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AGRI-ENVIRONMENTAL INDICATOR PROJECT

Agriculture and Agri-Food Canada

REPORT NO. 6

INDICATOR OF RISK OF WATER CONTAMINATION: METHODOLOGICAL DEVELOPMENT

WORKING DRAFT

Prepared by:

Bruce MacDonald and Harry Spaling

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1. INTRODUCTION

The purpose of this paper is to build on the hierarchical framework and concepts described in a companion paper to develop a methodology for the Indicator of Risk of Water Contamination (IROWC).[1] The methodology is proposed as a preliminary model only. It requires further development and continued testing to authenticate its utility among hierarchical levels, and at various temporal and spatial scales.

This discussion document describes the initial development and demonstration of IROWC, and suggests areas of further research, testing and implementation. The paper first briefly reviews a number of risk-based approaches to the analysis and assessment of water contamination by agriculture, based on previous research. It then proposes a methodology for IROWC using the budget approach. The proposed methodology is demonstrated utilizing preliminary data and, finally, suggestions for detailed implementation are presented.

2. RISK-BASED APPROACHES

The potential of nitrates or pesticides to contaminate water is frequently determined by screening or simulation models. Screening models assess and rank the leaching potential of inputs, generally on the basis of chemical properties. Simulation models incorporate various physical, chemical and biological attributes of the input and site to predict the amount of a contaminant which is available for contamination. Both models are apparent in risk-based approaches.

This section identifies and describes previous risk-based approaches to the analysis and assessment of nitrate and pesticide contamination of surface and ground water. It classifies the approaches into general categories and assesses their utility for IROWC using the desirable characteristics of IROWC identified in the companion paper.

2.1 RISK ANALYSIS USING SPATIAL INDICATORS

Spatial indicators generally map spatial attributes of IROWC components (e.g., precipitation, soil leaching potential, crop type) to identify areas at risk of contamination. GIS is a common tool for analysis and display.

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[1] MacDonald, K.B. and H. Spaling. 1995. Indicator of Risk of Water Contamination: Concepts and Principles. Working Paper. Draft of 21 March 1995, Agriculture and Agri-Food Canada, Ontario Land Resource Unit, Centre for Land and Biological Resources Research, Guelph, Ontario.

2.1.1 Vulnerability Mapping

Vulnerability maps identify and rate areas according to contamination potential. Vulnerability is based on the spatial association of attributes which affect contamination. Spatial analysis is usually integrated with a contaminant screening or modelling procedure.

A USDA study (Kellogg et al. 1992) developed vulnerability indexes for pesticide and nitrate contamination of shallow ground water. The Ground Water Vulnerability Index for Pesticides (GWVIP) is a function of soil-pesticide leaching, precipitation, and chemical use. The nitrate index (GWVIN) is based on precipitation and excess nitrogen fertilizer (i.e., applied minus crop uptake and removal). Index scores are grouped into relative categories of risk (high, moderate, low). Data are displayed at the national scale with county level resolution.

McRae (1991) also uses a vulnerability mapping procedure as part of a study to identify areas at risk of ground water contamination in Canada. It is less comprehensive than the USDA approach. The focus is on pesticide leaching potential, using only soil and landform characteristics derived from soil polygons on the Generalized Soil Landscape Maps (1:1,000,000). Areas vulnerable to pesticide leaching are characterized by sandy or sandy loam classes of soil texture, a slope gradient of 0-9%, and undulating, hummocky, level, or knoll and kettle classes of surface formation. McRae (1991) suggests improving the procedure by including ground water recharge data.

The above studies generalize on the basis of soil type (e.g., sandy soils are equated with high leaching potential), but caution is needed since leaching may be higher in some instances under other soils and certain management conditions (e.g., clay soils under ridge or no-tillage where preferential flow is greater).

In another application, Khan and Liang (1989) use a GIS-based approach to construct maps showing the likelihood of pesticide contamination of ground water for Hawaii (1500 km^2). A "contamination likelihood map" for various pesticides is based on an attenuation factor (AF) derived from the percent of applied chemical which leaches below the surface soil layers. A GIS is used to delineate the land area falling into different AF rating groups, and to calculate differences in contamination potential of the same pesticide for different crops.

These studies show that vulnerability mapping is capable of integrating IROWC components by determining their spatial association. The geographic scale and resolution of these studies suggest that the procedure is most useful at levels 4-3 or higher. The analysis can incorporate a wide range of spatial attributes, although data requirements are extensive. The product (map) is easily communicated and interpreted. Applications focus on the risk of ground water contamination, but the procedure could be adapted for surface water.

2.1.2 Risk Based on Buffer Strip Characterization

This approach analyzes the width of an area adjacent to surface waters (riparian zone) which acts as a protective strip to delay, assimilate or filter contaminants in runoff before they enter a lotic system.

A buffer strip is a permanently vegetated area of land situated between the contaminant source areas (i.e., agricultural fields) and the receiving waters. Buffer strips have the potential to reduce contamination from diffuse sources through the retention of nitrogen, phosphorous and pesticides in runoff. Several reviews have assessed the effectiveness of buffer strips as a mechanism to reduce surface water contamination from agriculture (Muscutt et al. 1993. Osborne and Kovacic 1993, Phillips 1989).

Effectiveness (i.e., contaminant retention potential) is directly related to width of the buffer strip, and the strip's environmental attributes and management characteristics. Width can be determined by i) defining a constant distance (i.e., fixed setback from all surface waters), ii) establishing a minimum-variable distance (i.e., beyond a minimum, width varies depending on environmental conditions and land use), or iii) using a variable distance based on environmental and land use factors. Variable buffer widths have been calculated for streams in a rural county of N. Carolina (Xiang 1993). This study incorporates a pollutant detention time model into a GIS framework to analyze effectiveness of buffer strip widths based on soil hydrology, land cover and topography.

Buffer strips are most suitable as indicators of risk for surface water. Risk analysis and assessment could be based on deviation from an optimum buffer width. Various contaminants could be integrated into one indicator by identifying the contaminant with the widest buffer requirement. Like vulnerability mapping, IROWC based on buffer strips is conducive to spatial presentation, which is easily communicated and interpreted. A need for comprehensive data bases suggests that application is most appropriate at level 3, and perhaps 4. A significant shortcoming is that subsurface drainage (tile drains) is not accounted for and could lead to an error in the calculation of risk for high rainfall areas (e.g., eastern Canada).

2.2 BUDGET APPROACHES TO RISK

Budget approaches use input-output analyses to determine the amount of a plant nutrient or a pesticide which is in excess of crop requirements and may be available to contaminate water.

Barry et al. (1993) use this approach to predict nitrogen balances for various crop rotations in southwestern and western Ontario, and two farming systems (cash crop and dairy) in western Ontario. The quantity of surplus nitrogen is adjusted by an average ground water recharge rate to estimate the potential NO_3-N concentration in shallow ground water (assumes an average ground water recharge of 160 mm yr^{-1} for southern Ontario based on a proportion (51.8%) of the long term mean annual discharge of ten agricultural watersheds). For example, estimated nitrate concentrations in ground water under a corn-soybean-winter wheat rotation in southwestern Ontario was 8.5 mg NO_3-N L^{-1} and 22.4 mg NO_3-N L^{-1} in western Ontario for the same cropping rotation, and 58.4 mg L^{-1} for a dairy farm.

A major initiative to determine annual nutrient budgets for N, P and K is underway for several locations in the lower Fraser Valley of British Columbia (B. Zebarth, personal communication, 1995). An annual nutrient balance (applied surplus or deficit) is based on type and number livestock, manure management, inorganic fertilizer applications, crop nutrient requirements, and atmospheric input. Estimates of soil release pathways include surface and ground water. Data on inputs, pathways and outputs are differentiated on a monthly basis.

Budget approaches are also represented in calculations of mass balance to determine the dissipation of pesticides. For example, Frank et al. (1991a, 1991b) have calculated the proportion of applied atrazine, cyanazine and metolachlor which is dissipated by runoff, subsurface tile drainage, leaching to shallow ground water, and degradation in clay loam soils of Ontario (field scale).

Risk is related to the quantity of a potential contaminant present in excess of a desirable or tolerable quantity. Risk is increased if a budget balance shows an excess amount. Unlike indicators which focus on inputs <u>or</u> outputs (e.g., application rates per hectare, yield), a budget approach to IROWC considers the excess amount available for contamination[2]. The above studies by Barry et al. (1993) and Frank et al. (1991a, 1992b) apply a budget approach at the farm and field scales (levels 1 and 2), but it is potentially applicable at higher levels, as demonstrated in the nutrient budget calculations in British Columbia where study areas represent an aggregation of farms. Also, crude nutrient budgets have been estimated at the national level (see Table 9 in the concept paper). Factors which alter the budget balance are readily incorporated (e.g., increasing use of conservation tillage). Notwithstanding the earlier reference to spatial indicators, a budget approach can be integrated with spatial analysis to differentiate areas at risk of surface or ground water contamination from nitrogen or pesticides.

2.3 RISK AS PROBABILITY OF EXCEEDENCE

This approach calculates an estimate of pesticide contamination in water, and expresses risk as a probability of exceeding an established or predefined water quality standard over a defined period of time.

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[2] In areas with an annual deficit balance, unusual events may still contribute to water contamination (e.g., high rainfall event (>50mm) on tile drained soils), but the relative contribution is likely to be considerably less than that of a surplus balance.

Varshney et al. (1993) used this approach in a risk-based evaluation of ground water contamination by atrazine, simazine and alachlor (southeast Iowa). The simulation model RUSTIC provides estimates of concentrations of each herbicide within the aquifer (27-years), from which cumulative frequency distributions of herbicide concentrations at various locations are calculated. The potential risk (R) associated with each herbicide is defined as:

$$R = Pr(C>C_s)$$

where Pr = probability of exceedence, C = predicted daily pesticide concentration in ground water (ppb), and C_s = maximum contaminant level (ppb) (EPA standards for each herbicide). For example, simazine exceeded C_s 35% of the time.

This approach is attractive for IROWC because it explicitly incorporates a water quality standard, although there is no guidance for its selection. The use of a simulation model (RUSTIC) to estimate pesticide contamination is dependent on a long historical record. For situations of insufficient data, alternative probability distributions for estimating the probability of exceedence of contaminant concentrations have been suggested by McBean and Rovers (1992). Even if temporal limitations are overcome, the geographic application of this approach is likely limited to level 2, and maybe 3. Expressing risk as a probability of exceeding a water quality standard is likely to be understood by non-technical people.

2.4 RISK IN RATING INDICES/INDEXES

Rating approaches rank contaminants according to their leaching potential. Ranking schemes for nitrates are based mainly on soil properties, whereas those for pesticides consider the physical and chemical properties of the material, as well as other soil, landscape and environmental characteristics. Land use and management are frequently held constant.

Khakural and Robert (1993) used LEACHM-N as a screening tool to develop soil NO_3 leaching potential (NLP) ratings from soil survey information and representative weather data. Seasonally accumulated values of leached NO_3 (>1.5 m) for a normal precipitation year were used as NLP indices. The ratings were added to the Soil Survey Information System (SSIS) to create NO_3 leaching potential rating maps for three Minnesota counties.

McRae (1991) developed a pesticide rating scheme for Canada which ranked pesticides based on three factors: leaching potential, volume of use, and overall regulatory concern. Pesticides are ranked according to their leaching potential using Gustafson's technique (based on partition coefficient between organic carbon and water, and half-life of the pesticide in soil). Rankings by volume of use refer to kg/annum of active ingredient. Ranking according to overall regulatory concern reflects a pesticide's status in the pesticide re-evaluation program. The scheme does not integrâte the three rankings, but prompts the user to consider each ranking in a sequential manner.

Other pesticide rating schemes also have been developed. For example, Warner (1985) proposed an environmental risk index which assigns a relative value to pesticides based on a tradeoff between its likelihood to contaminate (depending on persistence and mobility), and its toxicity. The index differentiates risk for surface and ground water, and is applied to various agricultural crops in Suffolk County, NY.

Rating approaches primarily focus on properties of contaminants and/or soils. For this reason, their utility for IROWC is limited, unless integrated into, or accompanied by, approaches (e.g., spatial analysis) which consider other IROWC components (e.g., climate, land use, management).

2.5 RISK AND EXPERT SYSTEMS

Expert systems are computer models which help non-experts to find solutions to complex problems. The models require not only quantitative data but also qualitative information in the form of decision rules, interpretation and judgement.

Crowe and Mutch (1994) have developed an expert systems approach (EXPRES) to evaluate the potential of a pesticide to contaminant ground water (see also Crowe 1994). Depending on study objectives, available data and time constraints, EXPRES helps the user to select a pesticide simulation model, prepare an input data set, carry out the assessment, and interpret the results. The model provides a relative assessment of the potential of each pesticide to contaminate ground water, including a quantitative prediction of the pesticide distribution and migration rates (time and depth). The model is applied to a "typical" field of potato, irrigated sugar beet, and corn in PEI, Alberta and Ontario, respectively. (The model contains 22 agricultural regions in Canada).

Although EXPRES is designed primarily as a screening tool for decision making (e.g., pesticide regulation), it has significant potential as an approach to IROWC. Presumably a user's objectives can be defined in terms of risk assessment. The main components of IROWC are incorporated in the model. However, the model's field-specific focus is likely to limit application to levels 1 and 2, although aggregation to level 3 and possibly 4 may be possible for homogeneous conditions at these higher levels. The model requires detailed data sets. While representative data sets are available with the model, ideally these should be generated for each application.

