and frequency of the feeding microhabitats important to this group of animals.

Lenat (1984) noted that sedimentation was the major problem leading to lower taxa richness and low stability of benthic macroinvertebrates at agricultural sites in North Carolina. Other researchers have found that densities and production of caddisflies increase with substrate particle sizes (Barber and Kevern 1973, Resh 1977, Rabeni and Minshall 1977). This suggests that the more homogenous sediments at Site 5 may account for much of the lower production of Trichoptera relative to Site 1.

In contrast, the density of Chironomidae has frequently been positively associated with substrates which retain fine detrital sediments (Minshall 1977, Rabeni and Minshall 1977). It has been shown that large detrital particles (sticks and pieces of wood > 3.95 mm) are predominant amongst large substrate sizes (cobbles and boulders), and small detrital particles (leaf fragments < 3.95 mm) more common in finer inorganic particles (Rabeni and Minshall 1977). The small detrital particles are available as food to gathering chironomids and contribute to the shift in community structure from Ephemeroptera, Plecoptera and Trichoptera to Chironomidae (Lenat 1984, Lenat and Crawford 1994) at agricultural sites. In addition, many Chironomidae are adapted for burrowing and can tolerate accumulations of fine sediments. The increased density and production by chironomids at Site 5 in Logan Drain supports this view.

Predators were the only group to register increased production downstream. This group was not expected to change across the study site. It is possible that the smaller mean individual biomass for each of the insect orders downstream made the hunting easier for the rather small predators found at both sites.

## 5.3 Biotic Indices

The EPT richness rating indicates the level of stress imposed on intolerant macroinvertebrate taxa. While not statistically significant ($t = 1.876$, $df = 18$, $p = 0.077$), the drop in mean value over the course of the 1995-1996 season from 10.1 to 9.2 suggests that the level of stress on these animals is greater at the downstream site. Other researchers have recorded similar population changes at sites impacted by agriculture (Lenat and Crawford 1994).

The Hilsenhoff biotic index was developed to assess the response of arthropod communities to organic and nutrient pollution, which can lead to lower dissolved oxygen levels. Tolerance values were developed for 359 taxa in Wisconsin streams. The values range from 0 to 10, with 0 being the least tolerant and 10 being the most tolerant to organic pollution. The site BI is the average tolerance value for all of the individuals collected. The mean biotic index values for both sites were between 4.51 and 5.50 and indicated good water quality with some organic pollution. The BI was significantly greater ($t = -2.559$, $df = 15$, $p = 0.022$) at the downstream site. The elevated index was primarily a result of the increased densities of Chironomidae, most of which are moderately tolerant of organic enrichment and have been assigned tolerance values ranging from 5 to 8. Reduced densities of intolerant Ephemeroptera, Plecoptera and Trichoptera taxa also led to the higher site biotic index value downstream. Hilsenhoff also developed a rapid assessment method (Hilsenhoff 1988b) in which taxa need only be identified to the family level. This method was used to evaluate conditions throughout the summer and fall of 1996. Again, the downstream FBI was greater than the upstream site. The much higher chironomid densities was almost solely responsible for elevating the downstream value. The FBI was in the same range as the biotic index for the previous year and indicated good water quality with some organic pollution.

## 6.0 Summary

Macroinvertebrates tolerant of organic enrichment and fine substrates (Diptera: Chironomidae, Tipulidae, Tabanidae, Ceratopogonidae, Simulidae) increased in mean annual density downstream from the agricultural field site. This group maintained mean annual biomass and annual secondary production levels similar to those obtained at the upstream site.

Macroinvertebrates intolerant of organic enrichment and preferring coarser substrates (Ephemeroptera, Plecoptera, Trichoptera) decreased in mean annual density, mean annual biomass and secondary production downstream of the field site.

Secondary production within three of the five functional feeding groups remained constant between sites. Reduced production from intolerant taxa was balanced by gains from tolerant taxa in these three groups. Production by filter-feeding collectors was 59% lower at the downstream site, primarily due to poor recruitment of the dominant hydropsychid caddisfly *Ceratopsyche slossonae*. It appears that the shift in substrate from an heterogeneous mix of fines, gravel, cobble and boulders at the upstream site to predominantly fines and gravel downstream has reduced the availability of suitable oviposition sites for *C. slossonae* and the other filter feeding caddisflies. The downstream substrate may also limit the range of current velocities and net attachment sites available. The 17% increase in production by predators downstream probably reflects the smaller mean individual biomass found in all major groups of animals and the increased density of prey at this site. The fact that other filter feeding taxa did not replace the caddisflies indicates that either the quantity and/or quality of suspended organic nutrients deteriorated downstream or that nutrient cycling is less efficient at this site.

Each of the biotic indices (EPT richness index, Hilsenhoff BI) showed that water quality declined downstream. The Hilsenhoff BI indicated some organic pollution at all of the study sites in Logan Drain but that water quality was good.

**Summary and Conclusions for Kintore Site**

A field-scale water balance calculation has allowed the determination of major transport pathways for leachate from an agricultural field. The dominant hydrological component in the water balance is the tile drainage network. About 60% of the total precipitation for 1995 and 1996 exits the field through the tile drains. A greater proportion of effluent flows from tiles located in the lower part of the field where the water table was consistently higher throughout the year. Indeed, water almost always flowed from tile outlet C11, located in the lowest part of the field (Fig. 6).

The tile drain outlets contributed over 80% of the increase in nitrate loading to a drainage ditch (Logan drain) located at the lower perimeter of the study field. In terms of mass of nitrate per hectare, tile drain outlet C2, located in the upper part of the field on the steepest slope, contributed more than tiles located in the lower part of the field. The greater contribution from upland areas is consistent with higher concentrations of nitrate in shallow groundwater in the upper areas. The concentration of nitrate in all tile outlets increased substantially following plough-down of the alfalfa crop on day 306, 1995 (Fig. 33). As well, the concentrations of nitrate and to a certain extent ammonium, also increased following manure application in the spring of 1996 (Figs. 33 and 36). In general, the concentration of nitrate in tile drain effluent was higher in 1996 under a corn crop than in 1995 under alfalfa.

Groundwater input at the top of the field was higher than groundwater output at the bottom. We believe that excess groundwater was siphoned off by the tile drainage system at the water table in the middle to lower parts of the field. Hence, groundwater which may have percolated past the tile drains in the upper part of the field may have entered the tile drainage system at a lower point in the field.

Groundwater contributed a significant amount of water to the drainage ditch (Cey et al., 1998a). However, the groundwater nitrate load to the ditch on an annual basis was much less than the load from tile outlets. We found good geochemical and isotopic evidence that denitrification was an active process in reducing the nitrate concentration of groundwater in the lower part of the study field and beneath an uncultivated strip between the field and the ditch (Cey et al., 1998b). As well, groundwater contributed proportionally more chloride (a conservative tracer) than nitrate to the ditch which is circumstantial evidence of nitrate attenuation in the groundwater flow system.

Distinctive shifts in the composition of benthic invertebrate communities toward assemblages more characteristic of degraded streams were observed from upstream to downstream in the study area of Logan Drain. Larger, more sensitive forms, especially mayflies, caddisflies and stoneflies were replaced by smaller, more tolerant chironomids. While total numerical abundance of invertebrates were similar at the top and bottom of the study reach, most individual animals were smaller downstream and secondary production was 27% lower. Two summary indices, often used in stream assessment, also indicated slight degradation of water quality downstream.

The upstream-downstream differences also indicated a fundamental shift in the trophic structure of the benthic community. Shredders and filter-feeders were less important downstream. Fine-particle collectors were more numerous downstream, but were of smaller size. Only predatory invertebrates exhibited greater production at the downstream site.

Overall, the nature and magnitude of the changes we observed in the benthic fauna of Logan Drain suggest that there is some eutrophication of the stream within the study reach. However, changes in the physical composition of the stream bed are of greater biological significance. These changes appear to be consistent with the effects of sedimentation of eroded soil and, perhaps, alterations in the hydraulic regime, both of which are likely results of agriculture. Next we give a brief summary of the interconnection between the biological and hydrological aspects of the Kintore study.

As discussed above, the agricultural activities contributed a minor amount of nitrate and even less ammonium to the Logan Drain. The tile drain outlets were the major sources of the nitrogen. Groundwater that discharged through the bottom sediments of the Logan Drain was generally quite low in nitrogen concentration. These results are consistent with the biological data that suggested that eutrophication of the Logan drain waters was very minor. On the other hand, biological effects on the drain were more likely due to an increased sediment load or changes in the drain substrate in the downstream direction. Unfortunately we did not measure suspended sediment loads in water samples taken from the Logan drain or from tile drainage effluent. However, in general, we did observe visually an increase in the amount of suspended sediment in water samples taken from the tile drain outlets during rainfall events by the Isco samplers. Water samples were most cloudy at or near the peak of the event hydrograph. As well, separation of many storm hydrographs into pre-event and event components revealed that a significant proportion of event water was entering the tiles. It is logical to assume that the event water would contain relatively high suspended sediment and nutrient loads. This is consistent with the findings from the biological study of Logan Drain.

# II Woodslee Site

## Part 1 Water Balance

## 1.0 Methodology

### 1.1 Introduction

Experimental plots with Brookston clay loam (fine-loamy, mixed, mesic Typic Argiaquoll) were located at the Eugene F. Whalen Exp. Farm, Agric. and Agri-Food Canada near Woodslee, Ontario (N42°12’37”, W82°44’42”). Each plot of total 16 plots was 15 m wide by 67 m long with <1% slope. Two 104 mm diam. drain pipes were installed at 7.5 m spacings and 0.6 m depth parallel to the length of each plot. Since 1995 the plots have been managed with no-tillage combined with three different water treatments: free tile drainage, D, controlled tile drainage with subirrigation, CD/S, and controlled tile drainage, CD. The plots with CD/S received extra water supplied from drain pipes as subirrigation when needed. Tile drainage was controlled in the CD/S and CD plots by changing heights of drain pipe outlets. Before 1995 several crop-tillage managements including moldboard plow, MP, and soil saver, SS, were practiced combined with intercropping, IP, and regular cropping systems. Only two water management systems, D and CD/S, were used before 1995. Corn (*Zea mays* L.) was grown in the plots except during 1995, when soybeans (*Glycine max* L.) were grown.

### 1.2 Atmosphere

Environmental factors were measured at an automated weather station located about 200 m northwest of the plots. Daily measurements used to estimate evapotranspiration and as input for a numerical model of water flow in soil (Part 2) were mean air temperature, total shortwave irradiance, and hourly precipitation.

**1.3 Unsaturated Zone**

To monitor temporal changes in soil water content, time domain reflectometry (TDR) electrodes (2 mm diam., 0.2 m and 0.4 m long) connected to 200S-shielded parallel cables were vertically inserted into the soil surface. Each length of TDR probe was duplicated in each of 8 plots out of the 16 total plots. Dielectric constant of the soil was determined with a metallic cable tester, the model 1502C (Tektronix, Beaverton, OR), via an impedance matching transformer and was converted to volumetric water content using the calibration curve proposed by Topp et al. (1980).

**1.4 Saturated Zone**

Multi-level piezometers with eight different screen depths of 0.8, 1.2, 1.8, 2.5, 3.1, 3.8, 4.4, and 5.0 m from the soil surface were installed in eight plots. One multi-level was located near a tile drain whereas the other was located between tiles in each plot. One time each month the hydraulic heads were measured at each level, and water samples were collected for $NH_4^+$, $NO_3^-$, and $Cl^-$ analyses. The electrical conductivity (model 30, YSI Inc., Yellow Spring, OH, U.S.A.), dissolved oxygen concentration (model 55, YSI Inc., Yellow Spring, OH, U.S.A.), Eh, and pH (Oakton pH/mV/C meter) of groundwater from each level were measured in-situ on a seasonal basis during the experiment. The field-saturated hydraulic conductivity of the subsoil was measured at the 4.4 and 5.0 m screen depths using the Hvorslev method (Fetter, 1994) in the summer of 1995.

Fertilizer grade granular KCl (18.5g $Cl^{-1}$ $m^{-2}$) was applied on October 28, 1993 and November 9, 1995 to the soil surface of each plot as a non-reactive tracer to monitor water flow in soil. Transport parameters for chloride will be estimated by fitting an analytical solution for a one-dimensional convective dispersion equation to concentrations measured at each screen depth with time. The analytical solution for the flux concentration with a delta spike input (Jury and Roth, 1990) is

$$C_f = \frac{C_T}{R}\left[\frac{L}{2\sqrt{\frac{\pi D_s t^3}{R}}}\right]\exp\left[\frac{-\left(L-\frac{Vt}{R}\right)^2}{\frac{4D_s t}{R}}\right]\exp(-kt) \quad \textbf{(4)}$$

where $C_f$ is the flux concentration, $C_T$ is a total mass of solute applied, R is the retardation factor (assumed equal to one), L is the sampling depth, $D_s$ is the dispersion coefficient, k is the decay constant (assumed equal to zero), and t is time.

**1.5 Surface Water Runoff and Tile Drains**

Temporal changes in quantities of tile drain and surface runoff water from each plot were automatically monitored. Part of tile drain and surface runoff water was automatically collected for chemical analyses. Details of the field layout and instrumentation were found in Drury et al. (1993).

