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# LAND USE / LAND COVER AND WATER QUALITY IN THE MUSKEGON RIVER WATERSHED, MICHIGAN: A CASE STUDY

By

Ranjeet John

A THESIS

Submitted to
Michigan State University
in partial fulfillment of the requirements
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## ABSTRACT

# LAND USE / LAND COVER AND WATER QUALITY IN THE MUSKEGON RIVER WATERSHED, MICHIGAN: A CASE STUDY

By

Ranjeet John

Water quality in the Muskegon River Watershed is a function of land use such as agriculture, residential development, industry, and transportation. Ongoing residential development around the lakes as well as nutrient-rich runoff from urban and agricultural landscapes affect water quality. Landsat 7 (ETM+) imagery was used to obtain an LULC map in the MRW through an unsupervised classification. Surface hydrological modeling was performed on a SRTM DEM to delineate first order sub-watersheds, which are more susceptible to non-point pollution as they have no upstream contributing flow. This study compares LULC (2001-2002) in the MRW with water quality indices such as total nitrogen (TN), total phosphorus (TP), specific conductivity, sensitive insect species (EPT taxa) and total invertebrate taxa. Results indicate that there is significant correlation between an increase in proportions of agricultural / urban land use within the watershed and water quality indices such as total phosphorus concentration. In addition, there was a negative correlation between the percentage of urban land use within the sub-watershed and sensitive insect taxa as well as invertebrate populations. TN and TP concentrations were also influenced by distance to urban and agricultural areas respectively.

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## TABLE OF CONTENTS

## List of Tables

## List of Figures

Images in this thesis are presented in color.

**List of Abbreviations**

| | |
|---|---|
| AOI | Area Of Interest |
| CAFO | Confined Animal Feeding Operation |
| CGCEO | Center for Global Change and Earth Observation |
| DEM | Digital Elevation model |
| DN | Digital Number |
| DOQQ | Digital Ortho Quarter Quads |
| DTED | Digital Elevation Terrain Data |
| EPA | Environmental Protection Agency |
| EPT | Ephemeroptera, Plecoptera & Trichoptera, |
| ETM+ | Enhanced Thematic Mapper |
| H.U.Index | Human Use Index |
| ISA | Impervious Surface Area |
| ISODATA | Iterative Self-Organizing Data Analysis Technique |
| IWI | Index of Watershed Indicators |
| LULC | Land Use / Land Cover |
| MDNR | Michigan Department of Natural Resources |
| MIRIS | Michigan Resource Information System |
| MMU | Minimum Mapping Unit |
| MRW | Muskegon River Watershed |
| MRWEAP | Muskegon River Watershed Ecological Assessment Program |
| NAWQA | National Water Quality Assessment project |
| NRCS | National Resource Conservation Service |
| NPS | Non-Point Source Pollution |
| SRTM | Shuttle Radar Topography Mission |
| TOA | Top Of the Atmosphere |
| TN | Total Nitrogen |
| TP | Total Phosphorus |
| TRFIC | Tropical Rainforest Information Center |
| UTM | Universal Transverse Mercator |
| WGS | World Geodetic System |
| WI | Wetness Index |

# CHAPTER 1 INTRODUCTION

## 1.1 Background

Human induced changes in the Muskegon River Watershed (MRW) in north-central Michigan have had an adverse effect on the quality of water in its lakes and streams. Water quality in the MRW is a function of the different land uses such as agriculture, residential development, industry, and transportation. Residential development around the lakes as well as the construction of roads and parking lots affect water quality. Water quality also be affected because of algal blooms and high sediment loads carried by runoff from urban and agricultural landscapes. The lakes and wetlands of the MRW have important recreational and economic uses, but some are in danger of hyper-eutrophication, a phenomenon caused by excessive nutrients in the runoff from urban and agricultural landscapes. Eutrophication is characterized by water-bodies being dominated by the same set of nuisance species that can tolerate the eutrophic conditions and leads to the subsequent reduction in the diversity of species (Carpenter et al., 1995). The increase in stream runoff is due to the cumulative impact of urban growth along the eastern shores of Lake Michigan (Pijanowski et al., 2002). This urban growth encompasses urban, sub-urban and near shore residential development, an d has increased in 2002.

## 1.2 Problem statement and objectives

There is an increase in urban development within the watershed, which affects water quality due to increase in surface runoff. The increase in impervious surface area due to urban development leads to a subsequent increase in water temperature in streams fed by surface runoff, which modify the stream hydrology through incision. Stream incision is defined as "the rate of incision in detachment-limited systems and by definition, determined by the stream's ability to erode the bed, usually by a combination of abrasion and plucking" (Whipple et al., 2000).

Water quality is degraded in certain sub-watersheds by inputs of nutrients such as nitrogen and phosphorus from towns and agricultural areas in the form of non-point source pollution (NPS). Confined animal feeding operations (CAFO's) act as a point source for phosphorus input. Excessive sedimentation, another reason for deterioration of water quality, is attributed to stream bank erosion, roads, and farm practices like drainage.

To obtain a firm understanding of the spatial and temporal nature of water quality, an ecological assessment of the watershed using remote sensing and GIS is necessary, as it is cost effective. The ecological assessment is carried out at the sub-watershed level rather than a basin wide study. This allows a cross comparison of the sub-watersheds in terms of their vulnerability to deterioration in water quality.

Ecological assessments are necessary in the protection, maintenance and restoration of ecological systems (US EPA, 1996). Land use indicators such as percentage of urban / built up and agriculture combined with topographic attributes such as elevation and slope help identify the anthropogenic activities that generate stress in the ecosystem under study.

The objectives of the research are:

1) The mapping of urban built up areas and agricultural land use at the sub-watershed level as well as areas susceptible to non-point source pollution.
2) Statistical analysis to correlate percentage of land use with water quality indicators

This research seeks to correlate land use / land cover (LULC) in the sub-watersheds with water quality indicators. In addition, an inventory of the LULC types was carried out in areas that were susceptible to non-point sources of pollution such as first order watersheds. The percentage of urban and agricultural land use within the sub-watersheds as well as the first order watersheds were regressed against various water quality indices such as total nitrogen, total phosphorus, conductivity as well as biological indicators such as sensitive insect populations (EPT taxa) and total invertebrate taxa to build predictive models.

### 1.3 Study area

The Muskegon River is 219 miles long from its start at Houghton and Higgins lakes down to its mouth at Muskegon Lake, and eventually, Lake Michigan. The watershed is located between latitude 43° to 44°30'N and longitude 84°30' to 86°W (Figure 1-1).

The MRW covers an area of almost 7000 square km and includes 94 tributaries, 183 stream segments and 95 dams (O'Neal, 1997). The watershed is within the counties of Wexford, Missaukee, Roscommon, Kalkaska, Crawford, Lake, Osceola, Clare, Newaygo, Mecosta, Montcalm and Muskegon. The primary tributaries of the Muskegon river include the West Branch of the Muskegon River, the Clam River, the Middle Branch River, the Hershey River, the Little Muskegon River, Bigelow Creek, Brooks Creek, and Cedar Creek (O'Neal, 1997). Some of the important cities in the watershed are Big Rapids, Newaygo and Muskegon.

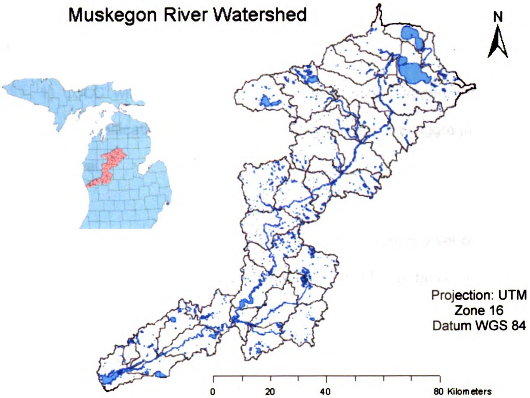


Figure 1-1. Muskegon River Watershed.