2.6 OVERALL ASSESSMENT

The approaches reviewed here are relevant to the needs of IROWC and generally define some potential methodological approaches. However, with the exception of the budgeting approach, they fail to incorporate some characteristics of IROWC identified as desirable in the concept paper. Specifically, most approaches provide a static assessment which does not incorporate specific farm management and production practices and does not readily allow for temporal analysis.

3. A PROPOSED METHODOLOGY FOR IROWC

Risk of water contamination is based on a concentration of a potential contaminant in water over time. Quite apart from the criteria used to define contamination, concentration depends on two variables:

1. the quantity of a potential contaminant, and
2. the amount of water available to dilute it.

The amount of water available to dilute a potential contaminant is determined by **water partitioning**. At any site, total precipitation is partitioned into different forms. For example, during the growing season, a certain proportion of water is consumed by crops, and another is "lost" by evaporation from the soil surface. The remainder, or **excess water** above soil storage capacity, moves off the site to recharge ground and surface water supplies.

3.1 CONSIDERATIONS FOR DEVELOPMENT OF THE METHODOLOGY

3.1.1 Hydrologic Considerations

In the proposed methodology, excess water plays a fundamental role in determining the level of risk of contamination. A water balance or budget approach can be used to determine the quantity of excess water which is available to transport any residual amount of a contaminant. The basis for this approach is the terrestrial hydrological cycle. In this cycle, the main input (precipitation) is transformed through various processes (infiltration, plant uptake, runoff) into output (evapotranspiration).

Ideally, the water balance of an area is represented by the difference between precipitation (P) and actual evapotranspiration (AE). However, data for AE is frequently lacking. It is possible to determine a crude water balance at any location based on annual or seasonal values of P and potential evapotranspiration (PE). When P exceeds PE, surplus moisture is assumed to exist and water accumulates in the soil until field capacity is reached. Any further excess is available as surface water (i.e., runoff, subsurface tile flow), or ground water recharge. When $P<PE$, a water deficit occurs, and there is no surplus moisture available.

A water balance approach can contribute to the proposed methodology in three ways. First, data on excess water can be distributed over time throughout the year. Seasonal or monthly values of water surplus or deficit can be calculated, and analyzed against the temporal distribution of excess nitrogen or pesticide residue. Second, the variation between years can be assessed. Third, the water balance approach partitions output flow into surface (e.g., runoff, seepage) and ground water (e.g., infiltration, deep percolation) components. Data on the proportion of flow through each pathway is needed to analyze and assess the risk of contamination according to destination.

Agriculture is directly dependent on the hydrological cycle (e.g., soil moisture for crops and drinking water for livestock), but agriculture also alters the cycle, particularly the transfer mechanisms and pathways of flow. For example, tillage practices affect infiltration rates, crop type and planting pattern influences runoff, and land drainage impacts percolation and lateral flow. These impacts become increasingly important at lower spatial and temporal scales. Thus, calculation of excess water at these scales needs to account for the effects of land management on the water balance of an agricultural area.

While the pathways of the hydrologic cycle are reasonably well understood, good quantitative estimates of the magnitude of flows in specific components are very difficult.

3.1.2 Nutrient Considerations

Nitrogen cycles in intensive, arable agriculture are generally characterized by the addition of nitrogen via fertilizer and/or animal manures, and the removal of nitrogen in the harvested crop and cover crops. Other inputs include atmospheric deposition and nitrogen fixation, while losses occur through leaching, denitrification and immobilization. These components of the nitrogen cycle, and their proportionate amounts, are shown for wheat production in Figure 1.

Nitrogen in the form of fertilizer and manures often constitutes a large fraction of the total nitrogen input. Generally, crops respond positively to increased fertilizer and manure inputs, but this response also increases nitrogen removed in the harvested material. Losses via leaching are also usually higher with increasing nitrogen input. As fertilizer nitrogen is increased, the efficiency of nitrogen use is decreased (i.e., percentage of total fertilizer nitrogen taken up by the plant declines). Nitrogen use efficiencies for crops have been broadly estimated at 60-50% (e.g., Briggs and Courtney 1989, Juergens-Gschwind 1989). This means that a considerable quantity of nitrogen not taken up by plants is potentially available for leaching or other dissipation pathways of the nitrogen cycle.

3.1.3 Pesticide Considerations

An ideal approach to analyzing the risk of pesticide contamination first determines the quantity of a pesticide, and its chemical derivatives, in the soil, and then ascertains the susceptibility of these chemicals to various transport and transformation processes (Mackay 1992). Following application, a pesticide is partitioned into various phases (e.g., vapour in soil air, sorbed by organic matter, solution in soil water). Of particular concern is the amount partitioned in mobile soil water, which represents the quantity potentially available for leaching or surface runoff. The proportion directed toward leaching can be relatively high for herbicides such as atrazine (Table 1). The risk of surface transport (in solution or attached to eroding soil particles) is especially high if rainfall closely follows pesticide application.

Quantitative models of chemical fate in soil are available to estimate the amount of a pesticide partitioned in soil water (e.g., PRZM, LEACHM). These models are generally site-specific and applicable only at lower hierarchical levels (levels 3 - 1). Their data demands are extensive.

Table 1 Fate of Atrazine in Soil

Loss By:	%
reaction	31.4
leaching	68.0
evaporation to air	0.6

Source: data from Mackay (1991)

At higher levels (e.g., levels 5-3), information is available on pesticide inputs (e.g., recommended application rates by crop, surveys of pesticide use), but understanding of and data on partitioning, transformation and transport are lacking. For this reason, the relative indicator of risk of pesticide contamination at level 5 considers pesticide input data only.

3.1.4 Key Questions

Based on the methodological considerations discussed above, three questions for investigation can be posed:

1. **is there sufficient excess water on average to move contaminants (vertically or laterally) to water sources (i.e., is there a risk?)**

   This question in reality deals not only with the amount of excess water, but also the proximity of the rooting zone to water resources (surface or ground). It can be rephrased as "what is the travel time for potential water contaminants to move to water resources?" It is only if the answer to this question is yes or the travel time is short enough to be of concern that the remaining two questions are important. This is not to suggest that the movement of contaminants in water out of the active rooting zone is not of concern. It is simply to suggest that under these circumstances, it does not constitute a risk of water contamination. This accumulation of a contaminant under dry conditions over the long term resembles the condition of geologic nitrate described by Follett and Walker (1989).

2. **what kinds of potential contaminants are present and in what amounts?**

   At a detailed level, the answer to this question is quite complex and includes sources such as atmospheric deposition, release from forms stored in soil and wetland components as well as the incremental inputs used in the course of land husbandry. In this discussion, the indicator is focused on the specific practices associated with agricultural land use. It may be thought of as a partial budgeting approach. In general then, the kinds and amounts of potential contaminants are directly related to the kind of crop grown and the type of agricultural system. From the standpoint of nutrients, there are three general sources; namely, commercial fertilizer, decomposing crop residues and animal manures. The amounts can be inferred from the current crop production recommendations, the amounts harvested in crop yields, and survey information about the amounts produced or sold. For pesticides, the kinds and amounts can be inferred from the types of crops, current recommended herbicide usage and surveys of actual use.

3. **what is the quantity of water in excess of that used for crop production and evaporation? and, what is the fate of the excess water?**

   These are probably the most difficult and the most critical questions because they control the extent and direction of movement and the quantity of water determines the dilution factor for all potential contaminants.

3.2 ASSUMPTIONS

At all but the lowest levels of the hierarchy, reasonable starting assumptions for agricultural areas include:

1. an approximate balance between nutrient additions and removals within the limits of nutrient use efficiency.

2. an indicator which estimates the impact of agricultural (anthropogenic) practices can be calculated using a **partial** budgeting approach which considers those inputs and outputs directly influenced by management such as the application of fertilizers, manures and pesticides, and the removal of harvested grain and stover.

3. trends in pesticide use can be estimated from i) a knowledge of the crops grown and the recommended pesticides associated with their production, and ii) the general level of pesticide usage (e.g., county level surveys).

4. for "average" conditions of soil, topography and drainage it is possible to estimate IROWC by crop (e.g., compared to non-crop or unmanaged conditions).

5. at lower levels of the hierarchy it is possible to improve the IROWC estimates by incorporating actual conditions of soil texture, topography, drainage etc and more detailed climatic data.

6. it is necessary to estimate the extent to which "friendly" practices (e.g., conservation tillage, buffer strips) reduce the IROWC from that observed under "average" conditions.

3.3 A PROPOSED BUDGET MODEL FOR IROWC

This section describes a model for IROWC based on a budget approach. The model is viewed as a component of a broader methodological approach to IROWC, which will integrate the budget approach with other tools (e.g., GIS) and approaches (e.g., spatial analysis of risk). The essential function of the model is to compare the flow-weighted mean concentration of each potential contaminant to an appropriate standard (e.g. drinking water). The risk of water contamination from non-crop and unmanaged sources is acknowledged, but the approach considers only the **added risk** of water contamination resulting from agricultural activities.

The **Potential Contaminant Concentration (PCC)** consists of two parts: i) determination of the Potential Contaminant Present (**PCP**) in mg/ha, and ii) calculation of Excess Water (**EW**) in l/ha, representing the available quantity of surplus water. Thus:

$$PCC_c = \frac{PCP\ (mg/ha)}{EW\ (l/ha)} \qquad 3.3\ (1)$$

c = contaminant type (e.g., nitrate, atrazine)
PCP = f(climate, crop, input rates, soil, management, etc.)
EW = f(precipitation, evaporation, crop moisture use (yield), soil).

At each level of the hierarchy, the differentiating characteristics and associated factors are incorporated into the determination of the amount or potential contaminant present (PCP). For example:

PCP_5 = f(crop type, crop area, nitrogen in fertilizer and manure, nitrogen in harvested crop, herbicide class (e.g., triazine), annual climate normals). The calculated value is a **relative** indication where the non-crop or unmanaged state is 0 and risk increases in proportion to nitrogen content of the harvested crops, and level of pesticide use.

PCP_3 = f(all information and constraints from higher levels plus crop rotations, soil texture, subsurface drainage, conservation practices, daily and monthly climate, post-harvest precipitation, specific pesticide, etc.). The calculated value is the **concentration** in mg/ha for the specific contaminant.

where 5 and 3 correspond to levels of the hierarchy. Excess water is determined by estimating the flows in the hydrological cycle in increasing detail at lower levels.

IROWC is characterized as the ratio of the average PCC to the concentration allowable to maintain a desired standard (e.g., drinking water). Thus:

$$IROWC_c = \frac{PCC_c \text{ mg/l}}{\text{Maximum Allowable concentration (mg/l)}_c} \qquad 3.3\ (2)$$

$$IROWC_{Composite} = \sum_{c=1}^{n} IROWC_c \qquad 3.3\ (3)$$

Ideally, this calculation would be carried out for each (precipitation) event which constitutes a risk for water contamination. In practice, this will only be possible at the most detailed level of assessment. The hierarchial levels at which the indicator is required (see Table 1 in the concept paper) determine the kinds of data available and the detail and precision which can be achieved.

In hierarchical fashion, we can start at the highest level and determine the constraints, and then consider lower levels within these. For example, it is feasible to characterize IROWC at the ecoregion or ecodistrict level (levels 4-5). The proportion of the ecoregion or ecodistrict occupied by various crops can be determined. At these levels, only a **relative** indicator based on annual average conditions can be estimated. It is sensitive to changes in crop type, crop area, livestock density and monthly climate. It is not feasible to incorporate "friendly" practices because it is not possible to associate them with specific crops (see Table 1 in the concept paper).

At lower levels (levels 1-3), the indicator will more nearly reflect **actual** conditions. It will require many more types of data in much greater detail to incorporate spatial variability in soil, topography and drainage; the degree of adoption of friendly practices and more specific details of climate. Temporal sensitivity is also better at lower levels allowing for calculations of risk on a monthly or event basis. With the calculation of IROWC on the basis of soil units (e.g. SLC for level 3, detailed maps for level 2 and on-site soil characterization for level 1), it should also be possible to partition the risk of contamination into surface or subsurface water and possibly, tile drainage water. In addition, with movement from highest to lowest levels, the probability of coincident occurrence (the certainty that the specific crop is grown on a known soil type using particular tillage practices, etc) increases, thereby enhancing the level of confidence for the estimate.

4. EXAMPLE APPLICATIONS OF THE PROPOSED METHODOLOGY

The following sections have been prepared at the request of the indicator team to demonstrate how the proposed methodology would characterize the risk of water contamination. A full implementation of the proposed methodology is a substantial undertaking requiring consultation and input from many experts from across Canada and internationally. In addition, it is to be expected that, as details of the methodology are worked out, there will be gaps in our knowledge and understanding which require original research, and gaps in the available data which will require alternative data sources or data collection activities. This could include coupling other agri-environmental indicators with IROWC, or direct incorporation of their data.

Consequently, THE SAMPLE **IROWC** CALCULATIONS SHOWN HERE ARE A DEMONSTRATION OF THE METHODOLOGY ONLY. THEY HAVE NOT BEEN EXTENSIVELY CHECKED AND SHOULD NOT BE TREATED AS RELIABLE.

These examples do however provide a demonstration of the kinds of information which are required, the kinds of resolution which can be achieved and, very clearly, the levels of effort and detail which are needed to calculate something approaching ACTUAL indicators of risk of water contamination.