## 2.0 RESULTS AND DISCUSSION

### 2.1 Evapotranspiration

Evapotranspiration, ET, was estimated using the decrease in TDR-measured soil water content during the summer and compared with evapotranspiration ($ET_{es}$) calculated based on measured environmental factors proposed by Tan and Fulton (1980, 1981). Soil water content measured with TDR for each depth was averaged over duplicate measurements in each of the 8 plots. Deviation of averaged TDR-measured water content such as standard error (S.E.) was very small, S.E. was between 0.003 and 0.016$m^3$ $m^{-3}$ for L=0.4m probes and between 0.003 and 0.024$m^3$ $m^{-3}$ for L=0.2m probes. When the soil surface was bare and the crop was immature, estimated $ET_{es}$ agreed well with TDR measurements(Fig.1). However, after Day 190 TDR measurements of water loss from the soil were much less than the predicted $Et_{es}$. Perhaps the crop roots extracted water from depths below the TDR sample volume after Day 190. We will assume that the $ET_{es}$ based on environmental factors reasonably represent actual ET in the field.

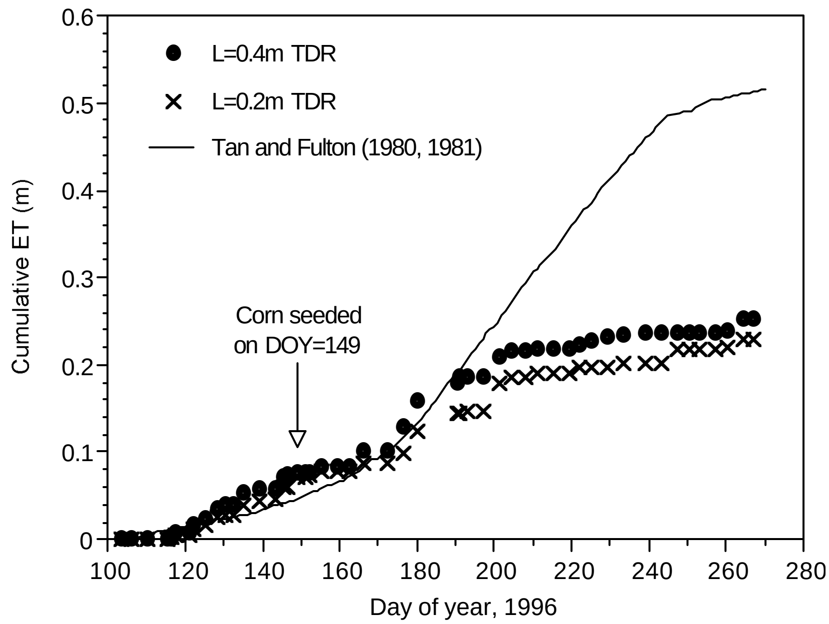



Figure 1. Comparison of cumulative evapo-transpiration between TDR-estimates and calculated by the procedure of Tan and Fulton (1980, 1981).

**2.2 Tile Flow and Surface Runoff**

The amounts of annual tile drainage and surface water runoff relative to annual precipitation were summarized in Table 1 for each water management practice. Data from the 8 plots were averaged over repetitions of treatments: triplicates of D and CD, duplicates of CD/S after 1995, and quadruplicates of D and CD/S before 1994. Each year, surface drainage of the D treatment was larger than the CD/S treatment. Tile drainage showed the opposite trend. This was not surprising since soil water content of CD/S might be larger than that of D and less water from the precipitation was sorbed into the wetter soil.

| Year | Precipitation (m) | Treatment | | | | | |
|---|---|---|---|---|---|---|---|
| | | D | | CD/S | | CD | |
| | | Surface | Tile | Surface | Tile | Surface | Tile |
| 1992 | 0.935 | 0.10 | 0.39 | 0.20 | 0.26 | - | - |
| 1993 | 0.688 | 0.08 | 0.29 | 0.15 | 0.25 | - | - |
| 1994 | 0.703 | 0.08 | 0.18 | 0.09 | 0.16 | - | - |
| 1995 | 0.608 | 0.05 | 0.32 | 0.09 | 0.30 | 0.07 | 0.27 |
| 1996 | 0.640 | 0.19 | 0.33 | 0.30 | 0.20 | 0.22 | 0.24 |

Note: D, CD/S, and CD indicate free drainage, controlled drainage with subirrigation, and controlled drainage, respectively.

Table 1. Partitioning water flow into surface and tile drainage. Amounts were expressed as relative values to annual precipitation.

Tile drainage from CD/S was smaller than that of D since drainage was controlled by risers connected to tile outlets about 0.2m higher than a regular tile outlet. The amount of tile drainage and surface runoff from CD plots was between those of D and CD/S. About 30 to 50% of annual precipitation was drained from the field into channel and river systems nearby via tile drains.

### 2.3 Groundwater flow

An annual water balance was calculated to estimate deep percolation (Table 2). Evapotranspiration, $ET_{es}$, was estimated from Eqs. (4), (5), and (13). It was assumed that the $Et_{es}$ of both soybeans and corn were estimated properly with Eq. (13).

| Year | Precipitation (m) | $Et_{es}$ (m) | Deep percolation with treatments (m) | | |
|---|---|---|---|---|---|
| | | | D | CD/S | CD |
| 1992 | 0.935 | 0.344 | 0.134 | 0.164 (0.006) | - |
| 1993 | 0.688 | 0.369 | 0.063 | 0.170 (0.124) | - |
| 1994 | 0.703 | 0.372 | 0.156 | 0.250 (0.096) | - |
| 1995 | 0.608 | 0.611 | -0.225 | -0.133 (0.103) | -0.214 |
| 1996 | 0.640 | 0.251 | 0.061 | 0.164 (0.096) | 0.099 |

Note: D, CD/S, and CD indicate free drainage, controlled drainage with subirrigation, and controlled drainage, respectively. Negative values mean water flow into the system across all the boundaries but the surface.

Table 2. Annual deep percolation (m) estimated with a water balance equation. $Et_{es}$ was estimated using measured weather data. Estimates for CD/S included an amount of water supplied as subirrigation, which was indicated between parentheses.

Annual changes in soil water content were assumed to be negligible. In years 1992, 1993, 1994, and 1996 the maximum $ET_{es}$ was 2-5 mm $d^{-1}$ and occurred between Days 175 and 225, whereas in 1995 the maximum $ET_{es}$ exceeded 7 mm $d^{-1}$ during that time period. About 8 to 20% of annual precipitation might contribute to deep percolation in the years of 1992, 1993, 1994, and 1996 in the D treatment. Smaller precipitation and larger $ET_{es}$ occurred in 1995 compared with other years which resulted in no deep percolation in the D and CD treatments. In the CD/S treatment all subirrigated water was assumed to contribute to deep percolation; however, detailed partitioning of subirrigated water into deep percolation and ET needs further investigation.

Monthly changes in precipitation, $ET_{es}$, and tile and surface drainage (Fig. 2) gives a general idea when water was supplied into and withheld from soil. In Jun., Jul., and Aug., $ET_{es}$ started exceeding precipitation, resulting in no surface runoff or tile drainage. During these months soil water might be used for ET. Following this period, soil was replenished with precipitation, which exceeded $ET_{es}$ demand, in Sep. and Oct. until tile and surface drainage started in Nov. Precipitation was counted as excess water after replenishing soil water and was mostly drained out during the months between December and May. Deep percolation probably occurs mainly during those months.

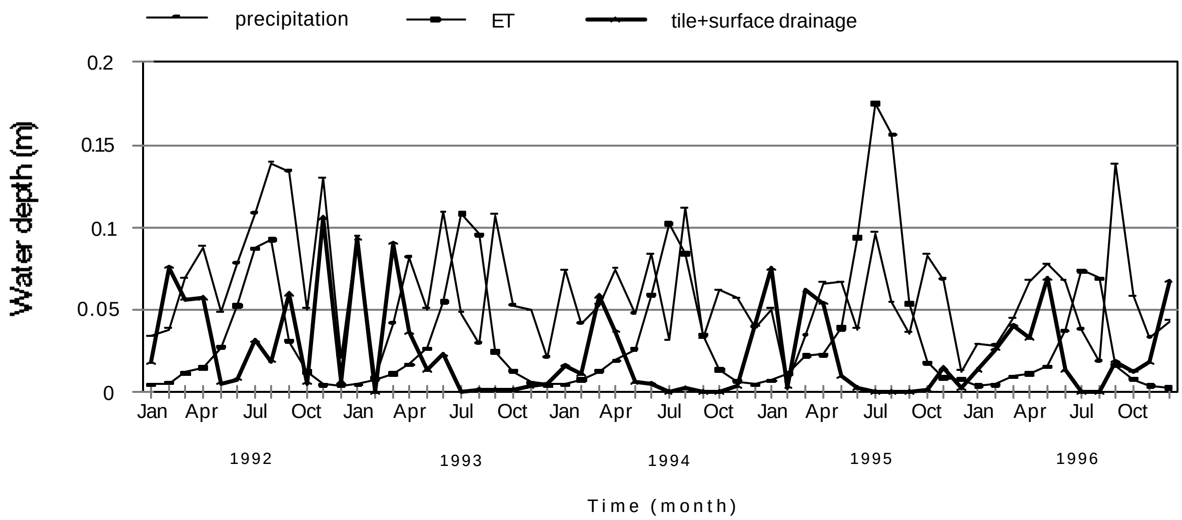



Figure 2 Temporal changes in monthly summed precipitation, evapo-transpiration, $ET_{es}$, and tile and surface drainage.

| Treatment | Vertical flow across the bottom (m) | | |
|---|---|---|---|
| | Near tile | Between tiles | Net |
| D | 0.00467 | -0.00727 | -0.00260 |
| CD/S | 0.00246 | -0.00225 | 0.00021 |
| CD | -0.00816 | 0.00077 | -0.00739 |

Note: Negative values mean that inflow to the system occurs across the bottom.

Table 3. Calculated vertical flow across the bottom of the system for different water treatments between Sep. 1995 and Dec. 1996. Two or more replicates for the treatments were averaged.

Vertical water flow across the arbitrary bottom boundary of the groundwater flow system was estimated using the Darcy's equation with measured hydraulic gradients between z=-4.4 and -5.0m and saturated hydraulic conductivity, $K_s=9.7x10^{-10}$m $s^{-1}$. Table 3 shows the average total water flow across the bottom between September 22, 1995 and December 9, 1996, when hydraulic heads were measured most frequently. Both the small $K_s$ and the small hydraulic gradients (which occurred year-round), resulted in very small estimates of groundwater flux across the bottom boundary. The CD/S treatment had a net downward flow, whereas the D and CD treatments had net upward flow. A comparison between deep percolation estimated with the water balance method (Table 2) and with Darcy's equation (Table 3), suggests that three-dimensional groundwater flow must be considered in the Darcy method. Huan (1994), however, reported that regional groundwater flow is horizontal and eastward towards Bell River with a hydraulic gradient of <0.002m

Temporal fluctuation of water pressure head at z=-4.4 m deep below the soil surface had a linear relationship with the hydraulic gradient (Fig. 3). The relationship was significantly correlated with a <1% level (y=-0.9525+0.2611x, r=0.38 where y and x represent the hydraulic gradient (m $m^{-1}$) and the water pressure head (m), respectively). When the water pressure head is larger than 3.65m, which usually occurred in spring, direction of water flow will be

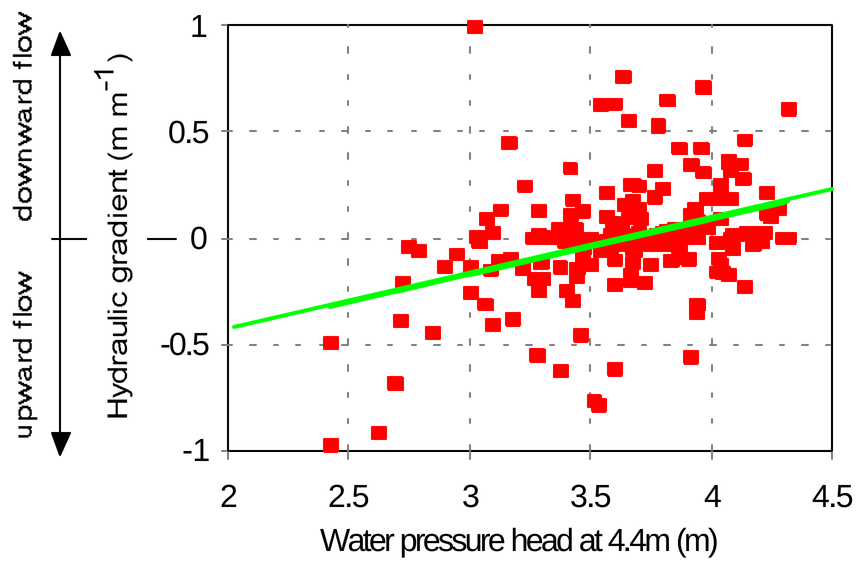



Figure 3. Relationship between the hydraulic gradient and water pressure head at z=-4.4m. All the data collected from the 16 sets of piezometers were used.

upward when the water pressure head was smaller than 3.65m, often downward. On the other hand, the direction of water flow will be observed in summer. According to the observation (Fig. 3) and Huan (1994), the $K_s$ value near the bottom boundary should be larger so that vertical deep percolation would be as large as that estimated with the water balance.

**3.0 Summary**

The CD/S and CD treatments increased surface runoff and decreased tile drainage by 10% compared with the D treatment. Net vertical water flow across the bottom boundary of the arbitrary system was outflow from the system for the CD/S treatment, although the amount was very small compared with other water removal by surface runoff and tile drainage. The results of the water balance calculation implied that more water should be removed from the system by horizontal and/or vertical deep groundwater flow. The underestimate in deep groundwater flow was probably due to an underestimate in measured $K_s$.