### 1.4 Hypothesis

The purpose of this research is to test whether there is a correlation between water quality and percentage of LULC within the MRW. The specific hypothesis is that LULC affects water quality within the MRW.

### 1.5 Sub-hypothesis

1) LULC within the MRW can be accurately mapped through an un-supervised classification of Landsat-7 (ETM+) imagery and aggregated to the sub-watershed level.

2. Water quality within the MRW, measured through a set of water quality indicators and aggregated to the sub-watershed level, can be correlated with the percentage of LULC.

### 1.6 Benefits of this research

This study, based on correlations between specific LULC types and water quality would not only assess the ecological integrity but also identify the sub-catchments within the MRW with greater proportions of urban and agricultural land use and thus help in better management of watershed. Once identified, the relationships between the various land uses and water quality indicators can be applied to other watersheds in the State of Michigan as well. This study of land–water interactions within the MRW is especially important at a time when urban sprawl threatens the ecological integrity of Michigan's water and natural resources.

# CHAPTER 2 LITERATURE REVIEW

## 2.1 Land use and water quality

Anthropogenic activities on land can have a detrimental effect on riparian bodies such as rivers and streams. The National Water Quality Assessment Program (NAWQA) was initiated by the USGS in 1991 to understand how human activities and natural processes affect water quality in this nation (USGS, 1999). This study came about partly due to the growing public concerns about the quality of the nation's water resources. The passing of the Clean Water Act in 1977, whose purpose was "to restore and maintain the chemical, physical, and biological integrity of the waters of the United States" led to a strong campaign by the public and private sectors to limit contaminants from point sources from entering streams (USGS, 1999). NAWQA findings indicate that streams in watersheds and basins with significant agricultural and urban development have higher levels of nutrients and pesticides, which contribute to higher growth of algal growth. The increase in impervious surfaces like paved lots and urban pavements increased surface runoff (USGS, 1999). Also, the NAWQA studies showed that streams in basins with steep slope and clayey soils were vulnerable to contamination due to stream runoff. In the continental United States, urban streams had the highest concentrations of pesticides such as chlordane and dieldrin (USGS, 1999). It was also found that concentrations of phosphorus were higher in urban areas than in rural areas and this in part due to the effluent from wastewater plants (USGS, 1999). The runoff from urban areas have elevated levels of phosphorus and nitrogen and caused the eutrophication in lakes, streams and reservoirs. Cities are important contributors to non-point pollution and homeowners with lawns apply just as much fertilizer and pesticides per unit area as farmers would on their farms. Studies conducted in the upper Midwest suggest that lawn care, through the application of nutrients rich in nitrogen and phosphorus contributes to nutrient rich runoff and that nutrient concentrations are more than those originating from impervious surface areas like roofs and paved surfaces like streets and driveways (Bannerman et al., 1993; Waschbusch et al., 2000; Steuer et al., 1997).

The sources of water pollution typically fall into two categories, 1) point source pollution and 2) non-point source pollution. Point source pollution originates from discrete locations that are spatially explicit and some examples are sewage treatment plants, industrial effluents and land disposal sites. Non-point sources of pollution originate from various diffuse sources that occur over a larger and broader geographical area. Some examples of non-point source of pollution include agricultural runoff, storm water, urban runoff and atmospheric deposition. Because of its diffuse nature, non-point source pollution cannot be isolated in a spatially explicit manner (USGS, 1999).

Agriculture is the most important source of non-point source pollution according to the US EPA (2000). Agricultural practices like the spraying of pesticides and herbicides, irrigation, planting and harvesting, and confined animal feedlots all contribute to NPS pollution. Another significant form of agricultural NPS pollution seems to be siltation (Rabeni & Smale, 1995). The nutrients most often considered in land-water interaction studies are nitrogen and phosphorus (Turner et al., 2001). Nitrogen concentrations in rivers are sensitive to land use patterns, the riparian zone structure and river flow (Cirmo and McDonnell, 1997). Accumulation of excess phosphorus in rivers and streams has been recognized as the cause for eutrophication (Carpenter et al., 1998). Similarly, concerns about nitrogen inputs into aquatic ecosystems have been raised (Mueller and Helsel, 1996; Vitousek and Howarth, 1991). Farmers apply nutrients such as nitrogen and phosphorus to their farm plots, but not all of it is absorbed by the plants. The nutrients in the soil are leached through runoff and find their way into lakes and streams where they cause eutrophication. Recent advances in technologies such as GIS and remote sensing have seen various studies conducted to assess agricultural NPS pollution at different geographical scales that include catchment, watershed, basin and landscape level assessments (Richards et al., 1993; Allan et al., 1997; Johnson & Gage, 1997; Harding et al., 1999; Lammert and Allan, 1999).

Regression analysis has been used to determine relationships between land use patterns and nitrogen / phosphorus concentrations (Osborne and Wiley, 1988). The land use patterns in the Salt River Basin, Illinois, were mapped from aerial photos and results indicated that urban land use and its distance to the stream was the most important variable in predicting nutrient concentrations in stream water (Osborne and Wiley, 1988). A study in the Minneapolis–St. Paul Metropolitan region demonstrated that lakes within watersheds dominated by forests and intact wetlands tended to be less eutrophic and have lower levels of chlorides and lead (Detenbeck et al., 1993). On the other hand, lakes within agricultural watersheds were more likely to be eutrophic. There was also a positive correlation between the percentage of urban land use and phosphorus in the Minneapolis area (Detenbeck et al., 1993). In another study, LULC within 62 catchments in the Saginaw River, Michigan was related to stream water chemistry (Johnson et al., 1997). The results of the study demonstrated that the land use / land cover had a strong influence on water quality but the predictive power of specific water quality indicators varied by season (Johnson et al., 1997). The spatial distribution and proportion of different land use / land cover types has been found to directly affect water quality (Hunsaker & Levine, 1995). Forested riparian zones had better water quality than deforested riparian zones with similar agricultural land use (Hunsaker & Levine, 1995). Soranno (1996) found significant relationships between land use and concentrations of phosphorus in the Lake Mendota watershed, Wisconsin. A GIS based predictive model was built where the phosphorus export coefficient varied by land use type. The study also measured the contribution of phosphorus to a lake as a function of distance. In addition, the results emphasized the importance of riparian vegetation in reducing forest runoff (Soranno et al., 1996).

Horton (1933) suggested that rainfall within a watershed either infiltrates the soil or is transported overland to streams as storm flow. Overland flow or sheet flow occurs when the precipitation intensity exceeds the soil infiltration capacity. However, hortonian overland flow, which is widely accepted as the case for degraded watersheds that are predominately agricultural, does not seem to hold for forested watersheds where abundant canopy cover prevents erosion and facilitates infiltration.

Urbanization is a significant land use today as it is associated with population, the economy and the conversion of other LULC types. The proliferation of impervious surface areas (ISA), in areas that are heavily vegetated reduces carbon sequestration (Milesi et al., 2003). In addition, the increase in impervious surface areas results in the alteration of sensible and latent heat fluxes leading to the formation of urban heat islands (Changnon, 1992). Urban development plays an important role in influencing the rate of runoff and erosion (Goudie, 1990). Urban growth undergoes several stages (Kibler, 1982). In the early stage of urban development, the logging of trees and vegetation may result in the decrease of evapotranspiration, interception and increase siltation as well as total suspended solids. The latter stages of growth include an increase in the construction of houses, streets and storm drains which in turn leads to a decrease in infiltration, increase in storm flows and a lower ground water table. As the number of residential and commercial buildings increase, there is a subsequent increase in paved surfaces liked roads and parking lots which in turn decrease the time of concentration which is the amount of time required for water to move from the most distant part of the watershed or catchment to the outlet and is a function of the percentage of impervious area and slope. The increase in impervious surfaces will result in higher peak discharges after rainfall events because of a decrease in infiltration and result in an increase in surface and storm water runoff (Booth, 1991). The effects of an increase in ISA are found when 10% of the watershed is covered with impervious surfaces, leading to the alteration of stream channels; rising water temperatures; a reduction in the diversity of aquatic insects and fish; and the degradation of wetlands and riparian zones (Beach, 2002). The response of rivers in terms of discharge due to increase in sediment loading due to changes in land use patterns such as deforestation has been documented (Ligon et al., 1995).