4.1 IROWC AS A RELATIVE MEASURE - BROAD LEVEL INTERPRETATION

Ideally, the proposed methodology would be implemented at all hierarchial levels. In reality, it is only possible to determine all parameters required at the lower levels. At the higher levels only a relative IROWC rating is possible based on the principles outlined; namely, an estimate of the amount of potential contaminant(s) available, the amount of excess water and the amount which could be accommodated within the standards of concern.

For the broad level assessments (levels 4 and 5), the IROWC is based on current land use practices which give an indication of the level and kinds of inputs and climate records which provide a rough estimate of the quantity of water in excess of crop requirements. These results could be presented at the level of ecozone, ecoregion or ecodistrict. The difference would be in the degree of resolution of the classes and the extent to which specific land management practices (such as conservation tillage, buffer strips, etc) could be considered. In addition, for the ecodistricts it would be possible to look at climate patterns for portions of the year as well as the degree of variation between years.

For this demonstration example, the results are presented at the ecodistrict (ED) level. This level and all larger spatial units are so large that the land use and management practices cannot be associated with specific soils, topography and drainage systems. Consequently, it is not possible to partition the IROWC between surface and ground water. Variations in specific land management practices, or variation throughout the year or between years, can be incorporated but are not illustrated in these examples.

The moisture conditions were estimated from climatic normals data for 1951-80. These data were extracted from the Land Potential Data Base (Kirkwood et al., 1983) and associated with ecodistrict (ED) and SLC polygons by taking the largest area of overlap.

In keeping with the first question posed (section 3.1.4) the relative annual excess water (REW) or deficit was calculated from the annual precipitation (P) and the potential evaporation (PE), using 1951-80 climatic normals data, as:

$$REW = \text{Annual } (P - PE) \qquad 4.1\ (1)$$

The estimate of REW was determined for the prairie region and southern Ontario (Figure 2). In the prairies, all but the fringe areas are characterized by REW less than zero. While there will be some movement of water to surface recharge areas at specific times of the year and also some movement to ground water, at a broad level this movement is negligible. This is consistent with the general landscape patterns of the great plains where most of the drainage is internalized into prairie sloughs; the water tables are quite deep with little connection to surface recharge areas and there are few external drainage channels. This assessment is consistent with the observations of de Jong and Kachanoski (1987). For further demonstrations of the proposed methodology, Southern Ontario is used as the sample region.

It is recognized that this approach to calculating annual excess water is not without problems. Actual evapotranspiration (AE) is only a fraction of PE during most of the day in regions such as the prairies. This means that the above approach may underestimate annual excess water. However, the approach is maintained because it is difficult to obtain data for AE at broad scales. A related problem is basing spatial screening on a single criterion (i.e., REW). In the prairie region, it is possible to have both a small REW and a large potential contaminant concentration (PCC). The criterion of EW is maintained over concentration because the fundamental matter of concern is water transport of the contaminant. Again, this does not imply that the steady accumulation of a contaminant under dry conditions and over the long term (e.g., geologic nitrate) is not important, but rather that its potential for transport to surface or ground water is significantly reduced.

Ideally, PCC should incorporate variation in nutrient uptake efficiency and time of uptake for each crop, as well as the temporal variability of excess water. Thus:

$$PCC = \frac{f(\text{crop type, nutrient uptake efficiency, time of uptake})}{\text{excess water } f(\text{time period}) \times \text{crop area}} \qquad 4.1\ (2)$$

Data constraints and the comprehensiveness of the analysis resulted in a simplified version of this equation for the purposes of this demonstration. This analysis used mean annual values for P and PE (1951-1980 climatic normals) to estimate REW, and a mean nutrient uptake efficiency of 60% for all crops over the entire growing season.

Based on the above rationale, the following series of equations were used to estimate the broad level IROWC for Southern Ontario:

$$RPCC_N = \frac{\sum_{ED} \text{Crop Area}_{\text{by crop}} \times \text{Nitrogen Harvested}_{\text{by crop}}^{3}}{\text{REW} \times \text{total crop area}} \qquad 4.1\ (3)$$

$$RPCC_M = \frac{\sum_{ED} \text{Nitrogen in Manure} \times 0.1}{\text{REW} \times \text{total crop area}} \qquad 4.1\ (4)$$

$$RPCC_H = \frac{\sum_{ED} \text{Crop Area}_{\text{by crop}} \times \text{Average rate for herbicides}_{\text{by crop}}}{\text{REW} \times \text{total crop area}} \qquad 4.1\ (5)$$

$$RPCC_{Ha} = RPCC_H \times \frac{(P\text{-}PE)_{May} + (P\text{-}PE)_{June} \times 0.5 + (P\text{-}PE)_{July} \times 0.25}{(\text{Annual Precipitation - PE from 1951-80 climatic normals})} \qquad 4.1\ (6)$$

RPCC = relative potential contaminant concentration for nitrogen in the harvested crop (N), long-term nitrogen from manure (M), herbicide (H), and herbicide concentration adjusted for the susceptible time (month) for transport (Ha). The herbicides atrazine and simazine are used for demonstration purposes.

$$IROWC_{(\text{level 4-5})} = (RPCC_N + RPCC_M)/10 + RPCC_{Ha}/0.01 \qquad 4.1\ (7)$$

The IROWC provides a measure of the relative amount by which the current standards are exceeded. The current drinking water standard for nitrogen is 10 mg/l and the standard for atrazine is 0.005 mg/l, and for simazine it is 0.01 mg/l. The latter standard is used to calculate IROWC for herbicides.

Three additional considerations were included in the these equations:

1. Data on nitrogen use efficiency plotted against proportion of optimum yield (Sander et al. 1994) shows an efficiency of 60-40% for corn. Data from Europe (Juergens-Gschwind 1989) also suggests a nitrogen use efficiency of 60% and further indicates the other sink terms for the additional nitrogen (e.g. denitrification to GHG, soil biomass, transport off-site by water). This implies that the quantity available for other

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[3] The calculation determines the average quantity of nitrogen harvested per hectare of cropped land in an ecodistrict based on the following levels of nitrogen in the harvested crop: Corn, grain and silage - 175 kg/ha; soybeans and white beans - 150 kg/ha; barley - 45 kg/ha; spring wheat - 55 kg/ha; winter wheat - 85 kg/ha; fall and spring rye and oats - 35 kg/ha; alfalfa and other tame hay - 200 kg/ha.

pathways is approximately equivalent to the Nitrogen Harvested (equation 3). This efficiency is assumed for all crops.

2. OMAF Publication 296 indicates the long-term value of manure. Generally, 50% of the total manure nitrogen (75% for poultry) applied in the spring is available in the year of application. In the second year approximately 10% of the remaining nitrogen becomes available, 5% in the third year. For this calculation, the nitrogen during the year of application is included in the estimate based on harvested crop. For subsequent years a factor of 10% is used to account for the long-term value and also for the nitrogen lost from manure applied (equation 4).

3. For herbicides, equation 6 is adjusted by the proportion of the excess moisture in May, June and July as these are the times the herbicide will be most susceptible to transport. These monthly figures are further weighed (1.0, 0.5 and 0.25 respectively) to account for declining susceptibility to movement off-site. After this time it is assumed to have been degraded by natural processes. Atrazine and simazine are aggregated as triazines, and specific properties of individual herbicides (e.g., solubility, half-life, partitioning coefficient) are not considered at levels 4-5.

Key features demonstrated in the example application at the level 4-5 assessment are trends and one option for calculating a composite indicator.

4.1.1 Temporal variation

Figure 3 shows the change between 1991 and 1981 in relative potential contaminant concentration for nitrogen (based on levels harvested in the crop) for crop distribution as reported in the census of agriculture. These results are reported in units of mg/l or ppm with the range going from a decrease of 5 or less to an increase of 5 or more. In general these results are quite encouraging for much of the province showing no change or a slight decline. The greatest increase is indicated in the ecodistrict which occupies north Middlesex and the western portion of Huron Counties. In addition, there is some indication of an increase in Eastern Ontario and Essex, Lambton counties and the fringe area. The map illustrates that this kind of calculation can indicate some changes which are potentially of the same order of magnitude as the drinking water standard.

4.1.2 Composite indicator

The ratio of the relative potential contaminant concentration (RPCC) to the maximum concentration allowed by current standards provides an indication of risk. If all water quality standards have the same level of importance (or risk if exceeded) then it is possible to provide a composite indication of risk by combining the risks for all separate potential contaminant components. This aspect is described by equation 7 and illustrated for Figures 4a, 4b and 4c.

Figure 4a shows the relative IROWC associated with nitrogen from crop production and also the residual from previous manure applications. Figure 4b shows the relative IROWC associated with triazine applications. Figure 4c shows the results of the combination of these component IROWC estimates.

4.2 IROWC AS AN ACTUAL MEASURE - ASSESSMENTS AT LOWER LEVELS (3-1)

As we move down the hierarchy, the resolution and accuracy of the IROWC should improve. In addition to the dominant factors affecting contamination at each higher level in the hierarchy, an assessment of risk at lower levels incorporates greater detail through refinement of parameters such as climate (e.g., monthly values of P-PE (or P-AE) to identify months when off-site movement is most likely), nutrient inputs (e.g., nitrogen uptake pattern of specific crops over the growing season to determine when surplus nitrogen is most likely available), tillage practices (e.g., conventional, conservation, no-till) and their effect on excess water and surplus nutrients, and processes such as subsurface tile drainage to further partition surface and ground water components.

An assessment of IROWC at lower levels incorporates these and other detailed parameters in order to estimate an actual concentration of a contaminant in water. In most cases, this will result in requirements for additional data and lead to increasingly complex analyses. It was not possible to develop a complete example demonstration of IROWC at lower levels. The following sections provide OUTLINES of how COMPONENTS of IROWC are assessed at lower levels. They also DEMONSTRATE the kinds of data which must be synthesized. The results are closer to ACTUAL <u>BUT THEY ARE STILL RELATIVE.</u>

Several attributes of IROWC are selected to demonstrate the type of parameters to be considered at lower levels. The geographic focus is on watersheds in south-western Ontario where flows and sediment yield have been monitored and, one small watershed in particular, Bamberg Creek.

4.2.1 Bamberg Creek Watershed

This watershed is a catchment located in Wilmot and Wellesley Townships in the Regional Municipality of Waterloo. Agriculture occupies about 81% of the watershed's land area in a mix of farm types (e.g., cash crop, livestock, mixed farming). A recent study of the watershed was undertaken to assess the state of the agricultural land base relative to its contributions to non-point source pollution (Ecologistics Limited 1993). The study integrated available information on soils, land use and farm inputs (fertilizers, manures and pesticides) into a GIS framework to produce maps (1:50,000) of nutrient and pesticide loading, and ground water contamination potential. The approach used approximates the preliminary steps of a budget approach. Furthermore, data on soils, land use, and inputs of nitrogen and pesticides used in this study indicate the information base available for a level 3 assessment.

Nitrogen inputs were estimated using fertilizer nitrogen recommendations for the various field crops grown. These were converted to a unit area loading for each land use category (assuming different proportions of specific crops in various land use systems) and weighted by the proportion of each land use assumed to have received fertilizer. Estimates of nitrogen from manure were based on animal units (census data), manure production per animal unit, and average nitrogen concentration of manure. These values were adjusted to account for an estimated nitrogen loss of 50% during storage, handling and application. Areal proportions of manure application per land use system were assumed to determine unit area loadings. The nitrogen values for fertilizer and manure, and total nitrogen, are shown by land use in Table 2. These data were used to produce a map of nitrogen loading (Figure 5).

Using 1986 census information, and data from the 1988 Ontario pesticide survey, the same basic approach was used to estimate pesticide loadings by land use category. Pesticides were not differentiated by type in this study. Crop groups were defined (row crops, small grains, hay and improved pasture) and assumptions made to estimate unit area pesticide loading for each group and distributed over the land use categories. Calculated pesticide loadings by land use category are shown in Table 2.

Finally, a map of soil transmissivity (i.e., ability of surficial soil (<1m) to transmit pollutants) was superimposed onto maps of nitrogen and pesticide loading to display the relative potential for ground water contamination from these contaminants.

4.2.2 Extending the Research

The focus of the Bamberg Creek study was on characterizing the amounts of potential contaminants present. From the standpoint of an IROWC, there are additional data and information requirements. The analysis presented results on an annual basis with no attempt to deal with changing conditions throughout the year. There was no attempt to develop complete budgets for nutrients or pesticides, and the complex and difficult task of partitioning water between surface, tile flow and ground water was not addressed. There are no easy ways of dealing with these concerns but the following sections indicate some possible approaches and data sources which can be used.

1. Temporal Distribution of Surplus Nitrogen and Excess Water. Nitrogen is present in the soil throughout the year and is normally added as fertilizer early in the growing season. The uptake of nitrogen by plants increases over the growing season. The pattern of nitrogen uptake can be analyzed in relation to surplus moisture distribution to determine the amount of mobile nitrogen in the soil profile.

Ontario data on plant uptake of nitrogen is available for some crops (M. Miller, personal communication). The quantity of nitrogen taken up by corn (grain and stover) is illustrated in Figure 6. Uptake commences after the first month of growth, increasing steadily between 32-

Table 2 Nitrogen and Pesticide Loading by Land Use in Bamberg Creek Watershed

Land use	Fertilizer N	Manure N	Total N	Pesticides
		- kg N ha^{-1} -		(kg a.i. ha^{-1})
Continuous crop	226	92	318	1.60
Corn system	190	133	323	1.08
Mixed system	165	162	327	0.73
Grain system	102	39	141	0.40
Hay system	126	342	468	0.33
Pasture system	54	235	289	0.10
Grazing system	0	169	169	

source: data from Ecologistics Limited (1993)

112 days after planting (DAP), and levelling off somewhat following silking (112 DAP). This pattern suggests that all of the applied nitrogen is surplus during the early stage of plant growth. During subsequent growth stages, nitrogen uptake increases steadily (i.e., the amount of surplus nitrogen decreases). Over the entire growing season, total nitrogen uptake is approximately 200 kg ha^{-1}.