**Part 2. Water Flow Model**

**1.0 Model description**

**1.0.1 Theory**

Two-dimensional liquid phase water flow in isotropic saturated-unsaturated soil was expressed as (Clements et al, 1994)where $\delta$ is

$$\delta C_w \frac{\theta}{\phi}\frac{\partial \psi}{\partial t} + \frac{\partial \theta}{\partial t} = -\frac{\partial}{\partial y}\left(-K\frac{\partial \psi}{\partial y}\right) - \frac{\partial}{\partial z}\left(-K\frac{\partial \psi}{\partial z} - K\right) \quad \textbf{(5)}$$

0 for $\psi<0$ and 1 for $\psi \geq 0$, $C_w$ the specific storage ($m^{-1}$), $\theta$ the volumetric water content ($m^3\ m^{-3}$), $\phi$ the soil porosity ($m^3\ m^{-3}$), t time (s), K the hydraulic conductivity ($m\ s^{-1}$), $\psi$ water pressure head (m), and z and y are in the Cartesian coordinate system (z the vertical and oriented positive upward; and y the horizontal and oriented positive rightward). Equation (2) was numerically solved using the Galerkin finite element method (Noborio et al., 1996). A schematic of a two-dimensional soil profile was separated into a finite number of triangular elements. It was assumed that coefficients and fluxes were constant within an element, but they could vary from one element to another (Istok, 1989). Applying the Galerkin weighted residual technique and integration by parts, the final form of Eq. (2) was expressed as the first–order differential equation (Wang and Anderson,1982)

$$[conductance]\{R\} \% [capacitance]\left\{* C_w \frac{2}{N} \frac{MR}{Mt} \% \frac{M2}{Mt}\right\} \; ' \; \{residuals\} \qquad \textbf{(6)}$$

where [ ] represented a matrix; { } represented a vector; {ψ} was the vector of unknown water pressure head for all nodes; and $\left\{* C_w \frac{2}{N} \frac{MR}{Mt} \% \frac{M2}{Mt}\right\}$ was the vector of differentiated water pressure head and water content for all nodes with respect to time t. The lumped–mass procedure proposed by Neuman (1973) was applied to the capacitance term in Eq. (3) to reduce numerical oscillations following suggestions of Pickens et al. (1979) and Celia et al. (1990). Following Celia et al. (1990), a combination of the modified Picard iteration based on an incremental solution procedure and the lumped–mass procedure was used to solve a mixed form of a water flow equation involving both water content and water pressure head as in Eq. (2). A time step of the calculation was automatically adjusted based on the number of iterations and the predetermined criteria (da Silva, 1990).

**1.0.2 Boundary Conditions**

At the soil surface nodes, evaporation and rainfall rates provided the Neuman boundary condition during a non-crop growing season. During a crop growing season, an evapotranspiration rate was estimated from daily environmental data measured near the experimental site and estimated crop growth rate.

An estimated daily evapotranspiration rate $ET_{es}$ (m $d^{-1}$) was calculated as:

$$ET_{es} = \alpha\kappa ET_{eq} \qquad \textbf{(7)}$$

where $\alpha$ is a reduction factor depending on average water pressure head of the root zone; and $\kappa$ is a crop coefficient empirically determined for a reference such as Class A pan or the equivalent evapotranspiration. Evaporation from the soil surface during the non-cropping season was assumed to be equal to $ET_{es}$ with the minimum $\kappa$=0.26 and $\alpha$=1.0. The equivalent evapotranspiration, $ET_{eq}$, in Southwestern Ontario was expressed as (Tan and Fulton, 1981)

$$ET_{eq} = \left(0.48 + 0.01 T_a\right)\left(0.56 R_s - 0.039\right) 10^{-3} \qquad \textbf{(8)}$$

where $T_a$ is the daily mean air temperature ($^{o}$C); and $R_s$ cumulative daily shortwave irradiance (m $d^{-1}$). To estimate values of $ET_{es}$ in m $s^{-1}$ for a surface boundary flux, daily $ET_{es}$ values were assumed to distribute over the calculated day length according to a sine

$$ET_{es} = \frac{\alpha\kappa ET_{es}}{3600.0} \frac{1.5707963}{DL} \sin\left(\left[DAYHR - 12.0 + \frac{DL}{2}\right]\frac{\pi}{DL}\right) \qquad \textbf{(9)}$$

function (Van Bavel and Lascano, 1987) as

where DL is the potential day length (h) calculated after Campbell (1985); and DAYHR is time of the day in a fraction of hour. Hourly precipitation was assumed to uniformly distribute over an hour and change in a piece wise fashion for the next hour.

In non-freezing seasons, a precipitation rate was used as a Neuman boundary flux as long as it was smaller than the maximum allowable water flux across the surface boundary. The maximum water fluxes, $Jmax_i$ (m $s^{-1}$), at a surface boundary node was calculated by

where variables with superscript t+$\Delta$t indicate newly calculated

$$Jmax_i = [conductance]\begin{Bmatrix} \vdots \\ \psi_i^{t+\Delta t} \\ \vdots \end{Bmatrix} + [capacitance]\begin{Bmatrix} \vdots \\ -C_{w_i}^{t+\Delta t}\frac{\theta_i^{t+\Delta t}}{\rho}\left(\frac{\psi_i^{t+\Delta t}-\psi_i^{t}}{\Delta t}\right)+\left(\frac{\theta_i^{t+\Delta t}-\theta_i^{t}}{\Delta t}\right) \\ \vdots \end{Bmatrix} - \begin{Bmatrix} \vdots \\ residual \\ \vdots \end{Bmatrix} \quad \textbf{(10)}$$

values with a time step $\Delta$t as if a surface node i was immediately saturated with water ($\theta_i^{t+\Delta t}=\theta_s$) and ponded with 0.1m of water depth ($\psi_i^{t+\Delta t}$=0.1m). At each surface node the calculated maximum water flux was compared with a precipitation rate, and a difference between the maximum flux and the precipitation rate was considered to be surface runoff.

At the bottom of the soil profile in the calculation domain, the Neuman boundary condition was applied. According to the observation of temporal changes in water pressure head at z=-4.4 and -5.0 m from the soil surface, the hydraulic gradient between these depths was linearly related to the water pressure head at z=-4.4 m. The water flux at the bottom of the domain, $J_b$ (m $s^{-1}$), was then expressed by

$$J_b = -K_s(-0.9525 + 0.2611\psi) \qquad \textbf{(11)}$$

where $K_s$ is the saturated hydraulic conductivity (m $s^{-1}$) of subsoil.

The Dirichlet boundary condition was applied at a tile drain represented by a single node of the finite elements. In both seasons, a tile drain was treated as the Dirichlet boundary with $\theta=\theta_s$ and $\psi$=0.0m. Other boundaries were treated as no-flow boundaries. Hydraulic conductivity of surrounding elements and their configuration were adjusted following Fipps et al. (1986) based on an electric resistance network model proposed by Vimoke et al. (1963).

**1.0.3 Model Parameters**

**1.0.3.1 Soil hydraulic parameters**

A soil water characteristic curve was expressed with the van Genuchten's equation (1980)

$$\Theta = \left[1 + (\alpha\psi)^n\right]^{-m} \qquad \textbf{(12)}$$

where $\Theta$ is the effective water content expressed by $(\theta - \theta_r)/(\theta_s - \theta_r)$; $\theta_r$ is the residual water content ($m^3$ $m^{-3}$); $\theta_s$ is the saturated water content ($m^3$ $m^{-3}$); n is a parameter, and m=1-1/n. Hydraulic conductivity was expressed by the Mualem-van Genuchten’s equation (van Genuchten, 1980) as

$$K = K_s \Theta^{0.5}\left[1 - \left(1 - \Theta^{1/m}\right)^m\right]^2 \qquad \textbf{(13)}$$

Around a tile drain, $K_s$ was reevaluated following Vimoke et al. (1963) and expressed (Fipps et al., 1986; Rogers and Fouss, 1989) as

$$K_{sd} = \frac{K_s}{R_d/2} \qquad \textbf{(14)}$$

where $K_{sd}$ is the saturated hydraulic conductivity around the tile drain (m $s^{-1}$); and $R_d$ is a resistance adjustment factor estimated by an electric resistance network. The resistance adjustment factor was estimated (Vimoke et al., 1963) as

$$R_d = \frac{8Z_o}{Z_o'} \qquad \textbf{(15)}$$

where $Z_o$’is the characteristic impedance of free space (– 376.7Ω) and $Z_o$ is the characteristic impedance of a type of transmission line that they used. Vimoke et al. (1963) defined $Z_o$ as

$$Z_o = 138\log(D) + 6.48 - 2.34A - 0.48B - 0.12C$$

$$\text{where } D = \frac{s}{r}$$

$$A = \frac{1 + 0.405D^{-4}}{1 - 0.405D^{-4}} \qquad \textbf{(16)}$$

$$B = \frac{1 + 0.163D^{-8}}{1 - 0.163D^{-8}}$$

$$C = \frac{1 + 0.067D^{-12}}{1 - 0.067D^{-12}}$$

in which s is the square mesh size surrounding a drain pipe node and r is an effective radius of the drain pipe. For a 0.1m dia. drain pipe, $R_d$ =1.876 with s=0.021m (Fipps et al., 1986) and r=0.0051m (Prasher et al., 1995).

**1.0.3.2 Crop parameters**

Parameters depending on the crop grown in the experimental plots were determined for corn since it was the typical crop for the plots. The crop coefficient, $\kappa$, of corn for the Class A pan was determined by Tan and Fulton (1980) and modified for $ET_{eq}$ as

$$\kappa = \begin{cases} 0.26 & 0 \le d \le 17 \\ 0.014(d - 17) + 0.26 & 17 < d \le 70 \\ 1.01 & 70 < d \le 105 \\ -0.028(d - 105) + 1.01 & 105 < d \le 131 \\ 0.26 & 131 < d \end{cases} \qquad \textbf{(17)}$$

where d is the day after seeding (d). Since Eq. (14) is valid only when the crop is not suffering from any drought stresses, the reduction factor $\alpha$ was introduced for a drought situation. Values of $\alpha$ for corn were originally determined against water content in the root zone of Fox sandy loam (Tan and Fulton, 1981). The relationship between $\alpha$ and soil water status was generalized by expressing $\alpha$ against water pressure head $\psi$, in which conversion was made to soil moisture characteristic data by Fulton (1970a), as

$$\alpha = \begin{cases} 1.0 & \psi \geq -39.8\,\mathrm{m} \\ 1.0 + 0.003792(\psi + 39.8) & \psi < -39.8\,\mathrm{m} \end{cases} \quad \textbf{(18)}$$

Temporal changes in rooting depth, $z_r$ (m), were approximated (Borg and Grimes, 1986) by

$$z_r = z_{r,\max}\left[0.5 + 0.5\sin\left(3.03\frac{d}{d_{tm}} - 1.47\right)\right] \quad \textbf{(19)}$$

where $z_{r,max}$ is the maximum rooting depth (m), and $d_{tm}$ is the number of days to maturity. Water uptake by roots was assumed to be uniformly distributed in the root zone. It was also assumed that values $z_r$ were constant during a calculating day and changed at midnight, which was the beginning of a new calculation day.

## 2.0 MATERIALS AND METHODS

A computer program for the two-dimensional finite element model of water flow in saturated-unsaturated soil was modified from the original program written in Fortran 77 for two-dimensional water, heat, and solute transport (Noborio, 1995), whose finite element solver was based on work of Wang and Anderson (1982). A two-dimensional soil profile of one-fourth of an experimental plot perpendicular to a tile line was divided into a number of triangular elements (Fig. 4). The total numbers of nodes and elements used for calculation were 450 and 258, respectively.

The soil parameters of the Brookston clay loam were estimated using soil textural data following Campbell (1985) and field/laboratory measurements (C.S. Tan, 1995, personal communication). Estimates of $\alpha$ and n values in Eqs. (7) and (8) were 0.5 and 1.195, respectively, and assumed to be constant in the entire calculation domain. Different values of $\theta_s$ and $K_s$ were assigned to soil layers of z=0-3.5m and 3.5-5.0m. Values of $\theta_s$ for the layers of z=0-3.5m and 3.5-5.0m were 0.52 and 0.43$m^3$ $m^{-3}$, respectively. An estimate of $K_s$ for z=0-3.5m, determined with an auger hole method was 8.56x$10^{-7}$, and one for z=-3.5-5.0m was 8.7x$10^{-10}$m $s^{-1}$, determined with a slug test, which was similar to the value reported by Huang (1994). The model used 1.2 times larger $K_s$ values to take account for reduction of the $K_s$ estimates by smearing auger holes. The model was run for a period when the soil was unfrozen on a Pentium 100 Mhz personal computer and/or an HP workstation using measured environmental data from 1994.

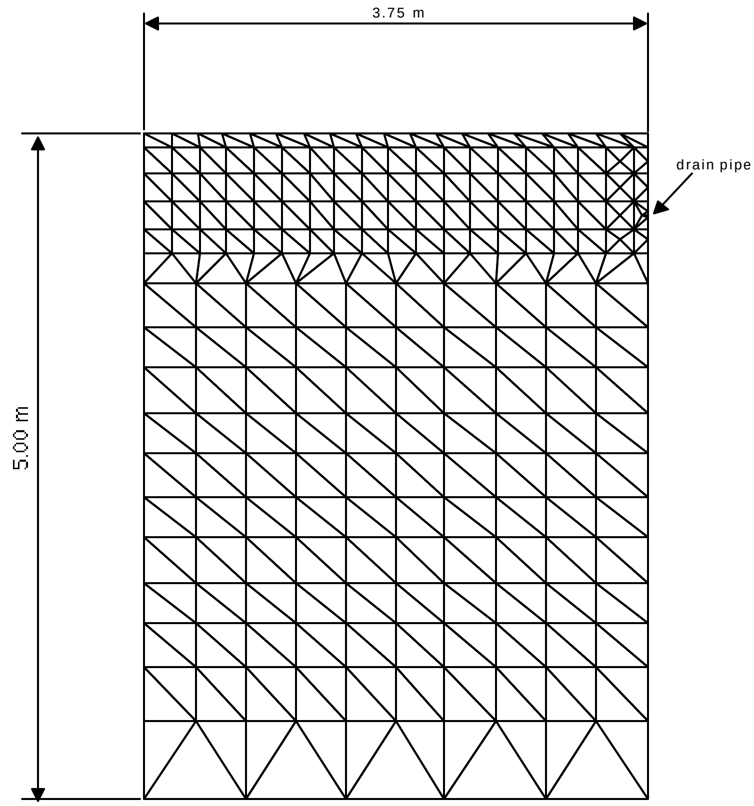



Figure 4. Finite element mesh used for calculation.