### 2.1.1 First order watersheds

First order streams are the uppermost, stream channels that do not have any upstream reaches and have perennial or intermittent flow (Gomi et al., 2002). First order watersheds make up a large portion (60-70%) of the catchment area (Siddle et al., 2000; Meyer and Wallace, 2001). The first order watersheds are important sources of nutrients organic matter and sediments for the higher order streams and their catchments (Gomi et al., 2002). The movement of organic matter and invertebrate species from the first order to higher order watersheds supports the fish population downstream (Wipfli and Gregorvich, 2002). In addition, leaf litter and large woody debris alter and control the stream morphology and provide habitats like riffles and pools for invertebrates and fish fry (Zimmerman and Church, 2001). Invertebrate species found in the first order streams serve as food for aquatic biota (Wipfli, 1997). The large woody debris also dams the stream channels and alters the channel reaches such as cascades and step pools (Halwas and Church, 2002). The riparian forest canopy in the first order streams attenuates incoming solar radiation and so controls the water temperature as well as the amount of light (Gomi et al., 2002).

Headwater systems that include hill slopes and first order watersheds control stream flow generation (Tsukamoto et al., 1982) and water chemistry in the stream (Likens et al., 1977). These headwater systems contain four topographic units (Hack and Goodlett, 1960) which are 1) hill slopes, 2) zero-order basins, 3) ephemeral channels emerging from the zero order basins called transitional channels, and 4) first order streams. The hillslopes do not have channelized flow. The zero order basins could be defined as an unchannelized hollow with converging contour lines (Tsukamoto et al., 1982). Temporary channels or transitionary channels may connect the zero order basin and first order streams (Tsukamoto et al., 1982). These ephemeral channels do not support the complete life cycles of macro invertebrate biota. Storm flow generation within the river basin is more rapid in first order watersheds owing to the small storage capacity. There is also a greater variation in peak flow discharge as compared to higher order watersheds (Gomi et al., 2002). The increase in impervious surface area due to urban development in the first order watersheds could increase the peak flow discharge and so are especially vulnerable to NPS pollution from urban landscapes (Gomi et al., 2002). In spite of the significant role of first order watersheds within the larger basin or catchment, their processes have been extensively modified by land use (Meyer and Wallace, 2001).

### 2.1.2 Land use in the Muskegon River Watershed

The Muskegon River Watershed (MRW) can be broadly classified into three major ecological zones: 1) the outwash bowl, 2) the morainal valleys, and 3) the freshwater estuary (Figure 2-1).

The headwaters of the Muskegon river originate in an outwash plain formed by deposits of stratified debris from glacial meltwater streams. The morainal valley which constitutes the mid–river section was created by deposits of unsorted debris (gravel, sand and boulders) left behind by retreating glaciers. The freshwater estuary or the mouth of the Muskegon river is a drowned river valley created due to coastal subsidence or through inundation by glacial melt water (Christopherson, 1995).

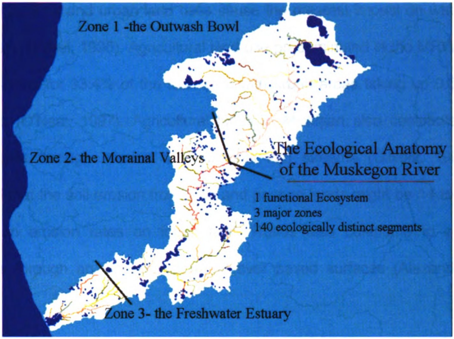


Figure 2-1. MRW: ecological classification

The history of land use within the MRW is relevant to the study as it is important to consider the past consequences of significant alterations to the watershed system. Human settlements in the MRW expanded as population increased drastically between 1810 to 1840 (O'Neal, 1997). This was largely influenced by intensive logging operations for copper and white pine. The logging caused extensive damage to the existing aquatic habitat. The water quality in the MRW was also affected by anthropogenic activities that increased in the early 1900's and reached a peak in the 1950's and 1960's. Nutrient and sediment pollution was common as well as extensive wetland reclamation in the vicinity of Muskegon lake (O'Neal, 1997).

Agricultural and urban land uses cause the greatest impact on water quality (Mueller and Helsel, 1996). Agricultural land use predominated in the MRW in 1997, and accounted for 33.4% of the total area with urban areas taking up 0.6% of the watershed (O'Neal, 1997). Agricultural lands in Michigan also contribute to poor water quality through erosion of sediments into streams (O'Neal, 1997). It is estimated that the soil erosion from crop and pasture lands might be 14 to 21 times higher than erosion rates on forest land. Roads also contribute to erosion of sediments through an increase in runoff over paved surfaces (Alexander et al., 1995).

The Muskegon river basin lies within two major land resource area classifications of the National Resource Conservation Service (NRCS). These are the northern lower Michigan sandy drift and the southern lower Michigan drift plain (O'Neal, 1997). Soil erosion in the form of annual sheet and rill erosion is 0.84 tons/acre for crops-pasture land and 0.04 tons/acre for forests in the northern lower Michigan sandy drift and 2.09 tons/acre for crops-pasture land and 0.15 tons/acre for forests (O'Neal, 1997). To help prevent the soil erosion, the NRCS has considered drainage, and forage improvements. The draining of land by deepening of existing streams destroys and eliminates many aquatic habitats. The removal of trees and other riparian vegetation leads to a degradation of canopy cover over the streams and leads to increased water temperatures (O'Neal, 1997).

Dams were constructed on Muskegon river at Big Rapids during 1866 and Newaygo in 1900 and dismantled in 1966 and 1969, respectively. Four major dams still remain on the Muskegon river and they include Reedsburg dam, Rogers dam, Hardy dam and Croton dam. Of these, Reedsburg dam is a wildlife flooding and the rest are hydroelectric dams (O'Neal, 1997). The dams pose serious environmental problems to the river as the natural flow regime of the river is altered and also change their physical, chemical and biological characteristics (Poff et al., 1997; Poff and Hart, 2002). The dams cause fish mortalities as they get caught in the hydroelectric turbines. Of the fish mortalities, as much as 70% are game fish (O'Neal, 1997). The dams also prevent the movement of aquatic insect larva, which serve as food for the various fish species. Changes in water quality and rising temperatures have been attributed to the major hydroelectric dams, namely, Croton, Hardy and Rogers's dams.

The current pattern of land use in Michigan indicates an increase in urban/built up area and a subsequent decrease in agricultural and pastureland. In 1952, the land use under agriculture in Michigan was approximately 72, 843 $km^2$ (Veatch, 1953). The area under agriculture decreased to 50,000$km^2$ and 49, 446 $km^2$ in 1987 (Natural Resources Inventory, 1987). The farmland loss can be attributed to urban sprawl or as in some cases in the MRW, re-growth of vegetation.

Though the population growth is less than the national average (6.9% as compared to 13.1%), there has been significant population shifts within Michigan's borders. The general trend is that people seem to move from urban areas to the suburbs and rural areas (Machemer et al., 1999). The increase in urban development seems to outstrip the population growth and is characteristic of urban sprawl. The increase could be attributed to the low cost of land in rural areas as compared to the cities. The increasing urbanization is alarming as Michigan's natural resources account for a large percentage (29%) of Michigan's economy (Machemer et al., 1999). The primary effect of urban development in the suburbs is fragmentation of forests from large contiguous tracts to small patches.