The temporal distribution of nitrogen uptake can be compared to the seasonal or monthly distribution of surplus moisture in order to estimate the quantity of nitrogen at risk of leaching. An actual estimate can be derived using data on monthly or daily climatic normals (e.g., precipitation, potential evapotranspiration) for a representative climate station. An example for the Bamberg Creek watershed using climate data from Elora Research Station is shown in Figure 6. There is a mean annual water surplus of 367 mm, but the distribution is such that a water deficit exists during the summer months (June - September). This suggests little risk of nitrogen movement during the growing season. Even 30 days after planting, when nitrogen uptake is nil (i.e., surplus nitrogen is high), the risk is low because the onset of water deficit coincides with planting of corn and fertilizer application (i.e., mid-late May). Of course individual rainfall events may contribute to movement of surplus nitrogen throughout the growing season, as well as residual nitrogen in the soil at post-harvest. This would require further detailed analyses using daily climate daily.

Crop uptake is an important, but incomplete, measure of recovery of applied nitrogen. Nitrogen is also partitioned in crop residue (surface stubble, roots) and in the soil as residual nitrogen. Like crop uptake, these also vary by crop type and rate of nitrogen application. It will be necessary to incorporate the changes in nutrients and water throughout the year to produce a level 3 IROWC.

2. The use of Computer Simulation Models to Characterize the Dynamics of specific Pesticides in Soil Water.

Another way to incorporate the approach and results of Bamberg Creek watershed study into an assessment of IROWC at level 3 is to determine the fate of specific pesticides using models which estimate residual pesticide movement out of the root zone. The study aggregates data on pesticide use but does not differentiate pesticides by type or rate of application. Pesticides vary in their physical and chemical properties, and in their soil-water interactions. An analysis which considers various chemical transformations and partitioning of pesticide residue into differing sinks is best approached by use of a pesticide fate model (e.g., LEACHM, GLEAMS, PRZM). The spatial information on pesticide loading in Bamberg Creek watershed could be used to target specific sites (field, farm) for more detailed risk analyses of pesticide movement.

In addition to site-specific applications, it is also possible to apply pesticide fate models at the watershed scale. A recent study by de Jong et al. (1994) developed and applied a methodology to investigate migration of atrazine through the soil profiles (<90 cm) of the Grand River watershed. This watershed contains the Bamberg Creek catchment. The study developed an integrated methodological framework consisting of 1) modified submodels of LEACHM to describe soil water flow (LEACHW) and atrazine movement (LEACHP), 2) pedotransfer functions based on soil texture and organic carbon content, 3) geostatistical analyses to extend soil and climate data, and model output, from point to areal format, and 4) GIS to produce maps of atrazine loading in the Grand River watershed (1:1,000,000).

Results of this study include model predictions of annual atrazine loading over time for several soils (Figure 7). Loadings vary by soil type, but tend to stabilize over time (after 5-8 years) for all soils. Two of the soil types are found in the Bamberg Creek watershed (Huron clay, Fox sand). Atrazine movement in the Huron soil is characterized by an incremental increase in loading, stabilizing after year three. Presumably, atrazine applied to this soil is readily degraded or its movement out of the soil profile is impeded (de Jong 1994). The Fox sand is distinguished by a relatively high atrazine loading, posing a greater risk of water contamination.

Seasonal distribution of atrazine loading is not known since model simulations were undertaken on an annual basis. Other factors (e.g., information on land use, crop rotations, and changes in water table depth) were excluded in the analysis, but the methodology has considerable potential for analyzing the risk of pesticide contamination of ground water, and for projecting the impact of changes in land use and pesticide management at the watershed scale.

3. Partition Excess Water into Surface Tile Drainage and Ground Water

A final suggestion for building on the Bamberg Creek watershed study to calculate a level 3 IROWC assessment is to partition the excess water component into surface or ground water. Ultimately, detailed analyses to quantify hydrologic pathways (e.g., runoff, base flow, deep percolation) is best undertaken by hydrology models and use of long term data on climate, stream flow, etc. However, it may be possible to use other data to approximate the water partitioning. These alternative data sources include:

- estimates based on stream flow to quantify amount of water available to move contaminants,
- analysis of land resource and management factors which influence moisture partitioning, and
- output from the hydrology submodel of models already used to study sediment, nutrient or pesticide movement.

The Monitoring and Systems Branch, Ontario Region, Environment Canada, collects data on stream flow and sediment levels for a number of watersheds in south-western Ontario, including the watershed where Bamberg Creek is located. If we assume that the ground water system is approaching equilibrium then stream flow will approximate the excess water available. This measurement does not provide information on the portion of the water which is surface runoff or the portion which has moved through the soil. Figure 8 shows the location of the 9 watersheds where stream flow data were available. Table 3 compares the excess water as determined from climatic normals data to the 1972-1991 mean annual stream flow. Values of the stream flow estimate suggest that the actual quantity of excess water is larger and more variable than the estimate based on climate. These results are to be expected because:

- the climate data has been generalized over relatively large areas, and
- actual evapotranspiration is always less than potential.

The partitioning of water between the soil surface and subsurface is determined by many factors. Rigorous estimates are quite difficult to make. There are, however, a number of factors of soil, landscape and management which can be used to provide a more qualitative assessment. The soil polygons shown on Figure 8 partition the watersheds into different areas depending on the land resource properties. Characteristics such as texture, slope and landform have a strong influence on water pathways. Sandy textured soils are more permeable and allow more flow through the soil and less runoff. Areas of greater slope steepness and more dissected landform are conducive to higher proportions of runoff. These factors are summarized in Table 4 for those soil polygons which are more than 50% within watershed boundaries.

Other factors which influence the water partitioning are the proximity to natural or artificial watercourses. The stream network shown in Figure 8 was obtained from Environment Canada. An analysis was done to determine the proportion of land adjacent to streams. Land which was within 200 meters of a stream was considered to be in close proximity. The rationale for the 200 meter buffer was that field boundaries tend to impede surface runoff and the traditional dimensions for fields in Ontario are 40 rods square or 200 x 200 meters. The stream proximity was calculated for each soil landscape polygon and, as shown in Table 4, it varied substantially across the watershed area.

Table 3 Comparison of Annual P-PE and Annual Stream Flow in Selected Watersheds

Watershed	Station No.	P-PE (mm)	steam flow (mm)
1	02GB001	220 - 320	384
2	02GE003	220 - 250	422
3	02GG007	160 - 320	323
4	02GC002	220 - 230	362
5	02GC018	220 - 230	391
6	02GC026	220 - 230	449
7	02GC021	220 - 230	512
8	02GC007	220 - 230	398
9	02GB007	220 - 230	312

Data Sources: 1)P-PE based on 1951-80 climatic normals (Kirkwood et al, 1983).
2)Runoff based on 1972-91 observation of hydrologic stations, Monitoring and System Branch, Environment Canada.

One of the more difficult aspects of water partitioning and, consequently, the destination of potential contaminants is flow through artificial subsurface drains. Water moving through this pathway will leach materials from within the soil profile but then is shunted to surface waters rather than contributing to base flow or ground water recharge. While the soil taxonomy and natural drainage state provide some basic indications of presence or absence of tile drains, maps of actual tile installations must be used to provide an realistic estimate of the proportion of drained land. Maps of tile drainage locations were available for portions of the watersheds in Oxford, Middlesex and Elgin counties. Table 4 shows these estimates where data were available and illustrates the substantial degree of variation to be expected.

These factors illustrate the complexity of factors associated with the task of partitioning water flows in the environment. Generally, models will be required to provide a realistic integration. These models may be primarily for the hydrologic cycle or they may model nutrient and pesticide flows in addition to water. For example, some hydrologic data should be available for the Bamberg Creek watershed because the CREAMS model generally partitions runoff and infiltration (using the SCS runoff curve number), and uses a water balance routing to account for storage in soil water, uptake by plants, and deep drainage.

Accurately partitioning excess water into surface and ground water components is a challenging hydrological task and an important area of further work.

Table 4 Some Land Resource and Management Factors which Affect the Partitioning of Excess Water for Selected Watersheds of Southern Ontario (data compiled at SLC level)

Water shed	SLC poly #	Surface Texture	Slope (%)	Local Landform	Stream Proximity*	Prop. of Land tiled
1	0078	Silt loam	1 - 3	Level	10.6	n/a
	0073	Clay loam	1 - 3	Undulating	8.0	n/a
	0452	Clay loam	1 - 3	Undulating	1.3	n/a
	0066	Clay loam	1 - 3	Level	5.6	n/a
	0153	Loam	1 - 3	Undulating	4.8	n/a
	0077	Loam	4 - 9	Undulating	3.6	n/a
	0036	Loam	4 - 9	Rolling	7.4	15.8
	0069	Loam	4 - 9	Undulating	6.6	n/a
	0070	Loam	10 - 15	Hummocky	5.4	n/a
	0471	Loamy sand	1 - 3	Undulating	8.5	n/a
	0470	Loamy sand	1 - 3	Undulating	7.7	n/a
	0049	Loamy sand	1 - 3	Undulating	12.2	4.5
	0067	Sandy loam	1 - 3	Undulating	5.1	n/a
	0068	Sandy loam	4 - 9	Rolling	5.3	n/a
2	0035	Silt loam	4 - 9	Undulating	5.0	47.7
	0028	Silt loam	4 - 9	Rolling	12.0	51.2
	0013	Silt loam	1 - 3	Undulating	9.0	26.2
	0038	Clay loam	1 - 3	Undulating	8.9	36.0
	0029	Loam	1 - 3	Undulating	7.9	37.3
	0034	Loam	4 - 9	Rolling	6.3	40.2
3	0026	Silt loam	1 - 3	Undulating	8.7	38.1
	0024	Clay loam	1 - 3	Level	3.8	44.9
	0020	Sand	1 - 3	Undulating	6.8	4.9
	0025	Sand	1 - 3	Undulating	5.7	19.2
6	0424	Silty clay loam	4 - 9	Undulating	13.3	17.7
	0423	Fine sand	1 - 3	Undulating	6.6	8.3
7	0422	Loamy fine sand	1 - 3	Undulating	11.4	n/a
9	0425	Clay loam	4 - 9	Ridged	0.0	15.5
	0145	Loam	1 - 3	Undulating	6.4	n/a
	0031	Loamy sand	1 - 3	Undulating	7.7	8.5

* proportion of land (%) within 200 meters (40 rods) of streams by SLC polygon

4.2.3 IROWC at the level of the SLC polygon

The discussions of data for a level 3 calculation of IROWC have dealt with watersheds, SLC polygons as shown in Figure 8 and small catchment areas. As the size of the spatial unit gets smaller, it becomes increasingly difficult to assemble comprehensive consistent data over large areas. For example, it was possible to compile census of agriculture data on crop distribution for SLC polygons. Other data such as climate, pesticide usage and farming systems cannot be directly related to SLC polygons. The climate station network is too sparse to be interpolated with precision. Pesticide usage is recorded on a substantially larger units (counties) in classes. Farming systems can be compiled from census but requirements for confidentiality preclude this compilation at the SLC level.

Based on the crop distribution, the relative potential contaminant concentration for nitrogen based on crop uptake was calculated for the watershed area. Figure 9 shows the results of this calculation for 1981 and 1991. This figure shows a substantial degree of change over the time period and also shows the increasing complexity which results as the spatial units become smaller (i.e., approaching an actual IROWC).

5. POTENTIAL PILOT AREAS FOR DETAILED IMPLEMENTATION OF IROWC

The proposed methodology for IROWC must be implemented and tested using accepted scientific principles and procedures. As it is implemented it will be important to review and refine the data and scientific basis to ensure the best result. Before IROWC can be accepted it must be validated. Testing must continue on an ongoing basis to ensure accuracy and relevance.

At the general level (4-5), IROWC results will be validated by comparison to data from the limited monitoring and experimental sites across Canada. As the IROWC methodology is developed, refined and finalized it will be appropriate to select one research group, with support from a team of technical experts, to assemble the data and implement the level 4-5 IROWC. Furthermore, the IROWC level 4-5 results will be compared to the IROWC calculated at more detailed watershed and farm levels.

The lower level (2-3) IROWC sites become important locations for development of more accurate indicators and also as comparison sites for the general level 4-5 IROWC. Their selection should be based on a variety of factors including:

- locations representative of the soils, land use and management and geographical areas of agriculture in Canada
- sites where data is available and collected on an ongoing basis to calculate the moisture balance and the budget of nutrients and chemicals in the environment
- sites where past and ongoing monitoring activities are measuring the parameters necessary to test moisture and budgeting models.

- in light of current resource constraints, it will likely be necessary to incorporate the detailed implementation of IROWC into closely related research projects which are already in progress.

The following is a list of some potential areas for detailed IROWC implementation:

1. the Lower Fraser River valley of B.C. where a major initiative is underway to estimate detailed nutrient budgets for specific agricultural locations. A focal point is the Abbotsford aquifer where a substantial project has been carried out to model ground water risks from intensive livestock-based agriculture.

2. the Lethbridge research station where projects relate to both dryland and irrigated agriculture and monitoring of nutrient and pesticide dynamics.