## 3.0 RESULTS AND DISCUSSION

Calculated cumulative tile drainage agreed well with measured values (Fig. 5A) although calculation underestimated measurements in the first 50d from the start of simulation. In this period soil was unfrozen in the field; however, the model did not account for excess water caused by thawing. Large discrepancies were found in the cumulative surface runoff (Fig. 5B). If water removed from the system based on the measured water balance (Table 2) was incorporated into the model, the simulated surface runoff may be significantly reduced. Moreover, the model did not consider water infiltration through cracks in the soil which formed in dry seasons.

A simple two-dimensional model was not sufficient to simulate the complex phenomena which occurred in the field, especially in structured, shrink-swell clay soils under a frozen environment. A field-scale study may not be large enough to incorporate the effects of regional groundwater flow on local water balance calculations.

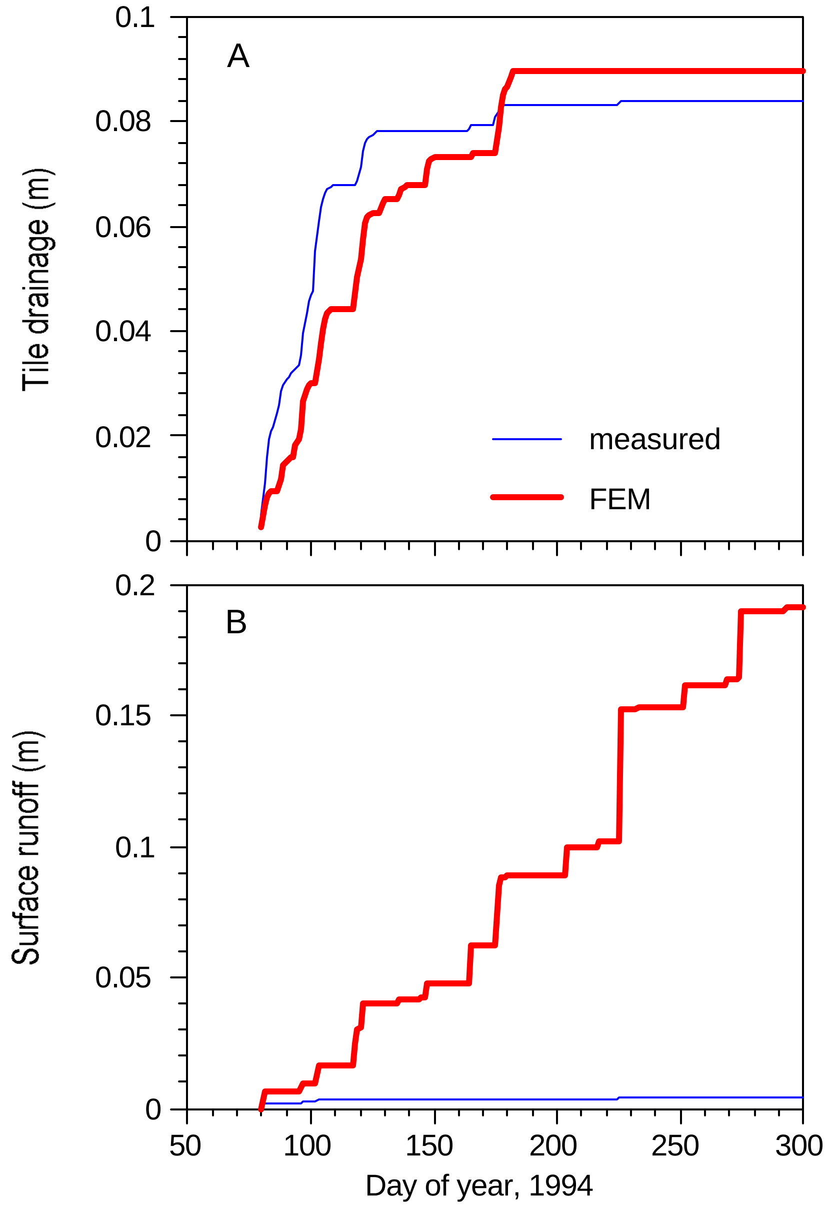



Figure 5. Comparisons of temporal changes in cumulative tile drainage (A) and surface runoff (B) between measured and simulated with the FEM model.

**Summary and Conclusions for Woodslee Site**

Field-scale evapotranspiration was reasonably well estimated with the empirical equation using measured environmental data, and comparable with measurements based on temporal changes in TDR-measured soil water content. Using measured surface runoff, measured tile drainage, and estimates of evapotranspiration, deep percolation was estimated. The water balance implied that more water should be removed from the system as deep percolation than the amount calculated using the Darcy’s equation with measured vertical hydraulic gradients and $K_s$. About 30-50% of annual precipitation was drained from the field, and about 8-20% of annual precipitation might contribute to deep percolation during winter.

Two-dimensional finite element model was developed to simulate water movement in soil. Calculated and measured cumulative tile drainage agreed well each other in a 250d simulation period. Since the model was designed to account for temporal and spatial changes in infiltration during rain, surface runoff was anticipated to be reasonably well calculated. Overestimates of surface runoff might have resulted from improper magnitude of deep percolation both in vertical and horizontal directions and/or preferential flow through soil cracks formed in dry seasons. In both scenarios, hydraulic conductivity measurements of shallow and deep soils were too low.

**Comparison of Kintore and Woodslee Sites**

There were significant differences between the topography and degree of geological heterogeneity of the Kintore and Woodslee sites. The Kintore site was more topographically complex with an overall decline in elevation of about 9 m from the northern to southern edges of the field, whereas the Woodslee site was essentially flat. In addition to the topographic differences, the Kintore site was also more geologically heterogeneous than the Woodslee site which was dominantly clay. These differences led to significant differences in the hydrology between the two sites as discussed below.

As expected, runoff was more significant at the Kintore site due to the increased slope and especially under the corn crop grown during 1996 due to a lack of ground cover. As well, there were significant differences in the groundwater flow patterns between the two sites. In the lower areas of the Kintore site, artesian conditions resulted in tile drain flow almost year-round. In contrast, natural tile drainage at Woodslee was restricted to the spring and winter periods when the water table was relatively high and during some major summer storms.

The main hydrologic similarities between the Kintore and Woodslee sites were the rates of evapotranspiration and the dominant role of the tile drains. At both sites (in particular at the Kintore site), the tile drainage network short-circuited infiltrating waters and reduced groundwater recharge. As well, the tile drains often diverted waters relatively rich in nitrogen directly into the surface water drains. It was especially evident at the Kintore site that water flowing through the groundwater system beneath the riparian zone and discharging to the surface drain contained much less nitrogen than the tile drain effluent.

There were numerous difficulties encountered at both sites in measuring the partitioning of water within the farm fields. The degree of geologic heterogeneity and the three-dimensional nature of the groundwater flow system at Kintore required the installation many monitoring wells (200 monitoring points). Therefore, the wells could only be sampled on a monthly basis for nitrogen and chloride analyses and measurement of hydraulic head values.

Event-related sampling and monitoring was not possible at the Kintore site due to the large number of wells. We now speculate based on the rapid response times of the tile drains that infiltration and recharge to the groundwater table occurs very rapidly during major storms and snowmelt events. It would have been beneficial to install a number of pressure transducers and automatic sampling devices in the wells to record the level and sample the shallow groundwater during these rapid recharge events. This should be the subject of future research.

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# Appendix A.

## EFFICIENCY AND ACCURACY OF DIFFERENT METHODS TO MEASURE POTENTIAL EVAPOTRANSPIRATION UNDER TEMPERATE CLIMATIC CONDITIONS

RICARDO HIRATA, MARY ANNE HARDY, DAVID RUDOLPH, AND GARY PARKIN

DEPT. OF EARTH SCIENCE, UNIVERSITY OF WATERLOO, WATERLOO, ONTARIO.

## Abstract

A study conducted in southern Ontario (Canada) analyzed nine different methods to evaluate potential evapotranpiration (E*tp*) based on techniques that are more efficient for temperate climatic conditions. They included *combination methods* (Penman-Monteith, Penman-Jensen, Penman-Wright-Jensen); *radiation methods* (Turc, Priestley-Taylor, FAO24, Hargreaves); and *temperature methods* (Blaney-Criddle).

The methods were tested in two different approaches. The first compared results of potential evapotranspiration (E*tp*) from nine techniques applied on an alfalfa crop, during the year of 1995. The second compared the same analytical methods to actual evapotranspiration (E*ta*) measured during the summer of 1996, using TDR (time domain reflectometry).

Based on daily averages, the Penman-Wright-Jensen, Penman-Wright and Priestley-Taylor methods compared most favorably to Penman-Monteith, which was considered the standard by which all analytical methods were judged. On a monthly base, the best techniques were Penman-Jensen, Turc and Priestley-Taylor. In both cases, the Hargreaves, Blaney-Criddle and FAO24 approaches gave the poorest comparison with Penman-Monteith. Comparing the results of measured actual evapotanspiration, the best methods were Penman-Monteith, Turc and Priestley-Taylor. The worst correlation was obtained by FAO24, Hargreaves and Blaney-Criddle.

The Penman-Monteith was the most trustworthy method for the temperate climatic conditions. If a limited amount of micro-climatic information is available, the Turc and Priestley-Taylor techniques should be considered. The Turc method was accurate enough to estimate E*tp* on a monthly base if only temperature and solar radiation are available. When the met-station has a hygrometer to record air humidity, the Priestley-Taylor technique is an option.

## INTRODUCTION

Evapotranspiration is one of the most difficult parameters to measure in a water balance study, not only because of meteorological data limitation, but because of the number of different analytical methods. Automatic met-stations and historical data sets, that permit a good calibration of evapotranspiration equations, are rare. As well, an extensive number of calibration parameters have been introduced to adapt methods to a specific site or condition. Jensen ***et al.*** (1990) describe 18 different approaches, but it is possible to add at least a dozen more new equations to their list. Any attempt to estimate evapotranspiration should include an analysis of a group of different methods and establish the best technique, based on availability of micro-climatological data. In many cases, the research is developed in a short time period and it is difficult to introduce corrections. Selection of the appropriate technique is important since methods have presented differences of potential evapotranspiration (E*tp*) of greater than 90% when applied in the same area and with the same climatic conditions.

The main objective of this paper was to compare eleven different analytical techniques to estimate evapotranspiration in southwestern Ontario (Canada). The field site was located in Zorra Township, 2km east of Kintore (N43°10', W81°00' and 340 m asl), approximately 40 km east of London. Et*p* methods chosen were: ***combination approaches***: Penman (1963) with original wind function (Jensen 1974); Kimberly-Penman (Wright 1982); Penman (1963) with Wright & Jensen (1972) wind function; and Penman-Monteith (Monteith 1965); ***radiation approaches***: Priestley & Taylor (1972); Turc (1961); FAO24 (Doorenbos & Pruitt 1977); Hargreaves *et al*. (1985);and ***temperature approach***: Blaney-Criddle FAO24 (Doorenbos & Pruitt 1977). In order to get an actual evaluation of each method, all formulations were used without adjustment or calibration, abiding by the original expressions proposed by the authors.

The accuracy of each method was evaluated by comparison to the Penman-Monteith technique and to actual evapotranspiration measurements (E*ta*). The E*ta* was obtained with TDR (time domain reflectometry) measurements of the decrease in surface soil-water content. Climatological data were collected through 1995 until October 1996. However, E*ta* was only measured from July to September 1996.

In order to define which technique offers the most accurate results for a minimum amount of climatological information, the number of micro-climatological parameters and the recommended minimum time periods (daily, weekly and monthly) were analyzed for the various methods. Another important aspect considered was the definition of which methods are more reliable and more difficult to apply in areas where calibration is not possible.

**POTENTIAL EVAPOTRANSPIRATION METHODS**

Nine different evapotranspiration methods were applied at the Kintore site. The evaluation of the different techniques was done on a daily, monthly and annual basis. Considering the type of data acquisition, the methods can be classified into three categories: ***temperature, radiation,*** and ***combination***. Table 1 gives the information normally required for each method. The equations used here were summarized by Jensen ***et al.*** (1990).

Temperature Based Approaches are used most frequently, because of their simplicity and low data requirements. Basically, the ***temperature*** approaches use air temperature (average, minimum and maximum), day length and sometimes air relative humidity and site latitude. According to Tanner (1967) many linear temperature methods have the following general form:

$$\boxed{E_{tr} = C_1 L_d T (C_2 - C_3 h)}$$ **Equation 1**

where E*tr* is alfalfa reference evapotranspiration; ***C1, C2*** and ***C3*** are constants; ***Ld*** is day length; ***T*** is air temperature; and ***h*** is air humidity term.