## 2.2 EPA monitoring programs in the US

### 2.2.1 The Environmental Monitoring and Assessment Program (EMAP)

The Environmental Monitoring and Assessment Program (EMAP) is a research program undertaken by the Environmental Protection Agency (EPA) to monitor and assess the status and trends of the ecological resources of the United States (Paulsen et al, 1991; US EPA, 1997). The EPA published a report titled "Ecological assessment of the mid-atlantic region, a landscape atlas". The atlas described ecological conditions across the mid-Atlantic region of the United States that included the states of Delaware, the District of Columbia, Maryland, Pennsylvania, Virginia, and West Virginia. The report was based on information derived from satellite imagery as well as other sources of geo-spatial information. In October 1997, the EPA released another report, which was its first index of watershed indicators (US EPA, 1997) that shared many similarities with the EMAP mid-Atlantic landscape atlas. The primary difference between the two reports is that the index of watershed indicators primarily deals with water quality issues but the mid-Atlantic landscape atlas documents the impact of land use / land cover on water quality. This study uses some of the methods mentioned in both reports to address the possible effects of land use / land cover on water quality.

### 2.2.2 EPT taxa

Environmental monitoring groups across the United States have adopted EPT (Ephemeroptera, Plecoptera, Trichoptera) taxa richness as a useful measure of stream water quality. The EPT taxa that include mayflies, caddis flies and stone flies evolved in streams with high levels of oxygen and in fast flowing waters. Any reduction in flow, depleted oxygen supply or increase in temperature results in a decrease in population. The widespread use of EPT taxa as a stressor indicator might be owing to its ease of use and effective tracking of water quality as well its habitat specific impact (Wallace et al., 1996). The use of EPT taxa is a follow–on to the historic use of benthic macro-invertebrates to evaluate water quality that dates back to early studies in the Illinois River (Richardson, 1928). Macro-invertebrate taxa were increasingly used to monitor water quality from the 1950's onwards and studies from the mid-century period began to cite EPT taxa as intolerant (Gaufin and Tarzwell, 1952).

## 2.3 Surface hydrological modeling

Topography is an important factor in determining the stream flow in forested uplands (Wolock and Price, 1994). It also defines the movement of water within a catchment area due to gravity, the spatial distribution of soil moisture (Burt and Butcher, 1985) and the water chemistry of the stream flow (Wolock et al., 1990). Surface hydrology is defined as "the spatial and temporal storage and redistribution of rainfall as it falls on or enters into the soil" (Engman, 1997). The prediction of spatial patterns and the rate of surface runoff require a hydrologic model and a categorization of the land surface (Zhang and Montgomery, 1994). Digital elevation model (DEM) data are arrays of regularly spaced elevation values referenced horizontally either to a Universal Transverse Mercator (UTM) projection or to a geographic coordinate system (USGS, 2000). Digital elevation models are being used in hydrological modeling studies (Bruneau et al., 1995) and for a variety of engineering as well as planning applications (Zhang and Montgomery, 1994).

### 2.3.1 Advantages of SRTM DEMs

The Shuttle Radar Topography Mission (SRTM) derived DEM, currently the highest resolution global DEM, was sampled at 1 arc second or 30m (Rabus et al., 2002). Until now, high resolution DEMs were obtained mostly from optical stereo data acquired from aerial photographs or satellite imagery. The DEMs obtained from these were not homogeneous due to the quality of image contrast.

Also, the presence of clouds or the lack of sunlight in stereo-pairs resulted in image artifacts (Rabus et al., 2002). The SRTM data set, on the other hand was obtained by a single technique, i.e., interferometic Synthetic Aperture Radar (InSAR) sampling in 11 days by the Space Shuttle Endeavour (Rabus et al., 2002). The absolute vertical accuracy is +/- 16 m and the absolute horizontal (90% circular error) accuracy is 20m (Sun et al., 2003). The SRTM global data set was obtained from the post processing of the C band radar interferometry which introduces an error in the vertical accuracy as the C band radar return effectively samples the height at the top of the canopy and unlike radar bands with longer wavelengths (P and L bands), does not reach the ground beneath. A validation of SRTM DEM's vertical accuracy using the Shuttle Laser Altimeter (SLA-02) showed that areas with sparse vegetative cover had absolute vertical accuracy that exceeded the mission specifications of 16m. Surface slope comparisons between the SRTM and Digital Elevation Terrain Data (DTED) DEM showed that slope derived from the SRTM DEM was superior to the slope derived from the DTED Level 1 (3 arc sec) DEM (Sun et al., 2003).

### 2.3.2 Digital terrain modeling and topographic attributes

Topographic attributes include primary attributes like elevation and slope as well as secondary attributes that are derived from combinations of primary data. The primary attributes like slope, catchment area, and specific catchment area are significant because they influence overland flow velocity and runoff rates, runoff volume, and steady state runoff rate respectively (Moore et al., 1991). The catchment area is a measure of the surface runoff of the landscape and it combines the effect of upslope contributing area as well as the local convergent and divergent flows within (Moore et al., 1991).

Secondary or compound data describe the spatial distribution and variability of specific processes that occur within a landscape such as wetness index (soil water content) or stream power (Moore et al., 1991). Wetness index is a second order derivative of slope and relates to the spatial distribution of soil saturation zones (Moore et al., 1991). Stream power is the erosive power of stream flow (Moore et al., 1991). The stream power index can be used to identify places where soil conservation practices that reduce the erosive power of the stream can be implemented (Moore et al., 1991).

Depending on the scale of regional planning, the fundamental units for water resource conservation are the basin, watershed or sub-watershed (Moore et al., 1991). The traditional mapping and delineation of watershed and sub watershed boundaries was centered on the stream network and therefore considered a conservative representation (Mark, 1983). The past decade has seen the rapid development in the field of hydrological modeling through the use of DEMs. The ARC GRID module uses an algorithm (O'Callaghan and Mark, 1984; Jenson and Domingue, 1988) that determines the flow direction of each element in 3x3 matrix to one of its eight neighbors in the direction of steepest descent (Moore et al., 1991).

# CHAPTER 3 METHODOLOGY

## 3.1 Surface hydrological modeling

### 3.1.1 Acquisition and pre-processing of SRTM derived DEMs

SRTM derived 30m DEMs were used to model surface flow in the MRW. These DEMs were downloaded from the USGS seamless web server (http://seamless.usgs.gov/) and merged to obtain coverage for the entire watershed. The grids were downloaded as floating point and in order to conserve disk space were converted to 16 bit unsigned integer by the use of the GRID command, INT (ESRI, 1994).

Errors in DEMs some times manifest in the form of "sinks" or "pits". These production artifacts are regions that have lesser elevation than the area around it. These sinks were identified with the SINK command in ARC GRID. The output grid was found to contain nodata cells caused during acquisition / production of the SRTM data. Cells that do not have a valid value assigned are termed as nodata cells (ESRI, 1994). The nodata cells were subsequently filled with a FOCAL MEAN command. After, the nodata cells were removed the individual grids (six in number) were projected UTM, Zone 16, datum & spheroid WGS 84. The SINK command (ESRI, 1994) was run again on the merged grid, as there might still be legitimate sinks of natural origin rather than artifacts due to the nature of radar backscatter. After the sinks had been identified, the FILL command, based on the algorithm developed by Jenson and Domingue (1988) was used to fill the artificial production artifacts.

### 3.1.2 Delineation of Strahler stream order using D-8 flowline derived stream network

In DEM analysis for hydrologic application, it is often assumed that all flow from a cell is directed towards one and only one of its neighbors and this assumption is referred to as the "deterministic eight-neighbors" or D-8 model (Fairfield and Leymarie, 1991). The D-8 flow-line algorithm is a standard algorithm that is used by the Environmental Systems Research Institute (ESRI) software to derive flow direction from a surface. The algorithm uses an input DEM to depict the direction of flow from each cell within a 3x3 window. There are eight valid directions or eight adjacent cells into which the flow can travel (ESRI, 1994). The flow direction determines the direction of flow from every cell or element in the grid and is determined by finding the direction of steepest flow (ESRI, 1994). The flow accumulation function computes accumulated flow as the weight of all the cells flowing into each down-slope cell in the output grid (ESRI, 1994). Cells with high flow accumulation have concentrated flow and are used to determine the stream network. A threshold value can be applied to the flow accumulation product to obtain the stream network within a watershed (Jenson and Domingue, 1988).