3. in Ontario, there are three small paired watersheds (Essex, Kettle and Kintore) where data collection and monitoring activities have been carried out for the past 10 years. At the Essex site and the adjacent Woodslee Research substation detailed hydrological measurements are being taken. At the Kintore site, there is a large project currently under way to characterize water movement and associated transport of chemicals.

4. in Eastern Canada there are a series of small watersheds which have been extensively characterized and monitored in association with the International Hydrological decade. Since characterization of the hydrological cycle is critical to the success of the IROWC methodology, it may be advantageous to use some of these catchment areas.

In addition to the above, a survey is underway to collect information about a network of Canadian watersheds (CANWANET) which have studies of relevance to IROWC (contact Jacques Millette). Most of the studies are at level 2. See attached sample.

6. ASSOCIATED RESEARCH AND COLLABORATION

Implementation, testing and validation of the proposed IROWC methodology is best undertaken by building on, and coordinating with, related research which is in progress or planned. This section briefly identifies several associated research activities which should contribute to further development and application of the IROWC methodology, and also suggests a general strategy of collaboration among research participants.

Research projects which have potential association with IROWC at the University of Guelph are listed below. This is a partial and incomplete list. Many other research projects are active at other universities and research institutions across Canada, which should also be considered for their contribution to the methodological development of IROWC. The projects listed below are selected and categorized relative to three broad components of IROWC which require further research and detailed data. The scale of implementation is also shown.

1. Estimation of surplus nutrients and pesticide residue, and their movement.

Nitrogen balance for agricultural watersheds M. Goss, Department of Land Resource Science	watershed
Retention of pesticides by Ontario soils L.J. Evans, Department of Land Resource Science	plot
Movement of chemicals and bacteria off/out of the root zone D. Rudolph Centre for Ground Water Research (Waterloo) and G. Kachanoski, Department of Land Resource Science	field/watershed

(and other associated research)

2. Contamination partitioned by surface or ground water.

Modelling the quality of surface and tile drainable water R. Rudra, School of Engineering	field
Field techniques for measuring the hydraulic conductivity of soil D.E. Elrick, Department of Land Resource Science	field
Impact of manure and fertilizer on nitrate contamination on ground water E. Beauchamp and G, Kachanoski, Department of Land Resource Science	field
Sustainable water use in southern Ontario R. Kreutzwiser, Department of Geography	regional

(and other associated research)

3. Effect of management practices on nutrient balance, pesticide fate and excess water.

Effect of management on unsaturated transport properties of soil G. Kachanoski, Department of Land Resource Science	field
Control of soil erosion and associated water pollution T. Dickenson. R. Rudra, School of Engineering	watershed
Variable rate application technology for N fertilizers (effect on leaching) G. Kachanoski. Department of Land Resource Science	field

(and other associated research)

A strategy of collaboration suggests that each research participant contribute its area of expertise to the development of IROWC. Such a strategy is briefly outlined below.

1.	Research and development of IROWC:	universities, agricultural research stations, centres of expertise (e.g., National Hydrology Research Institute)
2.	Collaborative partnerships:	federal-provincial agreements (e.g., Green Plan), stakeholder participation (participatory research), international consultation (e.g., OECD, Great Lakes Commission)
3.	Validation and calibration:	water quality monitoring programs

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FIGURE CAPTIONS

Figure 1 Nitrogen Inputs and Outputs for Wheat Production (source: Juergens-Gschwind 1989)

Figure 2 Climatic Moisture Index - P-PE in the Prairies and Southern Ontario

Figure 3 IROWC - Change in Relative Potential Contaminant Concentration (RPCC) for Nitrogen (1981- 1991) in Southern Ontario

Figure 4a IROWC - Relative Risk of Contamination in Southern Ontario (Nitrogen - 1991)

Figure 4b IROWC - Relative Risk of Contamination in Southern Ontario (Triazine - 1993)

Figure 4c IROWC - Relative Risk of Contamination in Southern Ontario (Nitrogen and Triazine)

Figure 5 Nitrogen Loading in Bamberg Creek Watershed (source: Ecologistics Limited 1993)

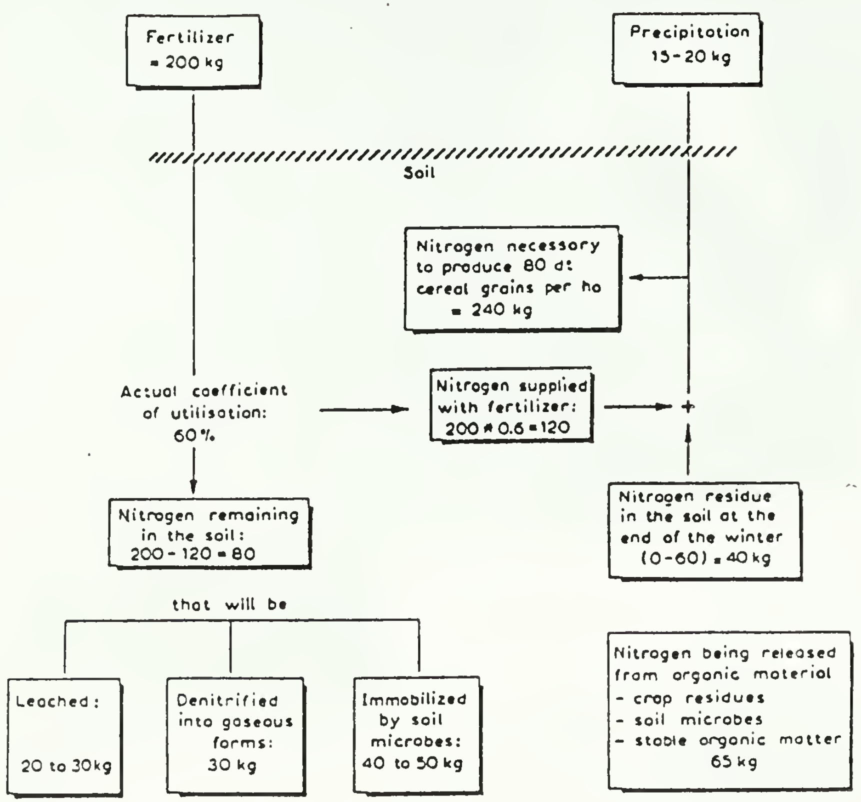
Figure 6 Nitrogen Uptake by Corn and Moisture Profile, Elora Research Station

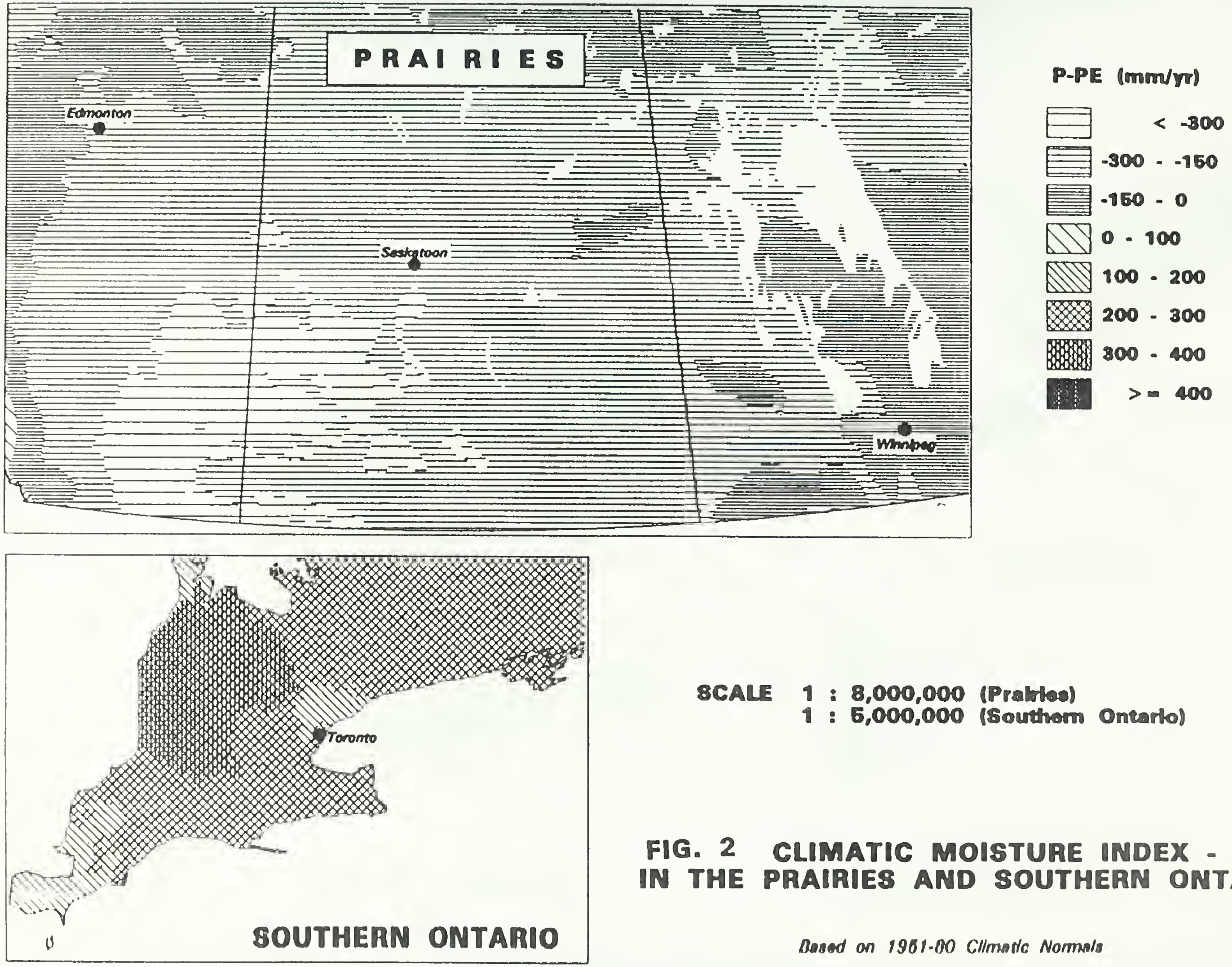
Figure 7 Predicted Annual Atrazine Loading (mg/m^2) Over Time at 90 cm Depth for Four Soils in the Grand River Watershed (source: de Jong et al. 1994)

Figure 8 IROWC - Spatial Framework for Analysis at Broad Level in Selected Watersheds of Southern Ontario

Figure 9 IROWC - Relative Potential Contaminant Concentration (RPCC) for Nitrogen in Selected Watersheds of Southern Ontario 1981 - 1991

FIG. 1 Nitrogen Inputs and Outputs for Wheat Production (source: Juergens-Gschwind 1989)





SCALE 1 : 8,000,000 (Prairies)
1 : 5,000,000 (Southern Ontario)

FIG. 2 CLIMATIC MOISTURE INDEX - P-PE IN THE PRAIRIES AND SOUTHERN ONTARIO

Based on 1951-80 Climatic Normals

FIG. 3 IROWC - CHANGE IN RELATIVE POTENTIAL CONTAMINANT CONCENTRATION (RPCC) IN SOUTHERN ONTARIO

Nitrogen (crop) 1981 - 91

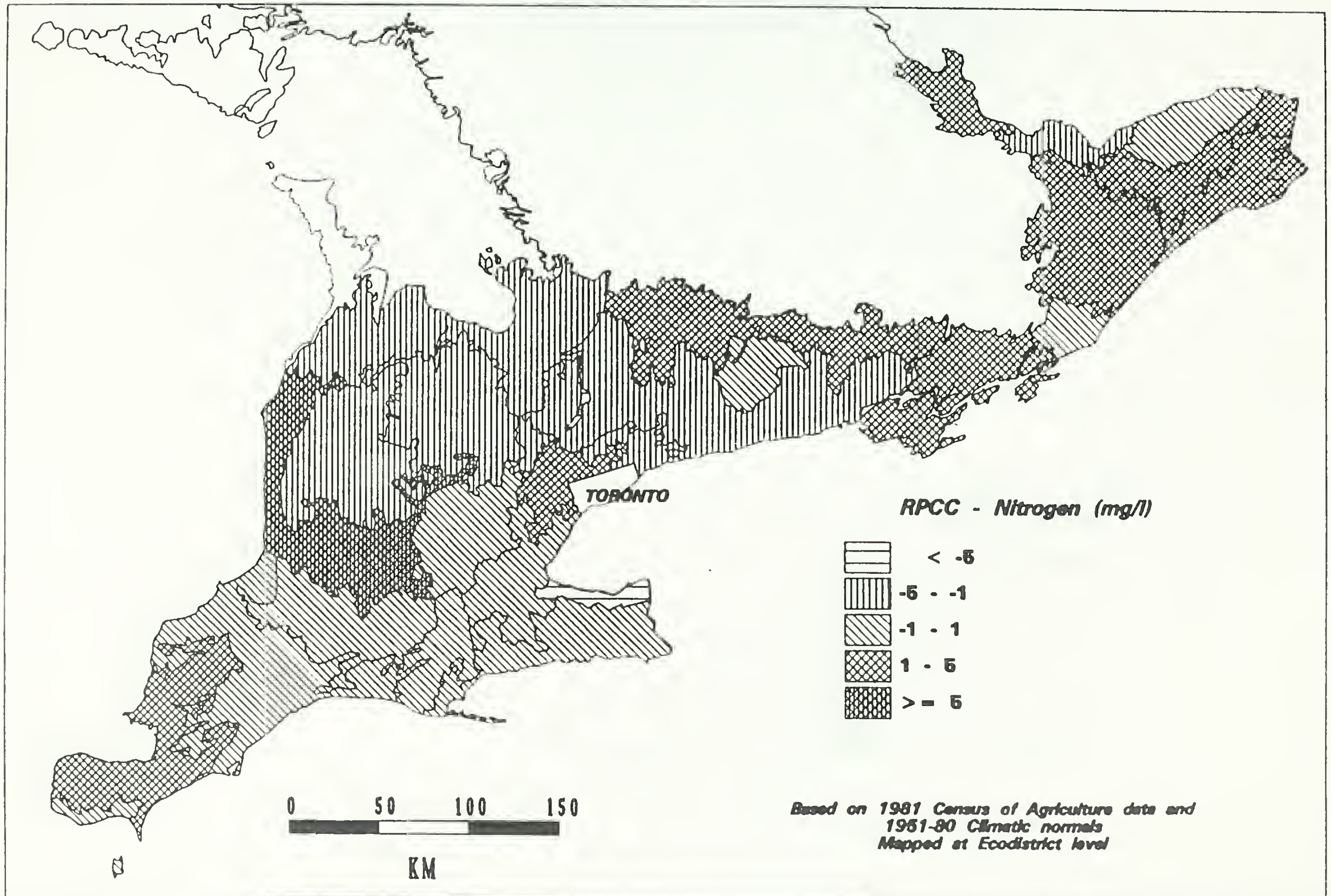


FIG. 4a IROWC - RELATIVE RISK OF CONTAMINATION IN SOUTHERN ONTARIO
Nitrogen (crop and residual manure) 1991