The FAO24 Blaney-Criddle is an empirical method originally developed by Blaney & Criddle (1945, 1950, 1962 ***in*** Jensen ***et al.*** 1990) and modified by Doorenbos & Pruitt (1977). The version used in this paper was presented by Allen & Pruitt (1986). According to these authors, the grass reference evapotranspiration can be related with Blaney-Criddle ***f*** factor by:

$$\boxed{\begin{gathered} Eto = a + bf \\ f = p(0.4T + 8.13) \\ a = 0.0043 RH_{min} - n/N - 1.41 \\ b = a_o + a_1 RH_{min} + a_2 n/N + a_3 U_d + a_4 RH_{min} n/N + a_5 RH_{min} U_d \end{gathered}}$$

**Equation 2**

where ***Eto*** is grass reference evapotranspiration (mm/d); ***p*** is mean daily percent of annual daytime hours; ***n/N*** is the ratio of possible to actual sunshine hours; ***RHmin*** is the minimum daytime relative humidity (%), ***Ud*** is the average daytime wind speed (m/s) at 2m height and $a_i$ is the Frevert regression coefficients (Jensen ***et al.*** 1990).

The radiation methods are based on the vertical energy balance at the soil surface. The net radiation flux is the sum of latent heat of vaporization *(8)* multiplied by potential evaporation *(Et)*, sensible heat flux to the air *(H)* and sensible heat flux to soil *(G)*. This approach generally uses net radiation and air temperature. The Radiation method assumes that the mass-transfer term of the Penman-combination method can be neglected if the air is saturated with water (Slatyer & McIlroy, 1961) and a constant supply of free water exists at the soil surface. Also as much as 80-90% of the net radiation is consumed in the evaporative process. According to Jensen (1966), the radiation methods generally assume one of the following forms:

$$\boxed{\begin{array}{c} lE = K_c[1 - \frac{(G+H)}{R_n}]R_n \\ lE = K_c[1 - a - \frac{R_b}{R_s} - \frac{(G+H)}{R_s}]R_s \end{array}}$$

**Equation 3**

where: *″* is the albedo; ***Rs*** is the solar radiation received at the earth's surface on a horizontal plane; ***Rn*** is the net radiation; ***Rb*** is the net outgoing long wave radiation and ***Kc*** is a constant of proportionality.

The Priestley & Taylor (1972) method is a simplification of the Penman Combination technique, with the exclusion of the aerodynamic component. As well, the Penman energy component was multiplied by ", to convert equilibrium potential evapotranspiration into potential evapotranspiration:

$$\boxed{lE_p = a\frac{\Delta}{\Delta + g}(R_n - G)}$$ **Equation 4**

where ) is the slope of the saturation vapor-pressure curve and *(* is the psychrometric constant.

The FAO24 (Doorenbos & Pruitt 1977) method is used if wind and humidity measurements are not available. Frevert *et al.* (1983 *in* Jensen *et al.* 1990) developed an equation for estimating *the b* term in the FAO24 formulation. In the original FAO24 method, *b* was calculated in graphic and tabular forms.

$$\boxed{\begin{array}{c} E_{to} = a + b[\frac{\Delta}{\Delta + g}R_S] \\ b = 0.9 - 0.0013RH_{mean} + 0.045U_d - 0.0002RH_{mean}U_d - 0.0000315RH_{mean}{}^2 - 0.0011U_d{}^2 \end{array}}$$ **Equation 5**

where *a* = -0.3mm/day; ***RHmean*** is the average of maximum and minimum daily relative humidity and ***Ud*** is the mean daily wind speed.

The Turc method calculates evapotranspiration for 10-day periods and was developed for temperate climates (Turc 1961). For relative humidity >50%:

$$\boxed{E_{to} = 0.013\frac{T}{(T+15)}(R_S+50)}$$ **Equation 6**

The Hargreaves method estimates solar radiation ($R_S$) from extraterrestrial radiation ($R_A$) (Hargreaves *et al.* 1985). The crop reference evapotranspiration is calculated by:

$$\boxed{E_{to}=0.002R_A\sqrt{TD}(T+17.8)}$$ **Equation 7**

where $R_A$ is given by the Duffie & Beckman (1980) equation and *TD* is the difference between mean monthly maximum and minimum temperatures.

The **Combination Approaches** are based on the Penman combination equation. The Penman (1948) method is the most widely known potential evapotranspiration method and perhaps one of the most reliable. The method uses net radiation, air temperature, wind speed and relative humidity. Basically, the equation follows this structure (Dingman, 1994):

$$\boxed{E \propto \frac{\Delta\ net\ radiation\ +\ g\ *\ mass\ transfer}{\Delta+g}}$$ **Equation 8**

where ) is the slope of the relation between saturation vapor pressure and temperature; *(** is the psychrometric constant modified by the canopy ratio, and *MT* is the mass transfer coefficient.

The Kimberly-Penman equation is basically the same as that presented by Penman (1963), with some modifications to the alfalfa-reference crop ($8E_{tr}$) in the mass transfer term and the wind function (*Wf*) (Wright 1982):

$$\boxed{\begin{aligned} &l E_{tr} = \frac{\Delta}{\Delta + g}(Rn - G) + \frac{g}{\Delta + g} 6.43 W_f (e^o{}_z - e_z) \\ &W_f = a_w + b_w U_2 \\ &a_w = 0.4 + 1.4 \exp\{-[(D - 173)/58]^2\} \\ &b_w = 0.007 + 0.004 \exp\{-[(D - 243)/80]^2\} \end{aligned}}$$

**Equation 9**

where $U_2$ is the wind speed at 2m height (km/d); *D* is the Julian day; $e^{o_z}$ is the saturation vapor pressure of air at height z; $e_z$ is the air vapor pressure at height z.

Jensen (1974) and Wright & Jensen (1972) introduced the following wind functions, respectively:

$$\boxed{W_f = 1 + 0.0062 U_2} \quad \boxed{W_f = 0.75 + 0.0115 U_2}$$

**Equation 10**

The Penman-Monteith (Monteith 1981) technique is one of the most encompassing equations to estimate evapotranspiration. This method includes, in addition to the original Penman equation, aerodynamic and surface resistance terms, aerodynamic resistance to sensible heat, vapor transfer ($r_a$), and surface resistance to vapor transfer ($r_c$):

$$\begin{aligned}
& l E_{tr} = \frac{\Delta}{\Delta + g^*}(R_n - G) + \frac{g}{\Delta + g^*} K_1 \frac{0.622\, l\, r}{P} \frac{1}{r_a}(e^o_z - e_z) \\
& r_a = \frac{\ln[(z_w - d)/z_{om}] \ln[(z_p - d)/z_{ov}]}{U_z (0.41)^2} \\
& g^* = g\,(1 + r_c / r_a)
\end{aligned}$$

**Equation 11**

where $z_w$ is the height of wind speed measurement; $z_p$ is the height of the psychrometer and temperature measurements; $Z_{om}$ is the roughness length (momentum); $d$ is the zero plane displacement of wind profile; $K_1$ is the unit dimension coefficient and $P$ is atmospheric pressure.

## FIELD SITE AND INSTRUMENTATION

The field site is located on a private farm near Kintore, Ontario. The soil series is the Embro silt loan (Orthic Grey-Brown Luvisol). Field crops during 1995 and 1996 were an alfalfa and timothy mixture and corn, respectively.

The meteorological station contained a silicon pyranometer (Li-Cor Model LI-20052) to measure incoming solar radiation ($R_S$), at 3m above ground level., air temperature and air relative humidity probes (Campbell Scientific Model 207) located at 1.35m above ground.. A cup anemometer (R.M. Young Model 207) located at 3.66m above ground and a tipping bucket rain gauge, located 3m from the met-station, were installed as well. Measurements of radiation, temperature and relative humidity were recorded every 60 seconds and stored on a Campbell Scientific 21x datalogger system. The wind speed and rain volume were recorded continuously and the average wind and total rain were stored every hour.

Some of the information required for evapotranspiration calculations was obtained from theoretical and empirical relationships. An estimation of vapor pressure deficit was calculated from relative humidity by the equation of Jensen *et al.* (1990). The anemometer measurements were corrected to 2m height by the Penman (1948 *in* Penman 1963) equation. Mean extraterrestrial ($R_A$) and cloudless ($R_{SO}$) solar radiation were calculated based on the site latitude and day of the year using equations from Gates (1962) and Budyko (1963), respectively. The soil heat ($G$) was based on a relationship between air and soil surface temperatures.

The net radiation, $R_n$, is the balance of incoming and outgoing radiation. In this paper, $R_n$ was defined by measured $R_S$, calculated albedo (0.23), $R_{SO}$, and $R_b$ (net outgoing long-wave radiation). To estimate $R_b$, we used the typical coefficients *a*=1.017 and *b=-0.06 (Rs/Rso>0.7)* or *a=1.126* and *b=-0.07* (*Rs/Rso<0.7*), according to Wright's (1982) *Rn* equation.

Sixteen TDR probes of 0.2 and 0.4m lengths were installed vertically from the soil surface. The method of Topp *et al.* (1980) was used to estimate soil-water content from measured soil dielectric constant. Tensiometers were installed near the TDR probes to ensure that any charge in water content was due to evaporation and not infiltration.

## RESULTS

### Potential Evapotranspiration Estimates

Figure 1 presents the results of nine methods of estimating monthly E*tp* that were selected for evaluation based on climatological factors averaged daily with data obtained hourly during 1995 at Kintore site. The monthly evapotranspiration data shown in this figure is the sum of daily E*tp*.

Four *combination methods* (Figure 1a) show a good correlation with each other over all months. The best correlation was in spring and fall, when the four lines were almost coincident. However, during the summer, including the peak-month of June, the E*tp* are slightly higher than Penman-Monteith. Similar results were obtained by Jensen *et al*. (1990) for approximately the same geographical conditions. These authors, comparing Penman-Monteith and other *combination approaches*, showed a similar behavior through the year: the results were more accurate in spring and fall, with dispersion in June and July (differences up to 50%).

At the Kintore experimental site, *radiation and temperature approaches* gave higher E*tp* variance between methods than the *combination* approaches (Figure 1b). In particular, the Hargreaves method gave a total seasonal E*tp* 18% higher than the Penman-Monteith technique. On the other hand, the Priestley-Taylor technique gave the best correlation with *combination approaches* for the 1995 seasonal E*tp*.

Jensen *et al.* (1990) applied 18 different evapotranspiration methods in eleven places around the world. Independent of the study area and climate condition, *combination approach* results were overestimated when they were compared to actual evapotranspiration from lysimeter data. Other approaches, such as *temperature, radiation and pan-evapotraspiration*, showed higher E*tp* dispersion, when analyzed under the same conditions. As at Kintore, analyzing data from the whole year, Jensen's study showed that temperature methods overestimate E*tp* compared to the Penman-Monteith method.

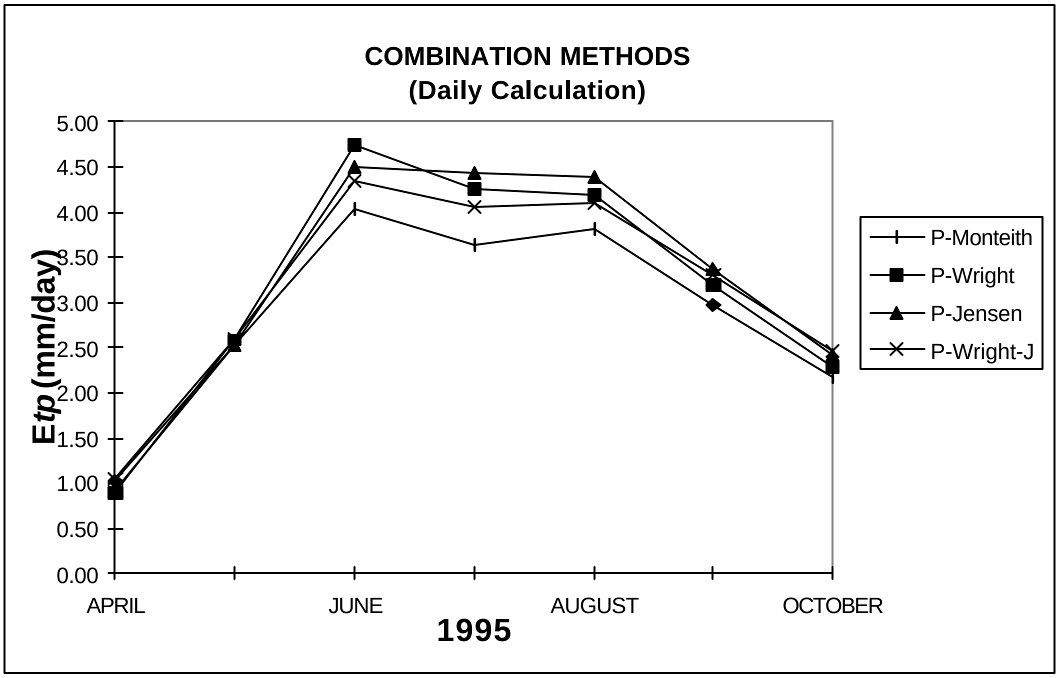


Figure 1a. Estimates of potential evapotranspiration by combination approaches (calculated on a daily base) compare with Penman-Monteith method.