The surface hydrology commands such as flow direction & flow accumulation were run on the MRW grid (Fig 3-1). To obtain the stream network, a threshold value was derived by adding the mean of the flow accumulation grid to the standard deviation and re-classed such that, all values greater than the mean + 1 standard deviation would be 1 otherwise 0. This process was repeated iteratively until the raster stream network derived from the SRTM DEM matched vector stream networks in two different datasets namely Michigan Resource Information System (MIRIS) as well as Digital Line Graphs (planimetric information obtained from various USGS maps). The threshold number, 3326, was found to be near the mean of the flow accumulation distribution. The threshold number was then multiplied by 0.003x 0.003 to obtain the area required to generate a first order stream (2.99 sq km) within the Muskegon river watershed.

The STREAMORDER command (Strahler) was run on the output grid. In the Strahler stream order, order increases when streams of the same order intersect (Strahler, 1957). Therefore, the intersections of two first order streams will give rise to a second order stream and intersection of two second order streams would give rise to a third order stream. However, the intersections of two different orders will not result in increase in order. This method is the most commonly used model for stream networks (ESRI, 1994).

The stream order grids were then converted to arc coverages using the STREAMLINE command. The WATERSHED command, which uses the streamline arc coverages as well as the flow direction grid as input was used to generate the sub-watersheds. These watersheds differ slightly from the MIRIS watershed as the stream order was derived from various sources (digitizing of existing maps or from aerial photos).

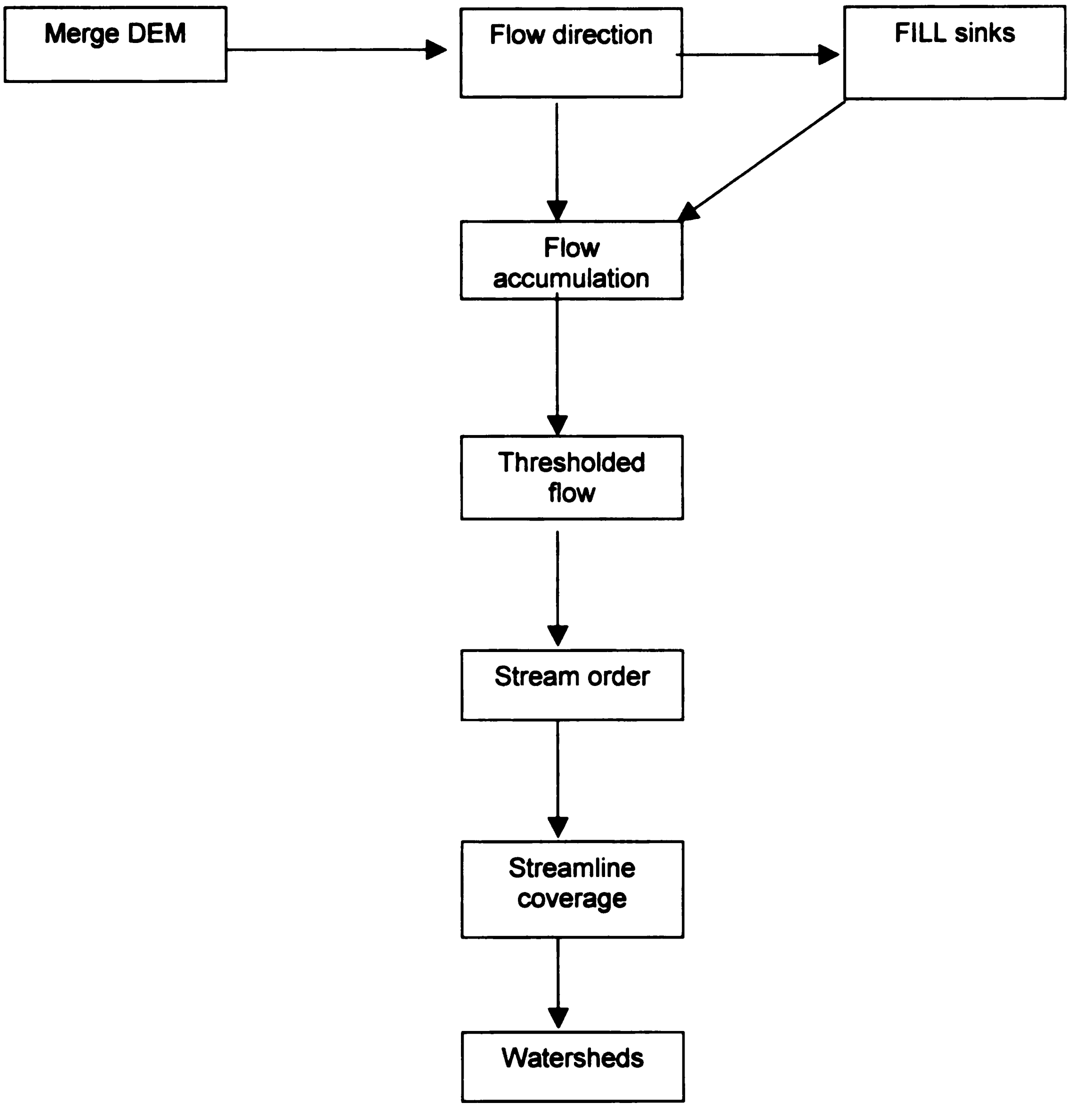


Figure 3-1. Flow chart to extract watersheds from DEM

### 3.1.3 Delineation of first order watersheds, percent slope and wetness index

In order to delineate the first order watersheds, a binary mask was applied on the Strahler stream order grid. The higher order streams were masked out leaving only the first order streams. The STREAMLINE command in ARC GRID was used on the first order stream grid along with the flow direction force grid to obtain an Arc-coverage of the first order streams. The WATERSHED command in ARC GRID was then run using the first order streams coverage and flow direction force grid. The first order watershed grid thus obtained was converted to a vector layer. The D-8 flow line algorithm models flow direction at $45^0$ angles and so the conversion of the raster watershed layer to a vector layer results in the formation of a number of spindle shaped polygon artifacts at the pour points (Figure 3-2). To correct for this anomaly, the first order watershed arc coverage was edited using the arc tools module in the ARC INFO software. The post processing of the first order watershed was time consuming, but absolutely necessary to delineate the first order watersheds within the sub–watersheds (Figure 3-3).

Figure 3-2. DEM flow direction artifacts created due to the nature of the D8 flow line algorithm

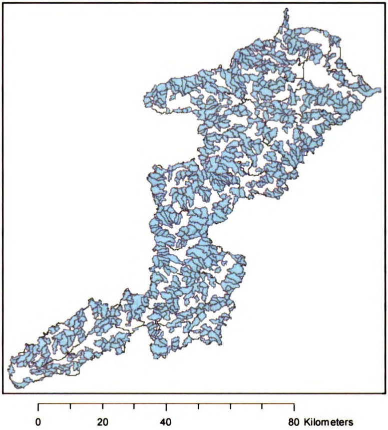


Figure 3-3. Muskegon first order watershed coverage obtained using binary mask of Strahler stream network

In addition to first order watersheds, percent slope and wetness index (W.I) were also obtained from the SRTM derived DEM. Slope identifies the maximum rate of change between each cell and its neighbors (ESRI, 1994). Percent slope was obtained using the SLOPE command using the percent rise option instead of degree slope and is useful in identifying areas of steep slope, which might be vulnerable to erosion.

The wetness index or topographic index is the log of the ratio of the catchment area and the percent slope (Wolock and Price, 1994). The wetness index helps in identification and delineation of wetlands (Moore et al., 1991). The equation for wetness index is

$$WI = \ln (a/\tan b)$$

$$\text{or } WI = \ln (a / (rise / run) + 0.0001) \qquad (3\text{-}1)$$

where a = flow accumulation
b = slope and tan b = slope percentage x 0.01

Implementing the wetness index for raster processing requires some adjustment. Wolock and Price (1994) suggest that flow accumulation or upslope contributing cells might be scaled by contour length, and in this case the cell side (Wolock and Price, 1994). The cell side was the resolution of an ETM+ pixel, i.e. 30m. In GRID, the wetness index was obtained using the following command

WI = LN ((FA + 1) * cell side) / (slope (percent rise) * 0.01 + 0.0001)

The wetness index was used to aid in the image classification of the ETM+ imagery by overlaying the wetness index grid over the classified image.