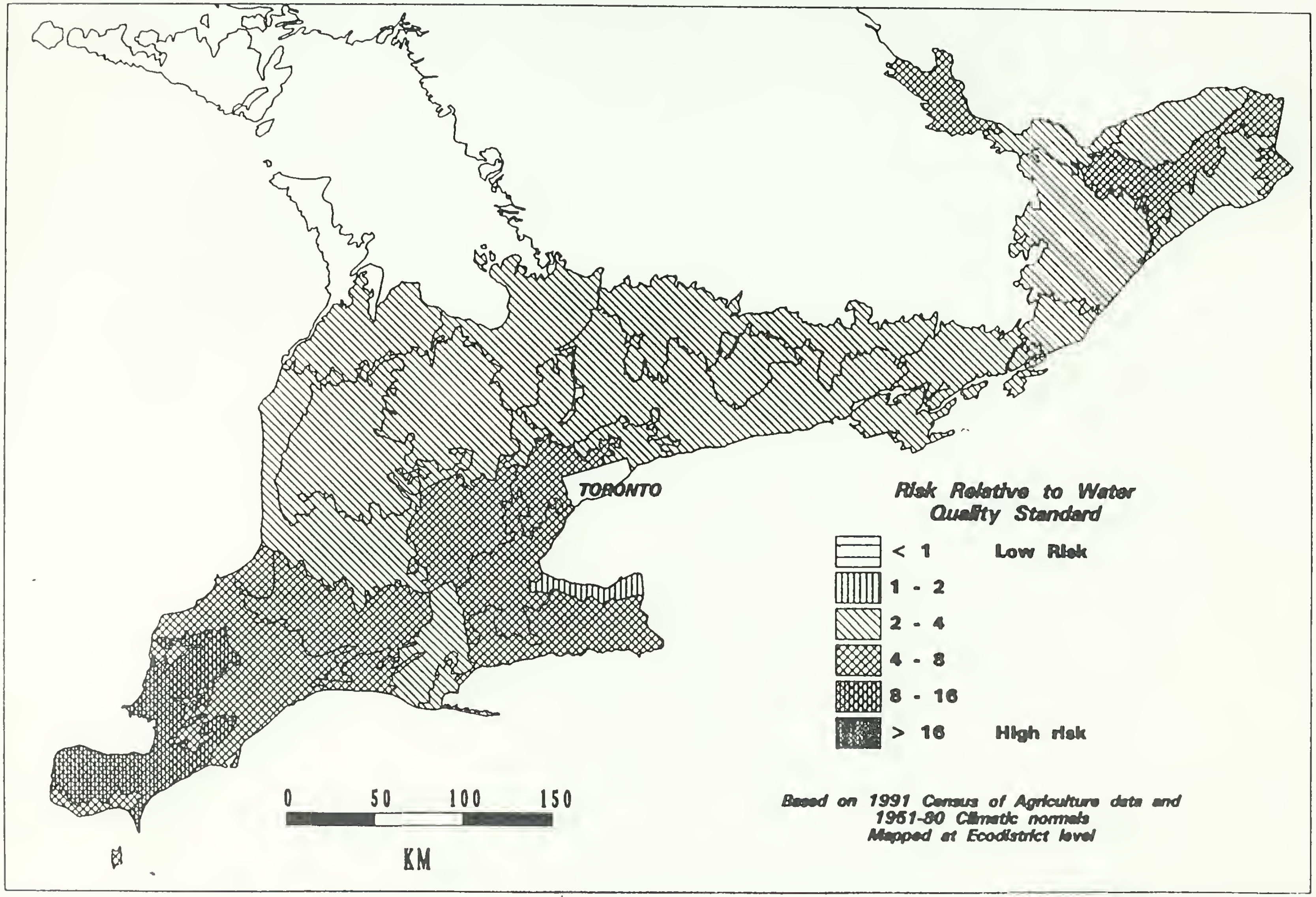


FIG. 4b IROWC - RELATIVE RISK OF CONTAMINATION IN SOUTHERN ONTARIO
Triazine 1993

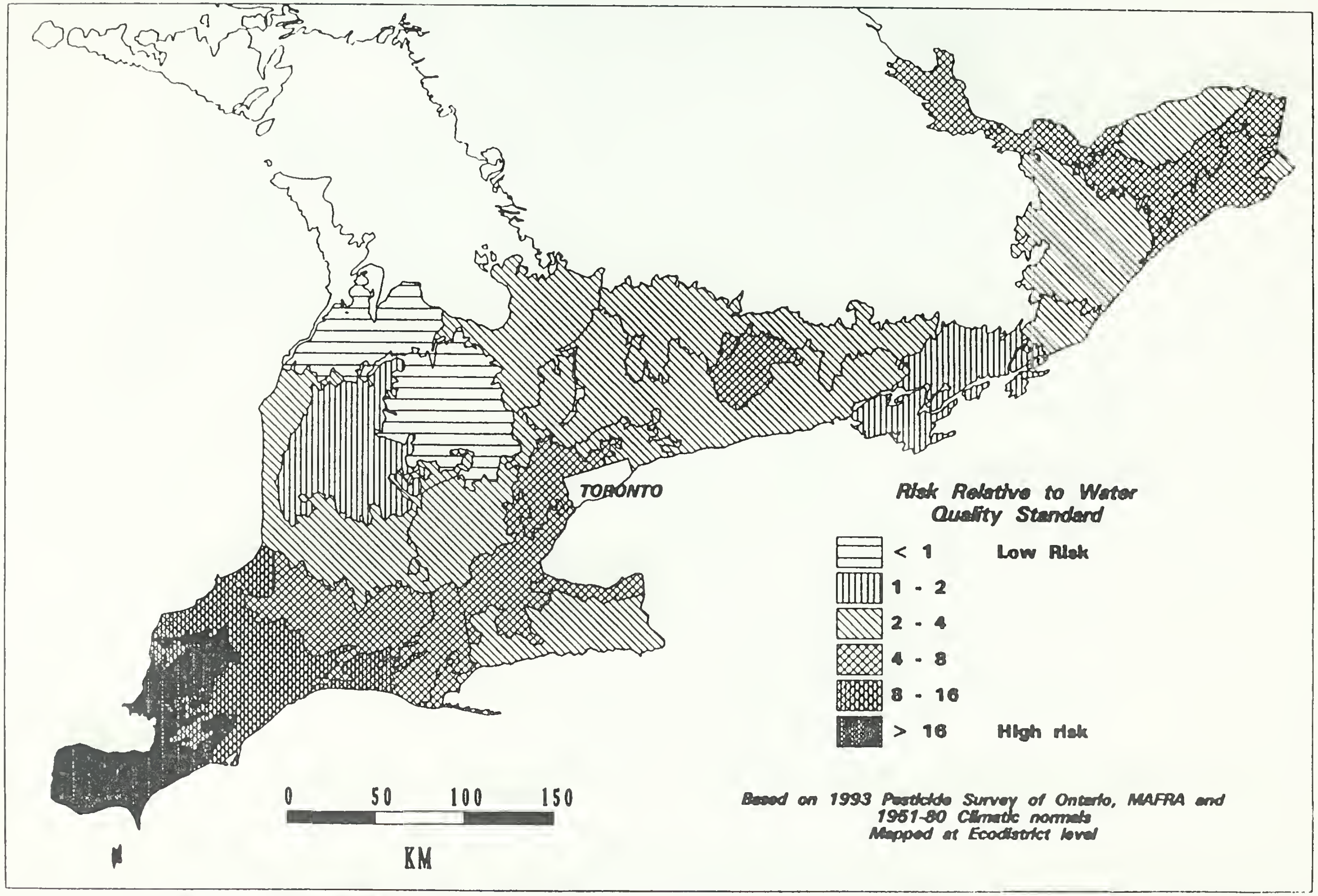


FIG. 4c IROWC - RELATIVE RISK OF CONTAMINATION IN SOUTHERN ONTARIO
Nitrogen and Triazine

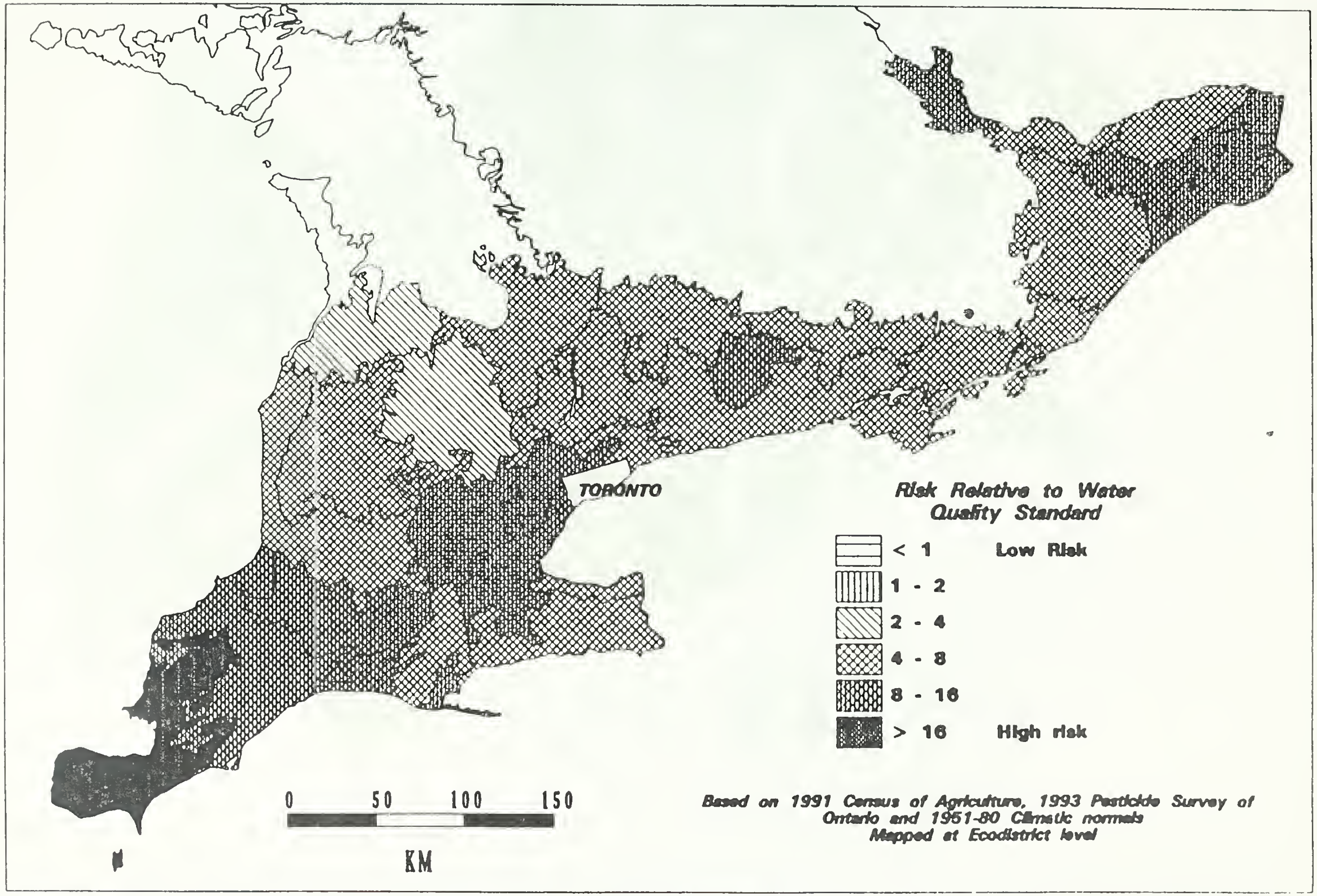


FIG. 5 Nitrogen Loading in Bamberg Creek Watershed (source: Ecologistics Limited 1993)