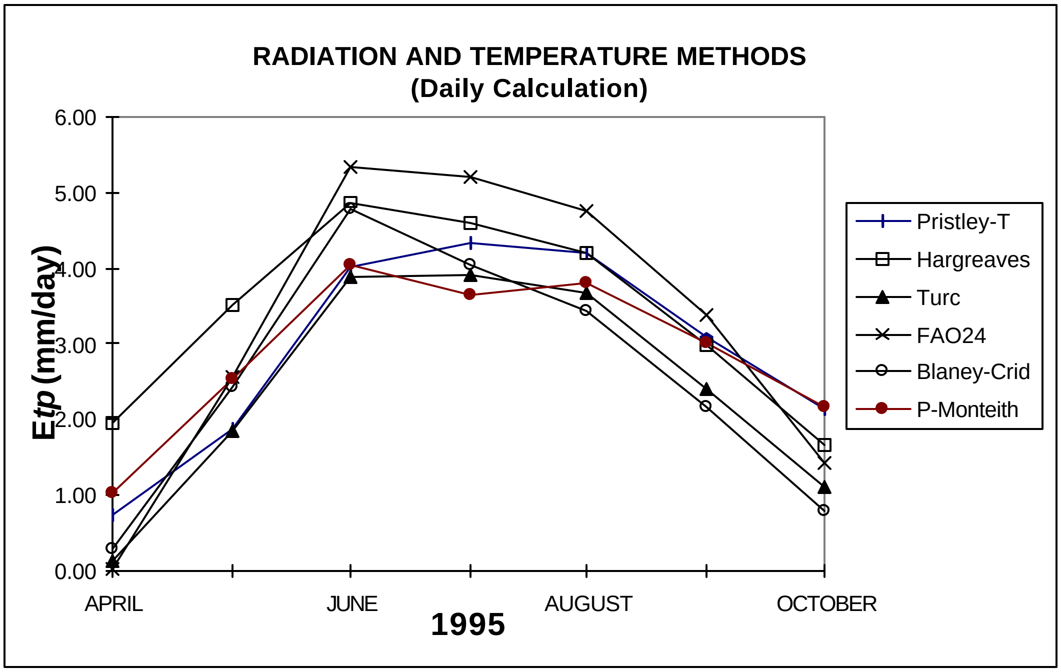


Figure 1b. Estimates of potential evapotranspiration by radiation and temperature approaches (calculated on a daily base) compare with Penman-Monteith method.

The Blaney-Criddle and FAO24 methods gave high results (20% above the combination methods) during June and exceptionally low values in April (71% below the combination methods). Because of these variations, the annual E*tp* for this method was about 10% lower than *the combination methods*. April was an exceptional month at Kintore in 1995, with low temperatures and many cloudy days ($R_S$=1.22 MJ/m$^2$d). Therefore *Radiation approaches*, with the exception of Hargreaves, will give low values of E*tp*. Hargreaves was not affected by April conditions, because its calculation is based on $R_A$ (extraterrestrial solar radiation), instead of $R_S$ (solar radiation at the surface). Generally in 1995, *radiation approaches* gave higher E*tp* than historical E*tp* data, especially for FAO radiation method.

Figure 2 compares nine E*tp* methods. In this case, the E*tp* was calculated using the monthly average of appropriate climatological factors. All methods overestimated E*tp* relative to results shown in Figures 1a and 1b. A possible explanation is that for the E*tp* calculation, almost all formulas used maximum and minimum values of temperature or relative humidity. If the E*tp* was calculated on a daily basis, these extreme values can affect only specific days, but when the calculation was based on a monthly average basis, these values affect the whole month.

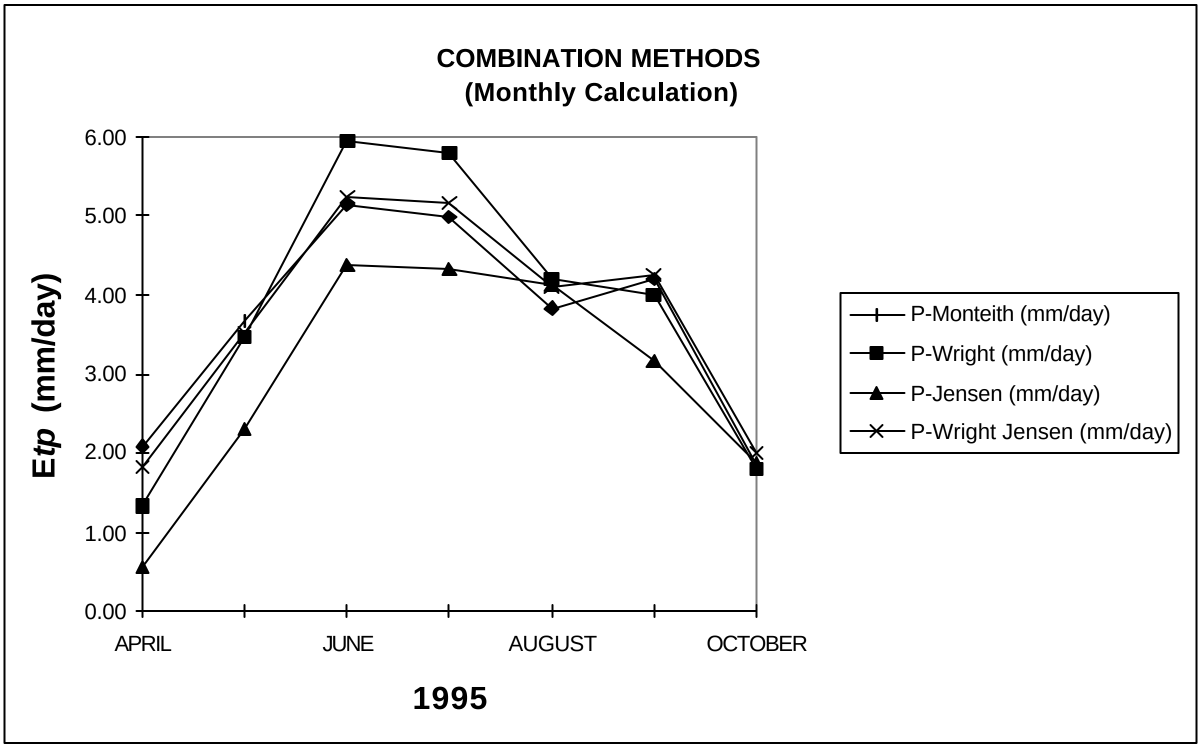


Figure 2a. Estimates of potential evapotranspiration by combination approaches (calculated in a monthly base) compare with Penman-Monteith method.

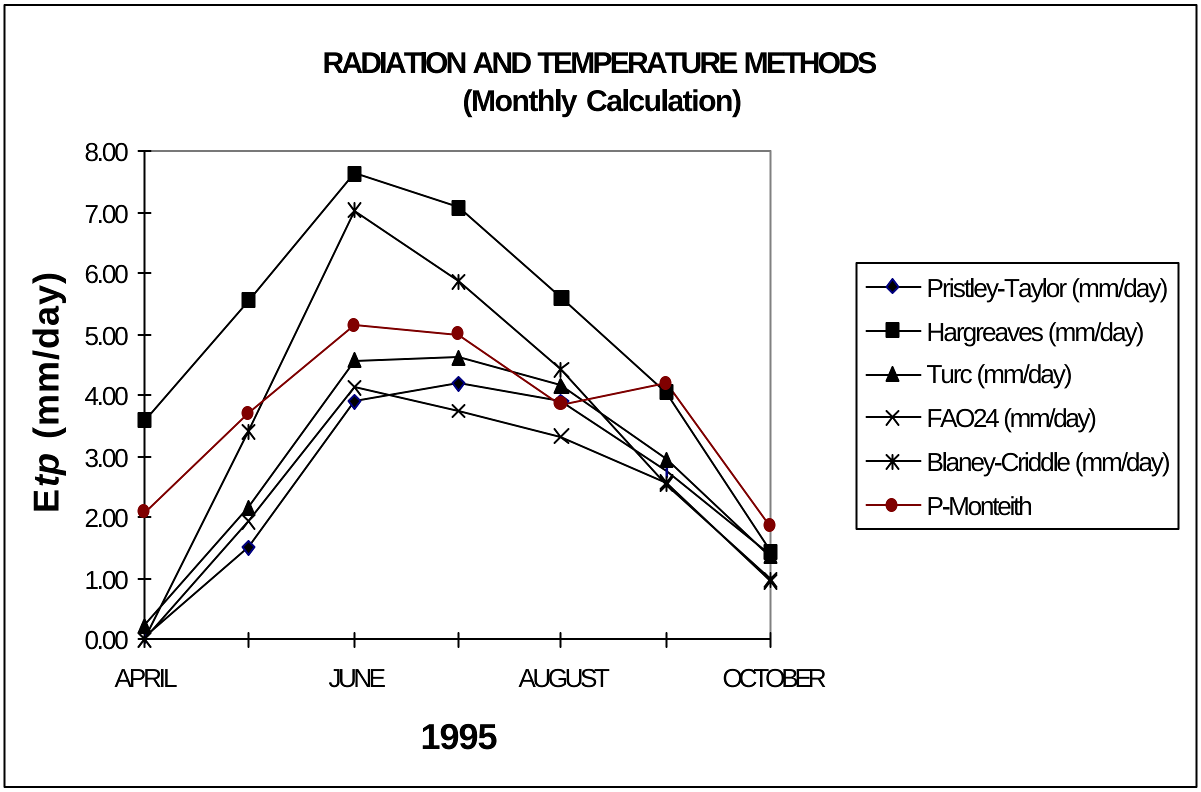


Figure 2b. Estimates of potential evapotranspiration by radiation and temperature approaches (calculated in a monthly base) compare with Penman-Monteith method.

All combination approaches, using the average monthly data, had the same behavior for all months, except for the Penman-Jensen method, that did not show a peak in September. This high E*tp* value was not apparent when the calculations were made with data averaged daily (Figure 1a).

Penman-Jensen technique uses just average air temperature to calculate vapor pressure deficit, instead of maximum and minimum temperature, that are required by other methods. September had a low T*min* (-1.5 C), which influenced the vapor pressure calculation of all techniques, other than the Penman-Jensen method.

The results from ***combination approaches***, based on daily measurements and calculations, were consistent with historical E*tp* series measured at Harrow C.D.C. (1951-1980), Guelph O.A.C. (1951-1980) and Vineland (Brown, not published), except for the peak-month (June for Kintore and July for other areas) when the values at Kintore were slightly higher. Guelph O.A.C. (approximately 120 km east of Kintore is the closest meteorological station that has historical E*tp* calculation series.

Table 2 presents ***standard errors of estimate*** (SEE), between E*tp* (daily and monthly bases) calculated by eight methods and

Penman-Monteith technique using daily averages. SEE was calculated from the absolute differences between two values and indicates how well each method was able to estimate E*tp,* based on a reference value*.* If the error or difference has a normal distribution, the SEE represents results of a maximum error with 68% of confidence.

For a monthly E*tp* based on monthly average of climatological factors, the best methods were Penman-Wright-Jensen, Penman-Wright, Penman-Jensen, Turc, Priestley-Taylor and FAO24, in this order. Hargreaves and Blaney-Criddle presented high SEE values, mainly in June and April, as a result of low $R_s$ and unusually high wind speeds for those months.

E*tp* based on a daily average of climatological factors showed similar results. The Penman-Wright-Jensen; Penman-Wright and Priestley-Taylor methods gave the best comparison, and the worst were associated with FAO24, Blaney-Criddle and Hargreaves. The two latter methods gave, in both cases (ie. based on monthly and daily calculations), the greatest difference in E*tp*, compared with the Penman-Monteith method. The similarity between E*tp* calculated by the *combination approaches* was anticipated because all methods use the same basic equations, with different techniques to obtain parameters.

Using Guelph historical E*tp* as a reference, Hargreaves and Blaney-Criddle gave a closer comparison than Penman-Monteith. These results mean that the simpler methods work better when micro-climatological parameters do not change significantly in relation to seasonal averages. The use of daily or hourly measurements and daily E*tp* calculations have shown superior results to those obtained by monthly measurement periods for all methods.

With the objective of understanding the length of time for which a specific method is applicable, it was possible to compare the daily E*tp* calculated for the month of July, using hourly average measurements. Based on the sensitivity of each method to daily variation in micro-climatological factors, it was possible to identify three different groups. The first group of most sensitive methods included all of the ***combination approaches***. A second group, including Priestley-Taylor, FAO24 and Blaney-Criddle, presented an intermediate performance and finally, a third one included the methods with low daily E*tp* variation, namely Hargreaves and Turc.

These results show that c*ombination approaches* work well on a daily and hourly basis and the recommended minimum time periods for *radiation and temperature approaches* varies from 5 to 10 days (Table 1). For the Penman-Monteith method, Jensen *et al*. (1990) stated that the best results were obtained when the time interval for data collection is hourly, with results calculated on a daily basis.

| METHOD | $T_{MEAN}$ | $T_{MAX}$ $T_{MIN}$ | $R_A$ | $R_{SO}$ | $R_S$ | $R_N$ | G | n/N | U | $RH_{MEAN}$ | $RH_{MIN}$ | (e°-e) |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Penman (1963) (Jensen 1974) | 3 | or $R_N$ and (e°-e) | | 3 | 3 | or $T_{MAX}$ $T_{MIN}$ | 3 | or $R_{SO}$ and $R_S$ | 3 | 3 | | or $R_H$ and T |
| Penman-Monteith (Monteith 1981) | 3 | or $R_N$ and (e°-e) | | 3 | 3 | or $T_{MAX}$ $T_{MIN}$ | 3 | or $R_{SO}$ and $R_S$ | 3 | 3 | | or $R_H$ and T |
| Kimberly-Penman (Wright 1982) | 3 | or $R_N$ and (e°-e) | | 3 | 3 | or $T_{MAX}$ $T_{MIN}$ | 3 | or $R_{SO}$ and $R_S$ | 3 | 3 | | or $R_H$ and T |
| Penman (1963) Wright & Jensen (1972) | 3 | or $R_N$ and (e°-e) | | 3 | 3 | or $T_{MAX}$ $T_{MIN}$ | 3 | or $R_{SO}$ and $R_S$ | 3 | 3 | | or $R_H$ and T |
| Priestley-Taylor Priestley & Taylor (1972) | 3 | or $R_N$ | | | 3 | or $T_{MAX}$ $T_{MIN}$ | 3 | | | 3 | | or $R_H$ and T |
| Turc (1961) | 3 | | | | 3 | | | | | | | |
| FAO24 Doorenbos & Pruitt (1977) | 3 | | | | 3 | | | or $R_{SO}$ and $R_S$ | 3 day-time | 3 | 3 | or $R_H$ and T |
| Hargreaves *et al.* (1985) | 3 | | 3 | | | | | | | | | |
| Blaney-Criddle (Doorenbos & Pruitt 1977) | 3 | | | | | | | or $R_{SO}$ and $R_S$ | 3 day-time | | 3 | |

**$T_{MEAN}$,$T_{MAX}$ $T_{MIN}$** = average, maximum and minimum air temperature; **$R_A$** = Extraterrestrial solar radiation; **$R_{SO}$** = solar radiation on a cloudless day; **$R_S$** = solar radiation received at the earth surface; **$R_N$** = Net radiation; **G** = Heat flux density to the ground; **n/N** = ratio between actual measured bright sunshine hours and maximum possible sunshine hours; **U =** Wind speed; **RH = Relative humidity; (e°-e)** = Saturated vapor pressure-water vapor pressure.