## 3.2 Classification of ETM+ imagery using unsupervised method

### 3.2.1 Imagery inventory

Four ETM+ (Landsat-7) images in the World Reference System (WRS-2) Path 21, Rows 29 & 30 and Path 22, Rows 29 & 30 were obtained from 2001 and 2002 (Table 3-1). The imagery was obtained from the TRFIC archive at the Center for Global Change and Earth Observations (CGCEO) at Michigan State University.

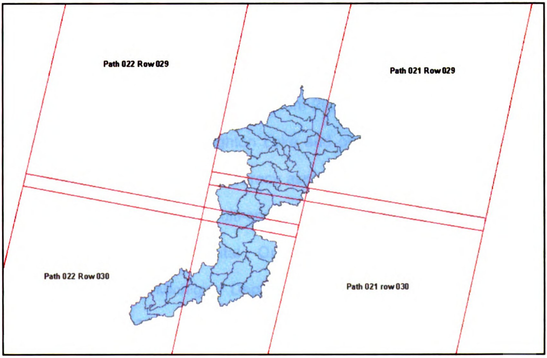


Figure 3-4. Spatial extent of MRW overlaid with WRS-2 (Landsat) footprint

| Landsat 7 | Date |
|---|---|
| ETM+ Path/row 22/29 | 9/7/2002 |
| ETM+ Path/row 22/30 | 7/2/2001 |
| ETM+ Path/row 21/29 | 6/25/2001 |
| ETM+ Path/row 21/30 | 7/14/2002 |

Table 3-1. Satellite data

The spatial extent of the MRW is such that it covers portions of four Landsat scenes (Figure 3-4). The portions of the ETM+ imagery mentioned above were then clipped using the subset option in ERDAS IMAGINE. The spatial database thus obtained includes subsets from 2001/2002 (ETM+) which would facilitate an inventory of the land use / land cover within the forty sub watersheds through the process of image classification.

In addition, two IKONOS-scenes from 2002 (September & October) were obtained over the MRW at Houghton Lake and over the Big Rapids-Haymarsh area to provide ground truth for accuracy assessment of the image classification process (Figure 3-6). The 4m multi-spectral data was pan-sharpened using the 1m panchromatic band. Digital Ortho-Quarter Quads (DOQQ's) were downloaded from the Michigan Department of Natural Resource's (MDNR), Spatial Data Library. The DOQQ's were then mosaicked and reprojected to UTM Zone 16 to be used as ground truth reference (Figure 3-5).

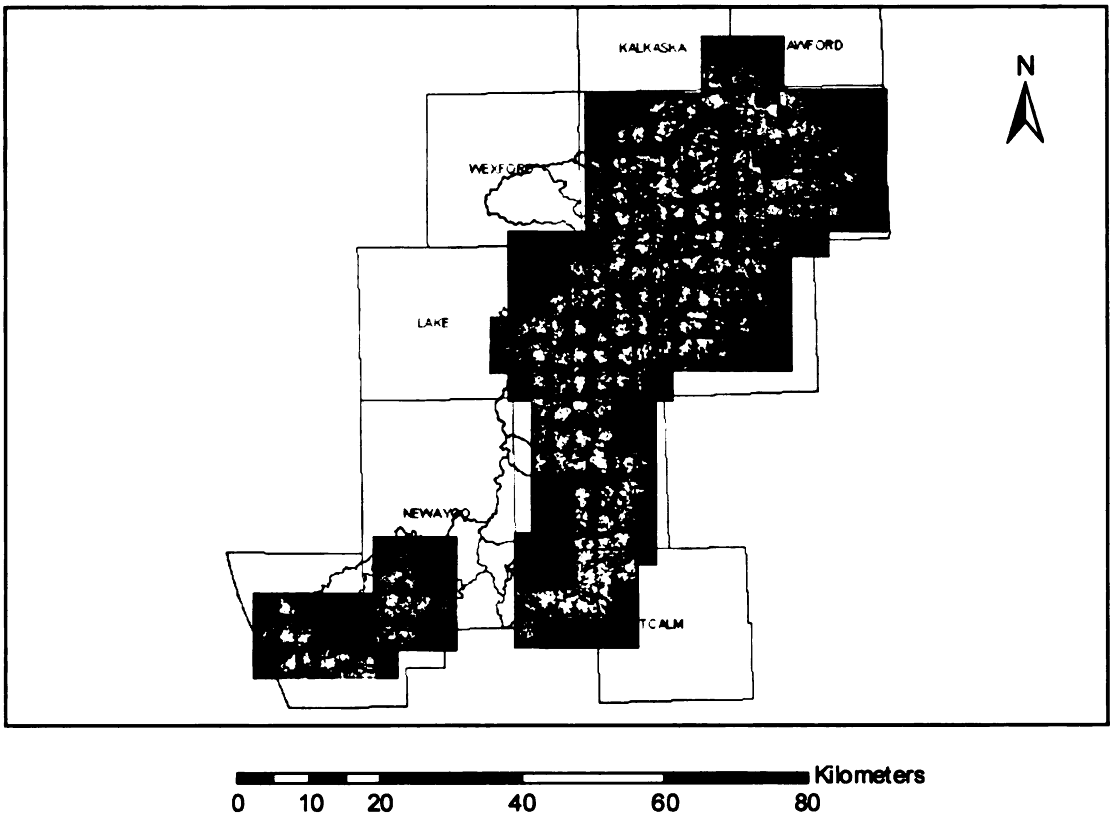


Figure 3-5. Spatial extent of MRW overlaid with county boundaries and Digital Ortho Quarter Quads (DOQQ's)

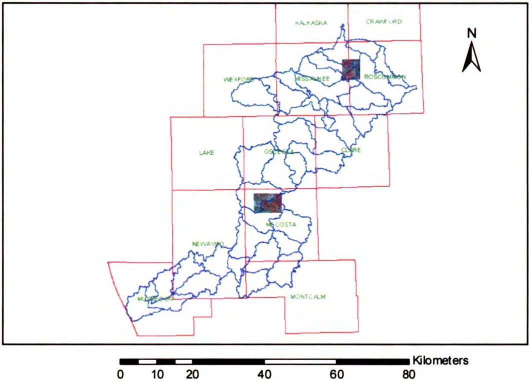


Figure 3-6. Spatial extent of MRW overlaid with county boundaries and pan sharpened IKONOS imagery

### 3.2.2 Image pre-processing and unsupervised classification

The Landsat data record is extremely important to earth sciences as it marks over three decades of earth observation. Its advantages include a high spatial resolution, medium temporal resolution, and an extensive swath (Teillet et al., 2001). However, in order to benefit from this impressive dataset, it is absolutely essential to process the digital numbers collected by the sensor by radiometric calibration to an absolute scale, in physical units (Teillet et al., 2001). Image pre-processing and radiometric correction was carried using ERDAS IMAGINE version 8.6 (Fig 3-7).