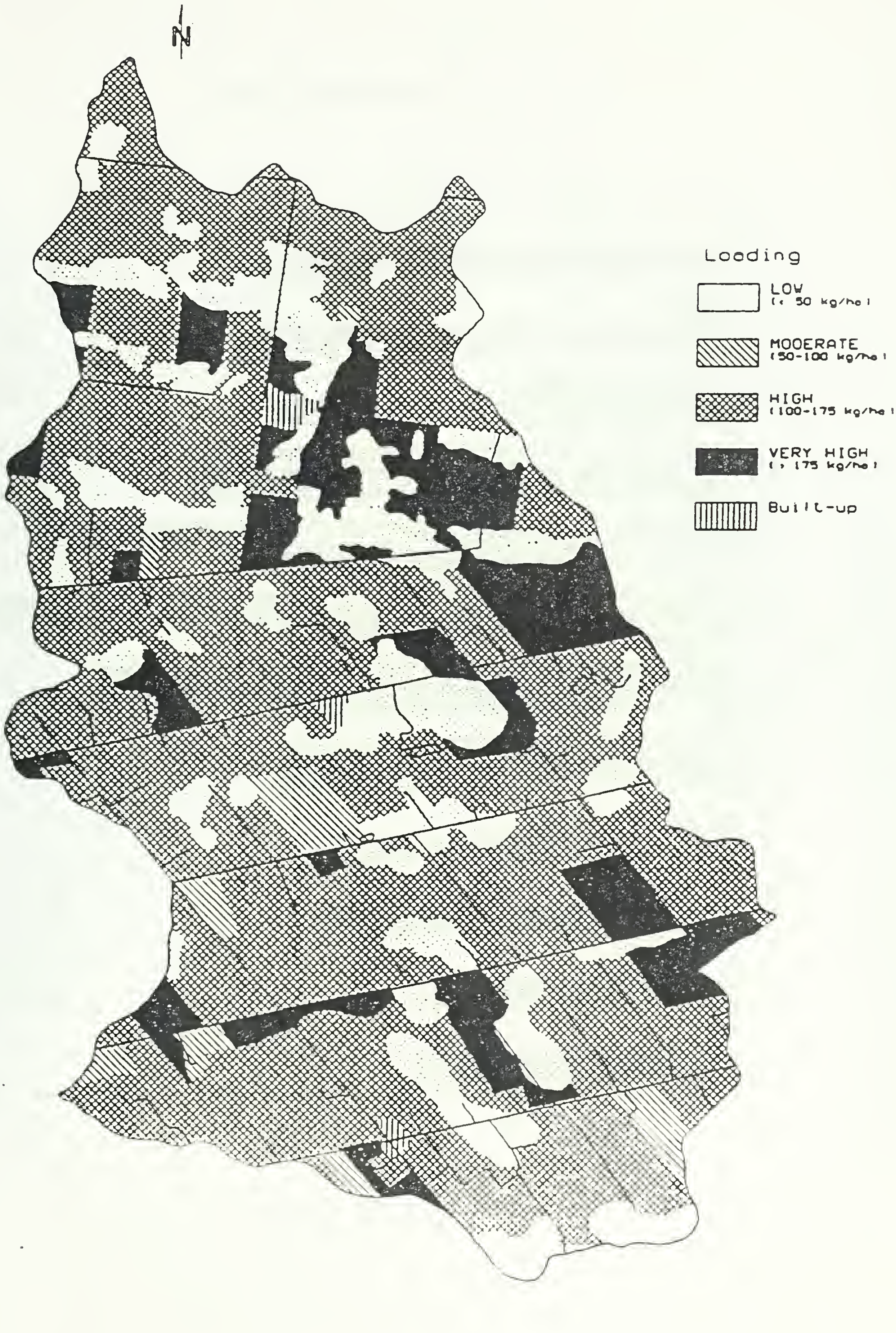
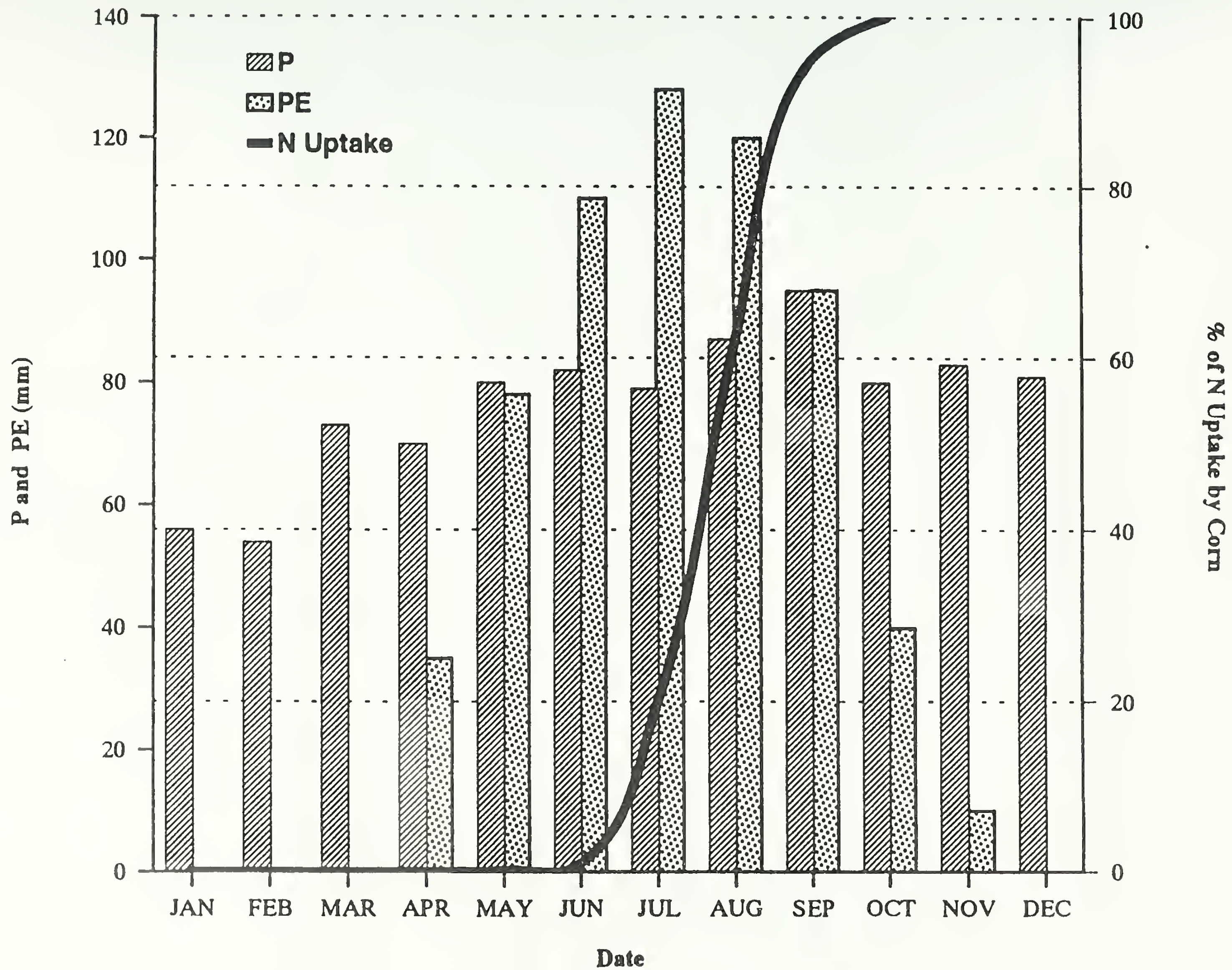


Fig. 6 Nitrogen Uptake by Corn and Moisture Profile - Elora Research Station



Data Sources: 1) N uptake from M.H. Miller, Univ. of Guelph (personal communication),
2) Climatic data from Canadian Climatic Centre, Environment Canada.

FIG. 7 Predicted Annual Atrazine Loading (mg/m^2) Over Time at 90 cm Depth for Four Soils in the Grand River Watershed (source: de Jong et al. 1994)

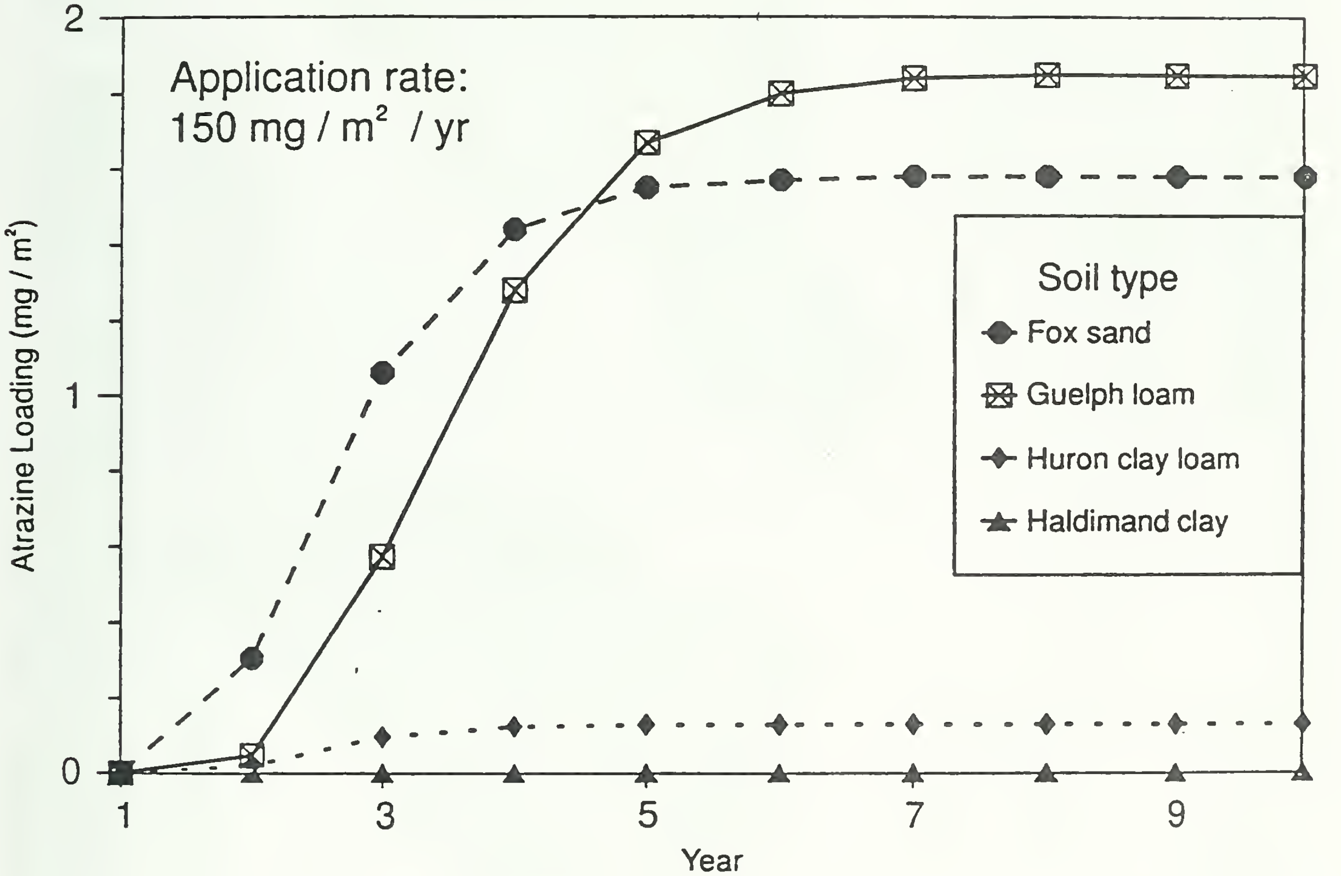
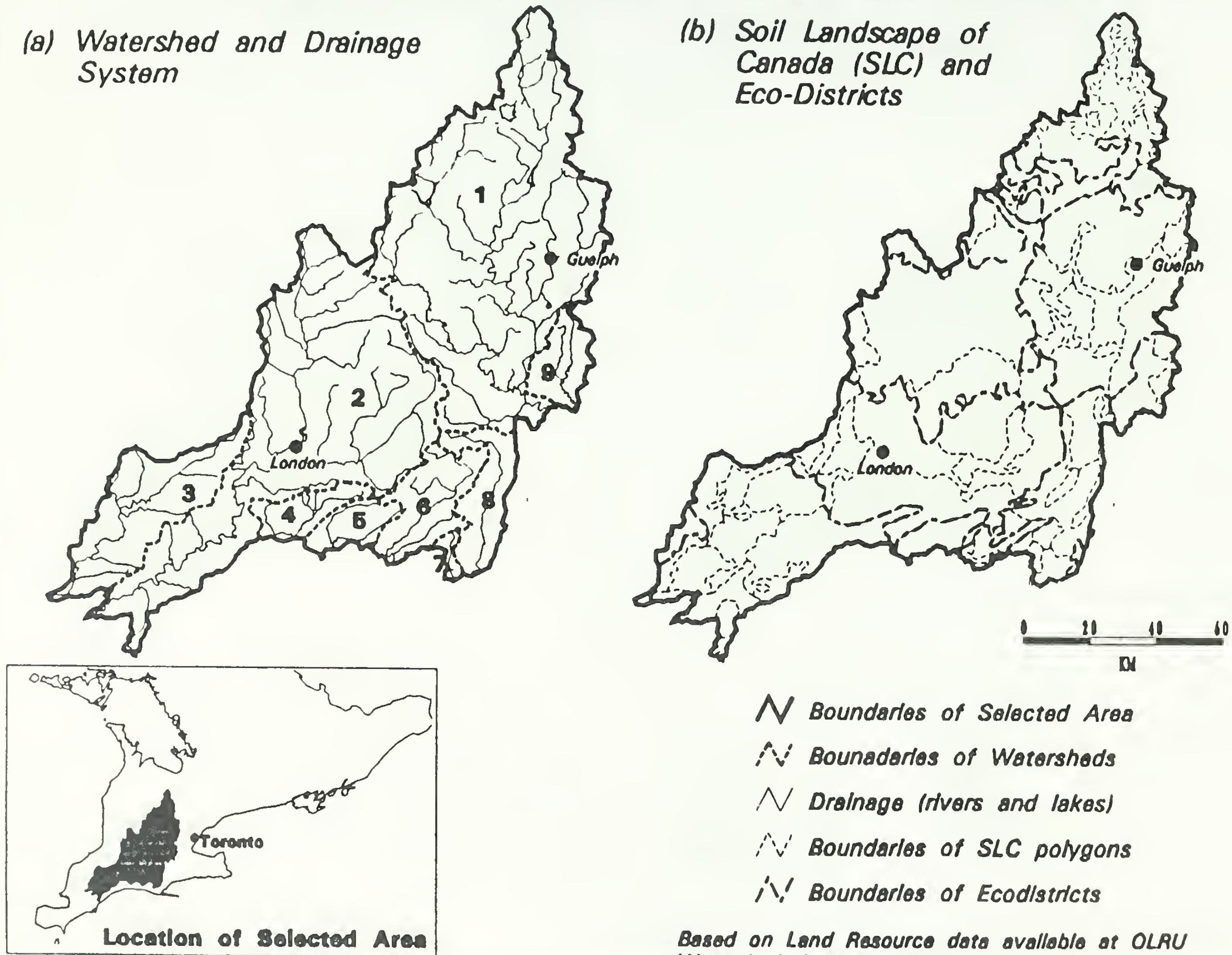