Table 1. Climatic data required for each technique to estimate potential evapotranspiration.

| | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|
| | *Etp* | P-Monteith | P-Wright | P-Jensen | P-W-J | P-T | Hargreaves | Turc | FAO24 | B-C |
| DAILY BASE | annual Etp (mm) | 618 | 677 | 690 | 671 | 623 | 728 | 519 | 695 | 548 |
| | SEE (mm/a) | - | 12.7 | 14.5 | 9.2 | 13.4 | 24.5 | 21.1 | 32.4 | 24.9 |
| | SEE (mm/day) | | 0.98 | 1.06 | 0.98 | 1 | 0.74 | 0.93 | 1.33 | 0.82 |
| MONTHLY BASE | annual Etp (mm) | 788 | 812 | 634 | 799 | 539 | 1065 | 611 | 507 | 741 |
| | SEE (mm/a) | | 16.1 | 28.5 | 5.92 | 41.8 | 52.5 | 32.6 | 46.7 | 41.6 |
| | SEE (mm/day) | | 1.34 | 0.44 | 1.11 | 0.71 | 2.75 | 0.69 | 0.75 | 1.73 |
| | | | | | | | | | | |
| Calculated Potential and Actual Evapotranspiration (summer 1996) | | | | | | | | | | |
| | *Etp* | P-Monteith | P-Wright | P-Jensen | P-W-J | P-T | Hargreaves | Turc | FAO24 | B-C |
| Calculated *Etp* | 4month Etp (mm) | 437 | 491 | 497 | 469 | 424 | 538 | 391 | 527 | 452 |
| | SEE (mm/day) | | 0.48 | 0.58 | 0.29 | 0.72 | 2.79 | 1.38 | 3 | 1.7 |
| Calculated Actual I | 4 month Eta (mm) | 81 | 82 | 89 | 86 | 70 | 115 | 59 | 75 | 61 |
| | SEE (mm/day) | | 0.25 | 0.34 | 0.17 | 0.42 | 2.03 | 0.85 | 2.2 | 1.1 |
| | | | | | | | | | | |
| P-W-J = Penman, Wright, Jensen | | P-T = Priestley-Taylor | | B-C = Blaney-Criddle | | | | | | |
| | | | | | | | | | | |
| Table 2. Standard error of estimate (SEE) of eight techniques in relation to Penman-Monteith method. SEE represents deviation with 68% of confidence. | | | | | | | | | | |
| Calculated Potential Evapotranspiration (1995) | | | | | | | | | | |

| | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|
| | AVERAGE AIR TEMPERATURE | | | | | | | | |
| | P-Montheit | P-Wright | P-Jensen | P-W-J | P-T | Hargreaves | Turc | FAO24 | B-C |
| +1% | -0.26 | -0.70 | 0.60 | -0.62 | 0.60 | 0.56 | 0.36 | 0.38 | 0.76 |
| - 1% | -1.55 | -0.83 | 0.57 | -0.73 | 0.58 | 0.56 | 0.49 | 0.42 | 0.76 |
| | MAXIMUM TEMPERATURE | | | | | | | | |
| | P-Montheit | P-Wright | P-Jensen | P-W-J | P-T | Hargreaves | Turc | FAO24 | B-C |
| +1% | 2.84 | 1.65 | -0.06 | 1.53 | -0.08 | 0.63 | 0.00 | 0.00 | 0.00 |
| - 1% | 0.71 | 1.28 | -0.06 | 1.19 | -0.06 | 0.71 | 0.00 | 0.00 | 0.00 |
| | MINIMUM TEMPERATURE | | | | | | | | |
| | P-Montheit | P-Wright | P-Jensen | P-W-J | P-T | Hargreaves | Turc | FAO24 | B-C |
| +1% | 0.27 | 0.14 | -0.01 | 0.12 | -0.01 | -0.19 | 0.00 | 0.00 | 0.00 |
| - 1% | -0.08 | 0.08 | -0.01 | 0.08 | -0.01 | -0.16 | 0.00 | 0.00 | 0.00 |
| | SOLAR RADIATION (Rs) | | | | | | | | |
| | P-Montheit | P-Wright | P-Jensen | P-W-J | P-T | Hargreaves | Turc | FAO24 | B-C |
| +1% | 0.98 | 0.67 | 0.79 | 0.68 | 0.92 | 0.00 | 0.89 | 1.11 | 0.81 |
| - 1% | 0.50 | 0.67 | 0.79 | 0.69 | 0.92 | 0.00 | 0.90 | 1.11 | 0.81 |
| | WIND | | | | | | | | |
| | P-Montheit | P-Wright | P-Jensen | P-W-J | P-T | Hargreaves | Turc | FAO24 | B-C |
| +1% | 0.36 | 0.14 | 0.05 | 0.14 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| - 1% | 0.05 | 0.14 | 0.05 | 0.15 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| | RELATIVE HUMIDITY | | | | | | | | |
| | P-Montheit | P-Wright | P-Jensen | P-W-J | P-T | Hargreaves | Turc | FAO24 | B-C |
| +1% | -1.92 | -0.63 | -0.57 | -0.59 | 0.19 | 0.00 | 0.00 | -0.99 | 0.00 |
| - 1% | -1.35 | -0.63 | -0.57 | -0.58 | 0.20 | 0.00 | 0.00 | -0.93 | 0.00 |

Table 3. Sensitivity between micro-climatological parameter variation and estimate of potential evaportanspiration.

The sensitivity of each method to changes in the parameters required to calculate E*tp* is given in Table 3. It presents the effect of the variation of each micro-climatological parameter (by +1% or -1%) on E*tp.* in percentage of daily evapotranspiration, for all nine methods. It is apparent that the combination methods are most sensitive to changes in relative humidity, solar radiation ($R_s$) and maximum and average temperatures. According to the information in Table 3 and observations from all methods using the daily basis calculation, wind and minimum temperature have less influence on E*tp* estimation than other parameters.

Jensen *et al.* (1990) state that the main cause of error when using the *combination methods* was the use of daily or monthly averages of micro-climatological measurements. According to the authors, "these parameters are often significantly out of phase with one another during the course of a diurnal cycle, thereby weighting cumulative evaporative demands over a 24-hour period, disproportionately when using daily average climatic parameters". This problem can be avoided, if E*tp* estimation is calculated on an hourly basis and summing individual values to obtain daily totals.

In Table 1 we presented the micro-climatological information required to calculate potential evapotranspiration for all nine methods examined in this paper. It is important to recognize the differences between estimated parameters and data that needs to be measured directly in the field for each method. A basic meteorological station is equipped with a thermometer, anemometer; hygrometer; radiometer; pluviometer, and sometimes a soil-temperature thermocouple. The *combination approaches* need all of the parameters that are given in the Table 1. In contrast, Hargreaves' method only needs air temperature, because $R_A$ can be estimated from altitude and latitude of the study area. Unfortunately, this method has presented the poorest performance amongst the studied methods. For a more reliable method, Turc can be used to estimate E*tp*. In addition to air temperature, $R_S$ measurements are also necessary. The Priestley-Taylor method is also to be considered if $RH_{mean}$ measurements are obtained regularly. The accuracy of Turc and Priestley-Taylor E*tp* are lower if used over periods less than 10 days.

Among the methods that do not use wind speed or humidity, Turc has presented one of the lowest errors on a monthly basis estimation in 1995, at Kintore. Comparing it to Penman-Monteith, the best fit months occur in summer and the highest errors occur in April and October. Wind speed was highest in April in 1995. This gave a reduction in Penman-Monteith E*tp*, but it did not affect Turc. In October, the Turc method gave a lower estimate of E*tp* than Penman-Monteith.. Although the Turc method presented some errors relative to Penman-Monteith, its simplicity and low information requirement make it one of the most attractive methods for temperate climatic areas, particularly if it is only necessary to calculate total monthly E*tp* using daily meteorological measurements.

The Priestley & Taylor method gave a similar performance as Turc. However, this method needs, in addition to the information required by Turc, the $RH_{mean}$. For monthly E*tp* based on daily calculations, the method gave a better comparison to Penman-Monteith than Turc. The advantage of this method is that it can estimate daily evapotranspiration without wind measurements, which are necessary for *combination approaches*.

**Actual Evapotranspiration Estimates**

Potential evapotranspiration measured in 1996 was different than that measured during 1995 for almost all methods. In 1995, June was the peak month, with high E*tp* values in July and August. In contrast, in 1996 the peak month was August (Figure 3). Analyzing individually each month in 1996, the values of E*tp* were higher than in 1995, except for Hargreaves. A possible reason was the lower incidence of total solar radiation, because of cloudy periods.

Analyzing the SEE for all methods, again calculated in comparison with Penman-Monteith values (Table 2), lower deviations were obtained by Penman-Wright & Jensen, followed by Penman-Wright, Penman-Jensen, Priestly-Taylor, and Turc. The worst results were associated with

FAO24, Hargreaves and Blaney-Criddle FAO24. Practically the same relationship was obtained using results from 1995. Using the Turc method, the calculated actual evapotranpiration (E*ta*) presented better (lower) SEE than for E*tp*.

This method was the only one that increased its performance when we compared the two different evapotranspiration parameters.

Calculated E*ta* using Penman-Monteith and Priestley-Taylor methods gave esentially the same values during June to September 1996. This period was characterized by no exceptional wind events, that did not cause significant changes in the wind factor of combination method parameters.

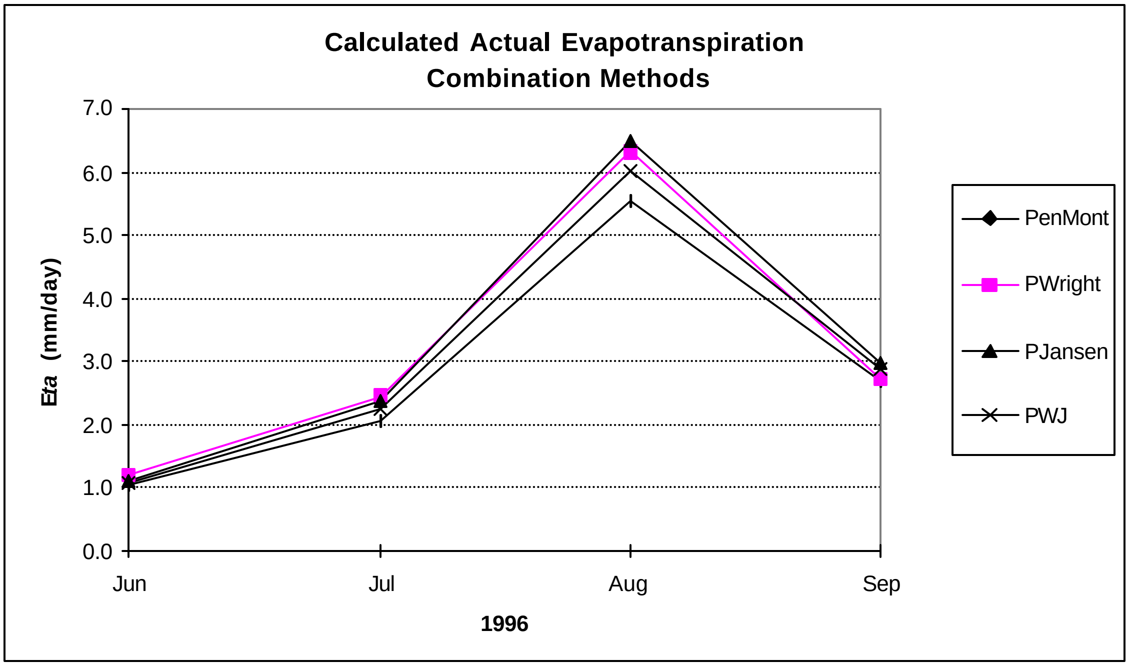


Figure 3a. Estimates of calculated actual evapotranspiration by combination approaches (calculated on a daily base).

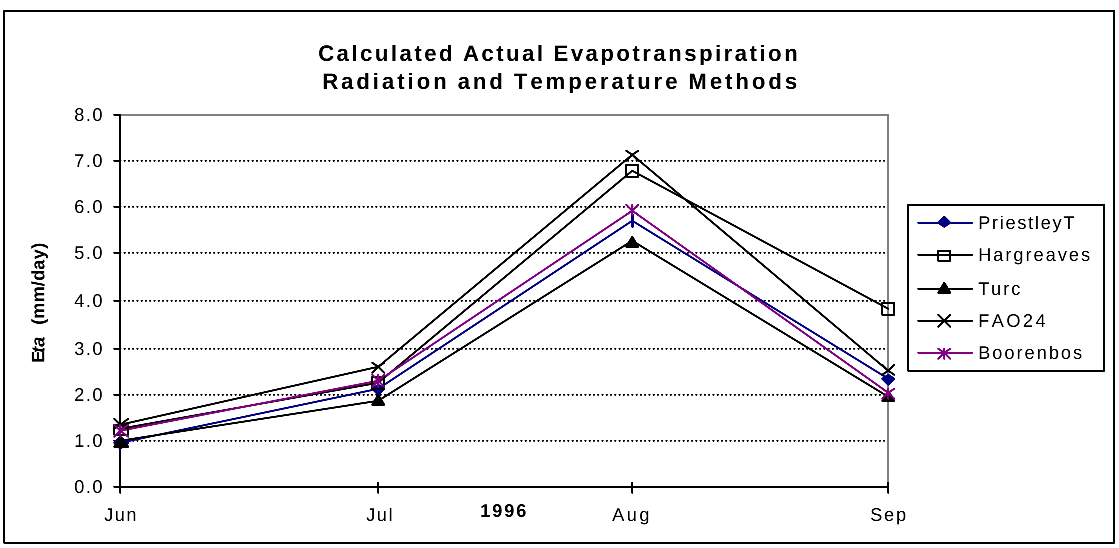


Figure 3b. Estimates of calculated actual evapotranspiration by radiation and temperature approaches (calculated in a monthly base).