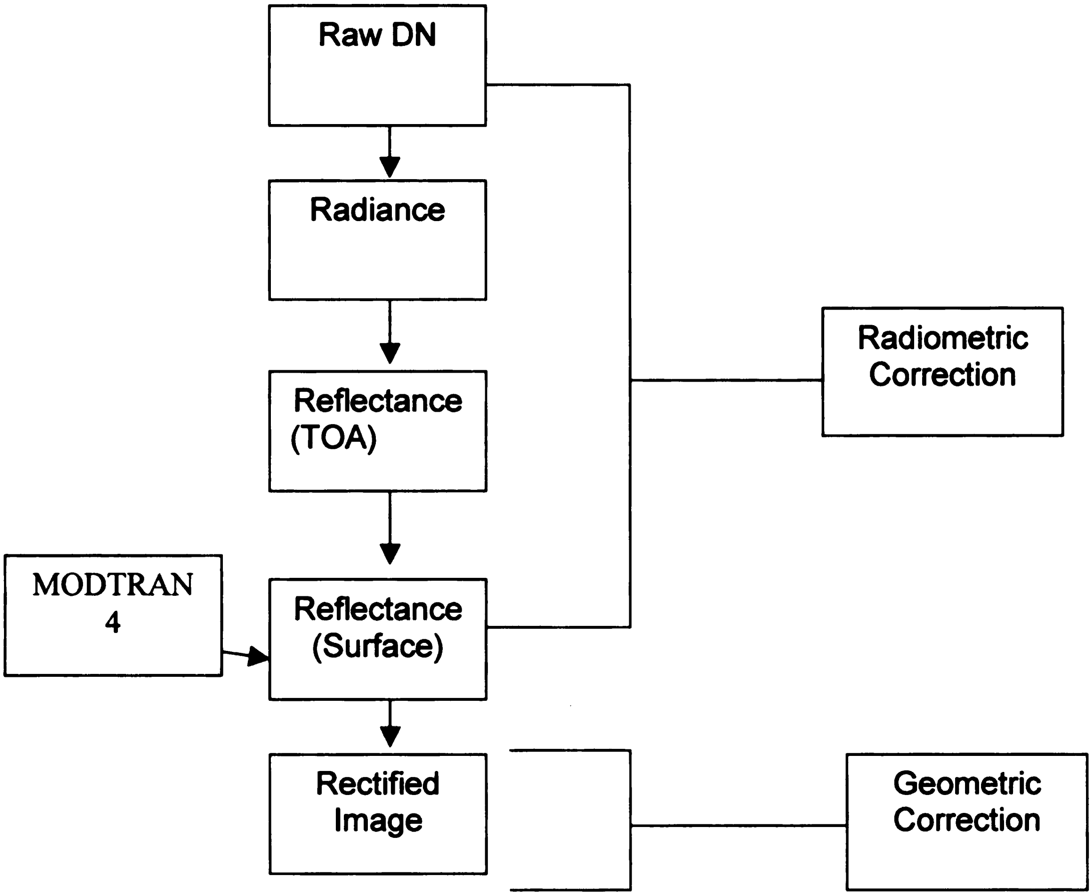


Figure 3-7. Flow chart for image processing of DN to surface reflectance

The raw digital numbers are first converted into physical units of radiance ($L_\lambda$) using calibration coefficients (Table 3-2) in a given spectral band

$$L_\lambda = [(LMAX - LMIN)]/255] \times DN + LMIN \tag{3-2}$$

Where LMAX = spectral radiance scaled to $DN_{max}$ in w/ ($m^2$.sr.μm)

LMIN = spectral radiance scaled to $DN_{min}$ in w/ ($m^2$.sr.μm)

DN = Digital Numbers

| BAND | ETM+ in w/(m$^2$.sr.$\mu$m) | |
|---|---|---|
| | LMAX | LMIN |
| 1 | 191.6 | -6.2 |
| 2 | 196.5 | -6.4 |
| 3 | 152.9 | -5.0 |
| 4 | 241.1 | -5.1 |
| 5 | 31.06 | -1.0 |
| 7 | 10.80 | -0.35 |

Table 3-2. ETM+ spectral radiance range in w/ (m$^2$.sr.$\mu$m)

When there is a need to compare imagery from different sensors, it is advantageous to use reflectance values instead of using radiance. The cosine effect of different sun angles can be removed and the differences due to variability in exo-atmospheric radiances between spectral bands can also be accounted for. Unitless planetary reflectance was obtained by the following equation

$$\rho = \frac{\pi \bullet L_\lambda \bullet d^2}{Esun_\lambda \bullet \cos\theta_s} \tag{3-3}$$

where $\rho_p$ = Unitless planetary reflectance
$L_\lambda$ = Spectral radiance at the sensor's aperture
$\theta_s$= Solar zenith
d = Earth-sun distance in astronomical units
Esun $_\lambda$ = Mean solar exo-atmospheric irradiances from table below

| Band | ETM+ Solar Spectral Irradiances Units: in $w/(m^2.sr.\mu m)$ |
|---|---|
| 1 | 1970.00 |
| 2 | 1843.00 |
| 3 | 1555.00 |
| 4 | 1047.00 |
| 5 | 227.00 |
| 7 | 80.53 |

Table 3-3. ETM+ solar spectral irradiances

The ETM+ radiance values were radiometrically corrected to at-sensor reflectance using the calibration parameters in the metadata and then to surface reflectance after correcting for distorting atmospheric effects. Haze in the upper atmosphere often causes satellite imagery to look faded due to Rayleigh scattering and ozone absorption. The top of the atmosphere (TOA) reflectance was then converted into surface reflectance by using the coefficients of slope and intercept, which in turn were obtained by running the MODTRAN 4 program.

Geodetic accuracy or the geographic navigation accuracy for a particular pixel is absolutely essential especially in the case of change detection studies. The ETM+ level 1G imagery products are systematically corrected, i.e., registration is performed without ground control (Goward et al., 2001). After systematic correction, ETM+ imagery has a geodetic error of less than 250m with 1 standard deviation (Masek et al., 2001). The ETM+ images were geometrically corrected to ensure proper alignment and scale and were rectified to the Universal Transverse Mercator (UTM) projection (Zone 16) and WGS 84 datum using a first order polynomial and nearest neighbor resampling. To correct for horizontal displacement and inaccuracy, GPS co-ordinates recorded at the road intersections within the watershed were used as ground control Points (GCPs) to serve as a reference in the process of geometric correction to geo-reference the imagery. The accuracy was within one pixel (Table 3-4).

| L-7 Path /Row | X Residual | Y Residual | RMS Error |
|---|---|---|---|
| 22/30 (ETM+) | 0.0678 | 0.0510 | 0.0849 |
| 21/30 (ETM+) | 0.0676 | 0.2258 | 0.2357 |
| 21/29 (ETM+) | 0.1882 | 0.3466 | 0.3944 |
| 22/29 (ETM+) | 0.2024 | 0.2024 | 0.2862 |

Table 3-4. ETM+ scenes and their geo-referencing parameters

An unsupervised classification was carried out on the ETM+ scenes through the use of the Iterative Self-Organizing Data Analysis Technique (ISODATA) algorithm. The ISODATA clustering method uses minimum spectral distance to iteratively classify pixels until the spectral distance patterns emerge in the form of clusters (Tou and Gonzalez, 1974). To perform ISODATA classification, the user needs to specify the maximum number of clusters thought necessary , the number of iterations to be run and the convergence threshold, which defines the maximum percentage of pixels that are allowed to remain unchanged between iterations.

The ISODATA algorithm was run on the ETM+ scenes after specifying 100 clusters, 80 iterations, and a convergence threshold of 0.98. The National Land Cover Database (NLCD 92) thematic map was created by an unsupervised classification using 100 clusters and so the same number of clusters was used in the generation of a classified product for this research (Vogelmann et al., 1998).

The classification was performed after geo-referencing so that the resultant thematic map would be more accurate (ERDAS, 2004).

Each image reached the convergence threshold before the maximum level of iterations. Visual interpretation helped in identification of eight distinct LULC classes from the Michigan Land Cover / Use Classification System (2000), namely 1) deciduous, 2) coniferous, 3) grass, 4) water, 5) wetland, 6) bare soil, 7) urban, and 8) agriculture. These classes were then recoded to an Anderson's level I classification (Anderson, 1976) with seven classes, 1) urban, 2) agriculture, 3) grass, 4) forest, 5) water, 6) wetland and 7) bare soil.