FIG. 8

IROWC - SPATIAL FRAMEWORK FOR ANALYSIS AT BROAD LEVEL IN SELECTED WATERSHEDS OF SOUTHERN ONTARIO

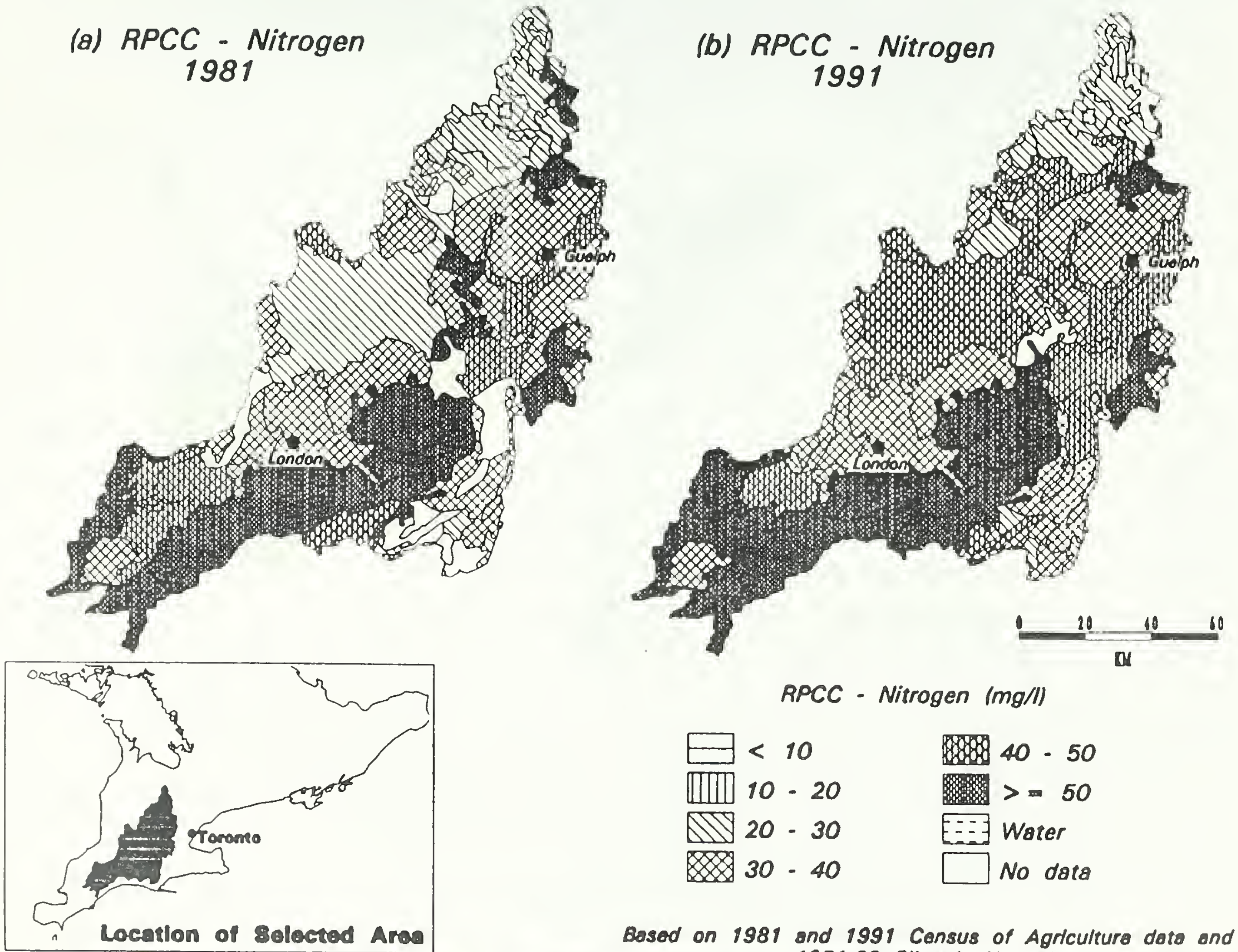
Watersheds, Drainage, Soil-Landscape and Eco-Units



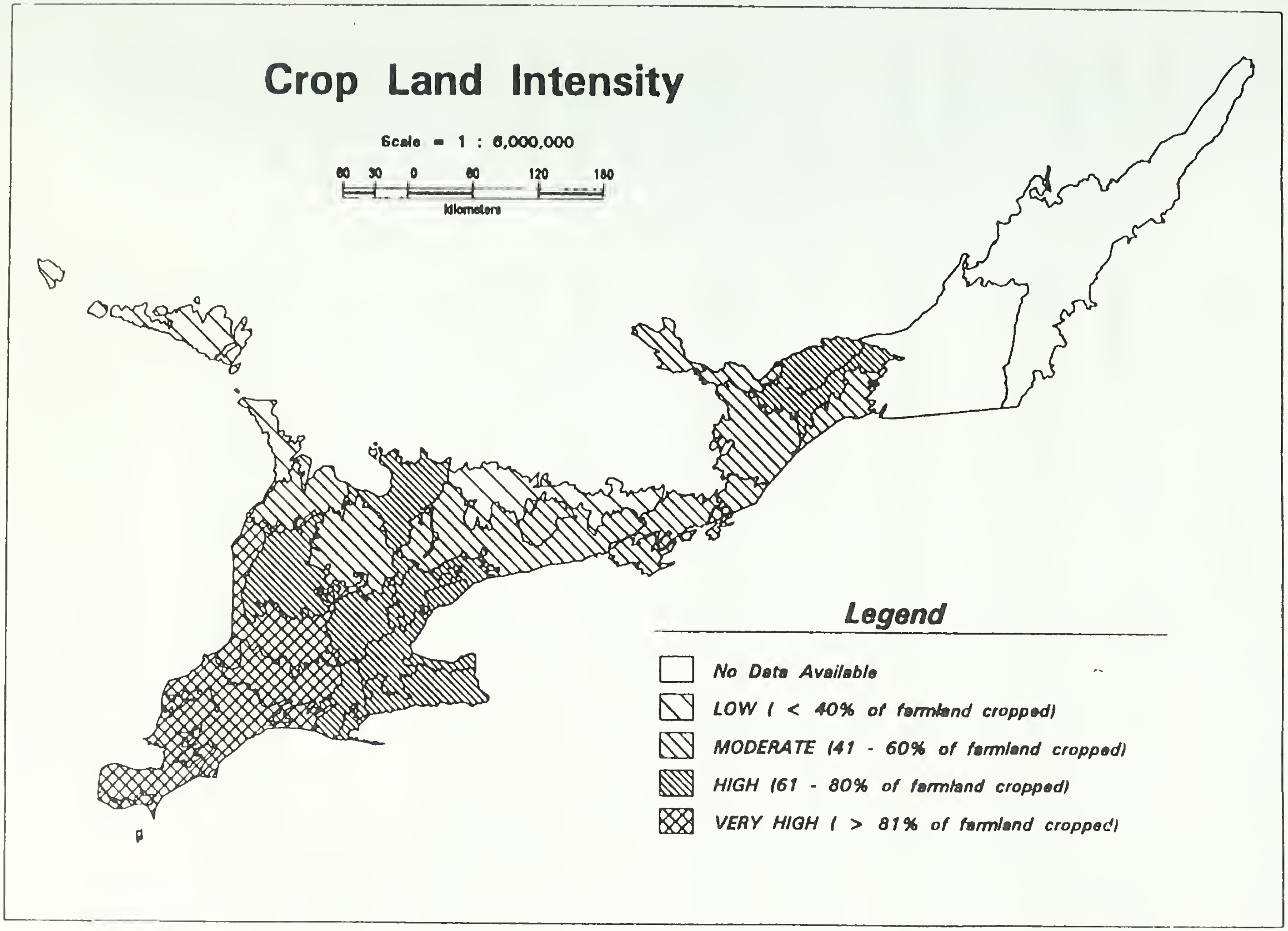
Based on Land Resource data available at OLRU
Watershed data provided by Environment Canada

IROWC - RELATIVE POTENTIAL CONTAMINANT CONCENTRATION (RPCC) IN SELECTED WATERSHEDS OF SOUTHERN ONTARIO

Nitrogen (crop) 1981 and 1991



Based on 1981 and 1991 Census of Agriculture data and 1951-80 Climatic Normals


Crop Land Intensity
Scale = 1 : 6,000,000
60 30 0 60 120 180
kilometers
Legend
No Data Available
LOW (< 40% of farmland cropped)
MODERATE (41 - 60% of farmland cropped)
HIGH (61 - 80% of farmland cropped)
VERY HIGH (> 81% of farmland cropped)

CANWANET
A network of Canadian watersheds

Title of Study :	Wilmot Watershed Study
Name of watershed :	Wilmot Valley
Contact Person :	Linnell Edwards
Location (province) :	Prince Edward Island
Objective of study :	Measure and model soil losses; develop soil conservation recipes for farmers; develop a local model; test existing models
Size in hectares :	420.00
Scale and Level :	1:10000 Level 2
Year study started :	04/01/86
Present status :	Ongoing
Description of area :	Three sub-watersheds studied: 140,203 and 420 ha in size; rolling topography with slopes up to 12 %; annual rainfall 1040 mm; soils are predominantly sandy loams; crops include potatoes followed by cereals, tobacco and soybeans
Annual Cost :	
Data collected :	Water flow, surface runoff, sediment quality: NO3. N, C, P2O5, K2O,

Title of Study :	Paired watershed study on the Bélair River Watershed
Name of watershed :	Bélair River Watershed
Contact Person :	Jacques Galichand
Location (province) :	Québec
Objective of study :	Measure and compare water quality of the paired watersheds; quantify the impact of the treatments on water quality and on the economy of the untreated watershed
Size in hectares :	1,030.00
Scale and Level :	1:5000 Level 1 or 2
Year study started :	01/11/94
Present status :	Ongoing
Description of area :	Two watersheds studied: 480 (of which 210 ha cultivated) and 550 (of which 350 ha cultivated) ha in size; topography mostly hills, with slopes up to 10%; mostly pastures and hay; dairy and pork production; manure problems; no pesticides used
Annual Cost :	$257,000.00
Data collected :	Water flow, water temperature, pH, air temperature, wind data, precipitation including snow, water quality parameters for samples taken weekly include NO3, nitrites, total N NH4-N, total P and K, dissolved P, BOD, fecal coliform count

Title of Study :	Black Brook Experimental Watershed Study		
Name of watershed :	Black Brook Experimental Watershed		
Contact Person :	Lien Chow	Location (province) :	New Brunswick
Objective of study :	Provide quantitative information on surface and subsurface runoff; evaluate the impact of various cropping systems environmental quality; validate and modify existing models for N.B.		
Size in hectares :	1,500.00	Scale and Level :	Variable Levels 1 to 2
Year study started :	04/01/90	Present status :	Ongoing
Description of area :	Gently rolling topography with slopes varying from 1 to 9%; mostly potato crop, rotation with grain, peas and hay; 65% of area agricultural land		
Annual Cost :			
Data collected :	Water flow at 8 locations, surface runoff, climatic data, water quality data: sediment, N-NO3, P, K, Mg, Ca for all locations, pesticides at the outlet;		

Title of Study :	Predicting pesticide migration through soils of the Great Lakes Basin		
Name of watershed :	Grand River Watershed		
Contact Person :	Reinder De Jong	Location (province) :	Ontario
Objective of study :	Develop methodology for predicting; conduct preliminary testing and evaluation of methodology		
Size in hectares :	680,000.00	Scale and Level :	1:250000 Level 4
Year study started :	04/01/89	Present status :	Ended in 1994
Description of area :	Main crops are corn and soybeans; soil types range from sand to silty clay; annual rainfall 850-975 mm		
Annual Cost :	$100,000.00		
Data collected :	none		