During the summer of 1996, measurement of E*ta* at the Kintore site was possible. Figure 4 shows the cumulative actual evapotranspiration for all methods studied and measured E*ta* from 0 to 0.2 and 0 to 0.4m depths below the soil surface. All methods show good correlation in the month of July (the fist part of the 0.4m depth TDR E*ta* curve). The values for 0.2m depth TDR were the lowest in relation to all methods, including 0.4m TDR. The 0.4m TDR probes include the influence of the corn root zone during that month. The difference between the two TDR probe lengths increases with  time, following plant growth. Normally, the evapotranspiration relationship for mature crops is 30% from soil and 70% from plants.

After August 1,  the calculated E*ta,* for all methods, increases faster than 0.4 TDR E*ta* (Figure 4). The principal reason for this is the calculated methods assumed no soil-water limitation. At Kintore and all South of Ontario, the months from May to August are characterized by deficit in the water balance. No artificial irrigation was implemented. Tensiometer measurements of the hydraulic head below the 0.4 m level indicated that after August 1 there was upward flow of water between 0.6 and 0.8 m depth.  Therefore, corn roots probably extracted water from below 0.4 m.  This water loss would not have been measured by TDR.

The measured E*ta* for the month of July (55mm), when the deficit was not so effective, showed that Penman-Monteith (64mm), Priestley-Taylor (65mm), and Turc (58mm) methods had presented the best results for E*ta.* In contrast, FAO24 (80mm) and Hargreaves (69mm) gave the poorest comparison.

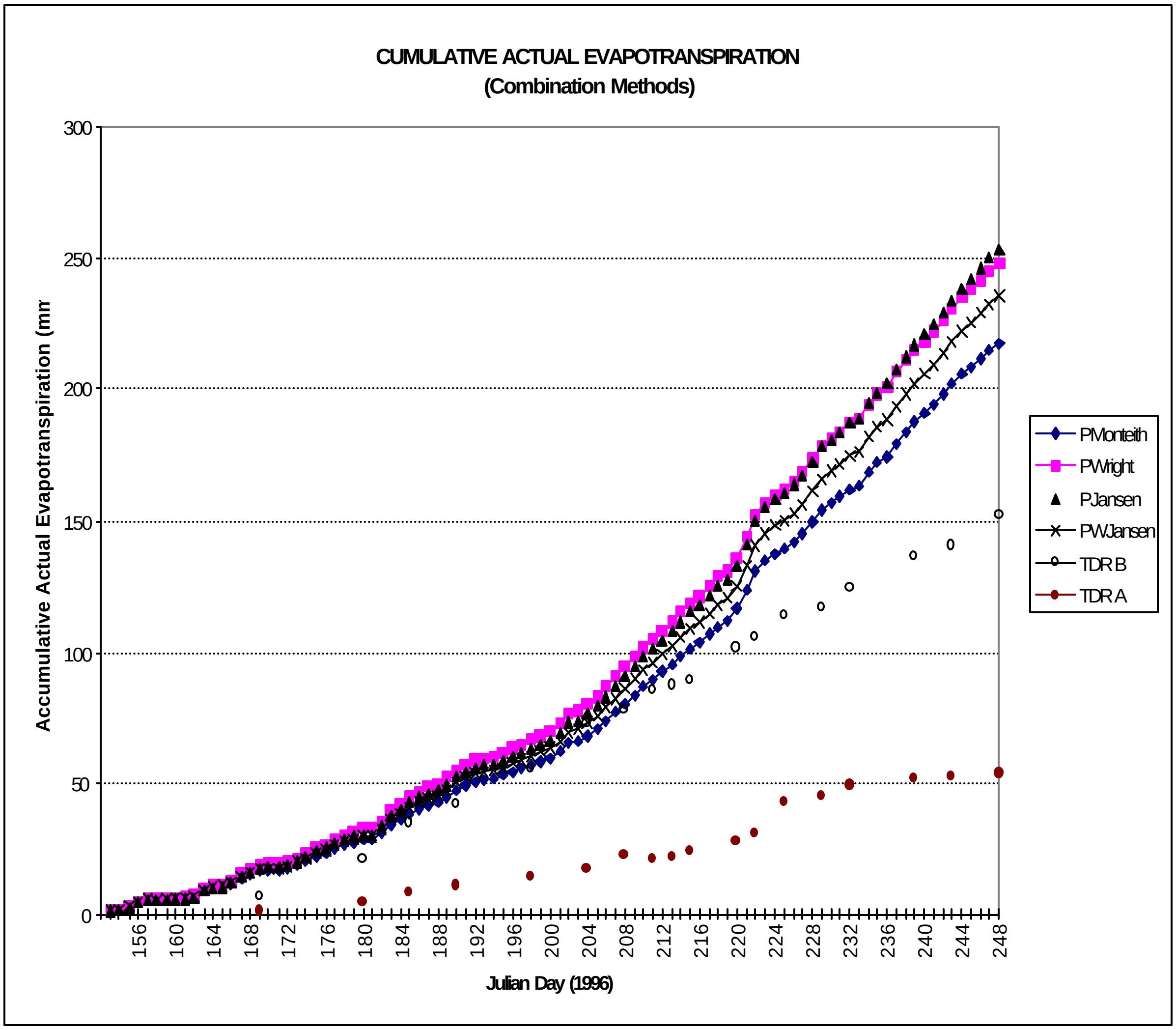


Figure 4a. Cumulative actual evapotranspiration calculated by combination methods, compared with measured actual evapotranspiration.

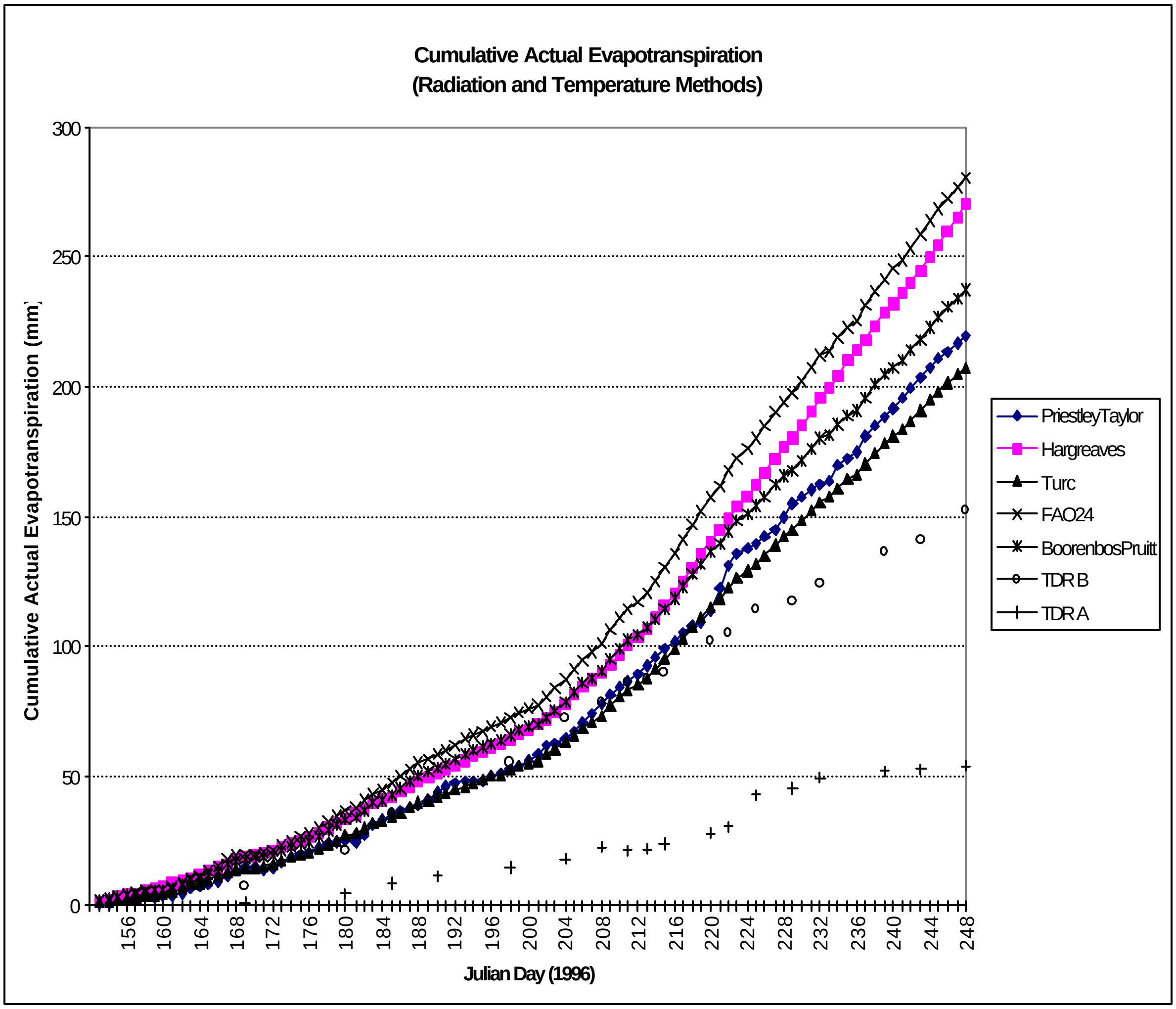


Figure 4b. Cumulative actual evapotranspiration calculated by temperature and radiation methods, compared with measured actual evapotranspiration.

Based on a linear regression analysis, we used the 0.4m TDR data to calculate an equation for E*ta* for the Kintore site. The curve is *y=1.67x-b, .w*here, *x* is the 1996 Julian day and *y* is the cumulative actual evapotranspiration (mm). *b* represents the *y* value, when *x* is 0. In this particular case, *b* is -270. The correlation coefficient is 0.98 (Figure 4a,b)

**CONCLUSIONS**

Nine different methods of estimating potential evapotranspiration were analyzed and their results were compared to an accepted standard method, namely the Penman-Monteith technique. The calculations were done on daily, monthly and seasonal bases and the main conclusions are given below. In ta similar fashion, we compared actual evapotranspiration, that was measured by TdR in the summer of 1996, and the same nine methods:

C The comparison of all methods with Penman-Monteith based on daily E*tp* calculations showed that Penman-Wright-Jensen, Penman-Wright and Priestley-Taylor methods had the best performance for establishing the monthly E*tp*, and, in contrast, Hargreaves, Blaney-Criddle and FAO24, the poorest. When the comparison was based on historical micro-climatological measurements, Penman-Monteith was one of the best methods to estimate monthly E*tp.* Hargreaves and Blaney-Criddle give better estimates of E*tp* under these conditions, because the methods are relatively insensitive to non-average climatological fluctuations. Although, the ***combination approaches*** tend to overestimate E*tp* in relation to historical Guelph E*tp* (University of Guelph met-station), the Penman-Monteith method was one of the most reliable and robust techniques, when the complete micro-climatological data were available. This fact is corroborated by many studies conducted in different humid and temperate climate zones.

- Actual evapotranspiration was measured during the summer of 1996 (June to September) at Kintore site, using TDR. The results, when they were compared with combination, temperature and radiation methods, showed that, Penman-Monteith, Priestley-Taylor and Turc techniques gave the closest fit to measured data. The worst predictions were associated with FAO24 and Hargreaves. Unfortunately, the results were not conclusive, because of the short period of TDR measurements.

- After August 1, 1996, it was observed that the calculated E*ta,* for all methods, increased faster than measured E*ta*. We postulate that the crop roots extracted water from below the

TDR sample volume. Thus, the calculated E*ta* was assumed to be correct.

- The measured E*ta,* when presented in a cumulative chart, showed a linear correlation with time. The linear equation can be express by: $y=1.67x-270$; where $y$ is the cumulative E*ta*; $x$ is the Julian day; and $b$ is the E*ta* intercept.

- If a limited amount of micro-climatological information is available, the Turc and Priestley-Taylor methods should be considered. The Turc method was accurate enough to estimate E*tp* on a monthly basis if only temperature and solar radiation ($R_{so}$) information is available. When the station has a hygrometer to record air humidity, the Priestley-Taylor technique is an option. However, at the Kintore experimental site, Priestley-Taylor did not give an improved comparison to Turc or Penman-Monteith..

- *Combination approaches* should be used to evaluate E*tp* on a daily basis. Some radiation methods, including Priestley-Taylor and FAO24, work well with a 5-10 day period. The Turc and Hargreaves methods have low sensitivity and the recommended minimum time period should be of the order 15 days or even monthly. The *temperature method* of Blaney-Criddle was designed for a 5 day period, but a minimum of 15 days should be considered. Independent of the method that was considered, the best monthly E*tp* calculated was obtained when all micro-climatological parameters were measured and E*tp* was calculated on a daily basis (or hourly, in some cases).

- The combination methods are most sensitive to errors in relative humidity, solar radiation, and temperature measurements.

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