The seven classes were defined as follows:

1) Urban: Developed areas that were characterized by 30 % or more area under constructed material (Asphalt & concrete).
2) Agriculture: Areas characterized by croplands, which include row crops (corn & soybean), small grain (wheat & barley) as well as pasture/hay.
3) Grass and shrublands: These include areas that have 25% or more area covered by grasses and shrubs
4) Forest: Areas that had 20% or more covered by broadleaved forest as well as coniferous forests.
5) Water: Areas under open water

6) Wetlands: Areas where the soil or substrate is periodically saturated with or covered with water (Cowardin et al., 1979). These areas are characterized by both forested and non-forested wetlands.
7) Bare: Area covered by bare rock, silt, sand or clay with little or no vegetation.

The accuracy assessment of the ETM+ derived LULC maps through image classification required the comparison of the classified imagery with reference or ground truth data (Congalton, 1988). The high- resolution (1m) pan-sharpened IKONOS images and DOQQ's were used as ground truth images and as a photo-interpretation key in the accuracy assessment process. The accuracy assessment utility in ERDAS IMAGINE 8.6 was used to generate 300 random points with 40 or more points per class using stratified random sampling. Congalton (1991) suggests that a minimum of 40-50 samples be collected for each land cover in the error matrix to adequately sample the dataset to be evaluated. Random sampling was used in an earlier assessment but was replaced by a stratified random sampling scheme as the former tends to under sample small but important classes such as urban areas in watersheds. It is for this reason that stratified random sampling was chosen where a minimum number of sample points (40-50) were collected for each class type (Congalton, 1988).

The randomly sampled co-ordinates were imported into Arcview 3.2 as a tab delimited file and saved as a point shapefile. This shapefile was overlaid over the high-resolution reference images, namely the 2002 IKONOS images and the Michigan DNR DOQQ's (1998). The reference targets were identified using visual interpretation and entered in the reference column of the accuracy assessment utility.

The relationship between the classified map product and the reference images are summarized in an error matrix (Table 3-4). The overall classification accuracy or the percentage of correctly classified pixels of the LULC product was 83.33% and was obtained by dividing the sum of the (correctly classified) pixels in the major diagonal by the total number of samples (Table 3-5). The producers accuracy (which is a measure of the reliability of the map) was derived by the total number of correct pixels for a land cover type divided by the total number of pixels of that land cover type derived from reference data, in other words, the column total. The users accuracy (a measure of the adequacy for each category) was derived by dividing the total number of correct pixels for a land cover type divided by the total number of pixels that were accurately classified in that land cover class. The Kappa ($K_{hat}$) statistic, a measure of percent accuracy within an overall measurement of classifier accuracy (Congalton, 1991), was 0.8054. The Kappa statistic is a better measure of accuracy than an overall assessment as it considers inter class agreement (Fitzgerald and Lees, 1994).

$$K_{hat} = \frac{N\sum_{i=1}^{r} x_{ii} - \sum_{i=1}^{r}(x_{i+} \times x_{+i})}{N^2 - \sum_{i=1}^{r}(x_{i+} \times x_{+i})} \tag{3-4}$$

where r is the number of rows in the matrix, N is the total number of observations, $x_{ii}$ is the number of observations in row *i* and column *i*, $x_{i+}$ and $x_{+i}$ are the marginal totals for row *i* and column *i* respectively.

| Class Name | Reference Totals | Classified Totals | Number Correct | Producers Accuracy | Users Accuracy | Kappa Statistic |
|---|---|---|---|---|---|---|
| Urban | 31 | 41 | 26 | 83.87% | 63.41% | 0.592 |
| Agriculture | 46 | 43 | 32 | 69.57% | 74.42% | 0.6979 |
| Herbaceous | 46 | 44 | 41 | 89.13% | 93.18% | 0.9195 |
| Forest | 51 | 48 | 46 | 90.20% | 95.83% | 0.9498 |
| Water | 40 | 41 | 40 | 100.00% | 97.56% | 0.9719 |
| Wetland | 39 | 43 | 33 | 84.62% | 76.74% | 0.7327 |
| Bare soil | 47 | 40 | 32 | 68.09% | 80.00% | 0.7628 |

Table 3-5. Accuracy assessment for MRW LULC mosaic

| | Reference | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|
| Class | Urb. | Ag. | Herb. | For. | Wat. | Wet. | Ba. | Row | UA |
| U. | 26 | 7 | 0 | 0 | 0 | 0 | 8 | 41 | 63.41 |
| Ag. | 1 | 32 | 1 | 0 | 0 | 2 | 7 | 43 | 74.42 |
| Herb. | 0 | 2 | 41 | 0 | 0 | 1 | 0 | 44 | 93.18 |
| For. | 0 | 0 | 0 | 46 | 0 | 2 | 0 | 48 | 95.83 |
| Wat. | 0 | 0 | 0 | 0 | 40 | 1 | 0 | 41 | 97.56 |
| Wetl. | 0 | 1 | 4 | 5 | 0 | 33 | 0 | 43 | 76.74 |
| Bare. | 4 | 4 | 0 | 0 | 0 | 0 | 32 | 40 | 80 |
| col tot | 31 | 46 | 46 | 51 | 40 | 39 | 47 | 250 | Sum |
| Prod | 83.87 | 69.56 | 89.1 | 90.19 | 100 | 84.61 | 68.08 | 300 | Total |

Table 3-6. Confusion matrix for MRW LULC mosaic

The reference totals column (Table 3-6) refers to the number of pixels in each land cover identified within the high-resolution reference imagery. The classified totals are the number of pixels in each land cover classification mosaic from 2001. The "number correct" is the number of pixels in the classification that match the high-resolution reference imagery. Overall classification accuracy produced from this assessment was 83.3% with an overall kappa statistic of 0.80.

The errors of omission and commission were most apparent within the urban, agriculture, grassland and wetland land cover types. Recently harvested agricultural fields exhibit high reflectance, are similar to highly reflective surfaces within an urban area. The construction materials in urban structures like cement and concrete have similar spectral properties to the bare soil in an agricultural area that has been recently harvested. Therefore, an agricultural plot can often be mistaken for a urban area owing to the spectral properties Similarly there were instances of omission and commission between fallow fields that exhibited re-growth and land with 50% or less herbaceous cover. Another land cover type that had errors of omission and commission was the wetland cover type. Lowland forests that are seasonally flooded can be labeled as forest or wetland depending the date of acquisition. Leafon imagery resulted in a spectral signature that was representative of forests whereas a leafoff imagery especially in late fall resulted in a classified product that depicted a forested wetland. The labeling of the unsupervised clusters to wetlands was verified by overlaying the image with a vector layer of wetlands obtained from the MNDR Spatial Library.

### 3.2.3 LULC within MRW

The classified imagery from the ETM+ scenes was converted into Arc Grids using the Import/Export module in ERDAS IMAGINE. The ARC GRID files were overlaid with an AOI (Area of Interest) of the sub-watershed and the catchment area under each LULC type was obtained and tabulated. The raw count was converted into an area measurement by multiplying the raw count with 0.09 (the area of the pixel in hectares) for conversion into hectares. The area in hectares was then converted into percentage values to obtain the proportion of land use/land cover within the sub-watershed. The minimum mapping unit (MMU) was defined by size of the 3x3 majority filter (nine 30m pixels) used to remove the salt and pepper effect caused due to noise in the imagery. The proportion of land use and land cover in the Muskegon River Basin is 3.677% urban, 18.11 % agricultural, 40.5% forested, 15.89 % under herbaceous, 0.26 % bare, 4.06 % water and 17.48% consists of wetlands.

| | 2001<br>Area (in ha) | 2001<br>Area (in %) |
|---|---|---|
| Urban | 26010.18 | 3.677342 |
| Agriculture | 128145.24 | 18.11729 |
| Herbaceous | 112410.45 | 15.89269 |
| Forest | 286471.62 | 40.50161 |
| Water | 28754.46 | 4.065331 |
| Wetland | 123643.35 | 17.48081 |
| Bare soil | 1873.89 | 0.264932 |
| Total | 707309.19 | 100 |

Table 3-7. LULC within MRW

### 3.2.4 Mapping of surface water quality indicators

Surface water quality indicators were collected by teams of student and research assistants lead by Dr. Jan Stevenson, Department of Zoology, Michigan State University. Between 2001 and 2003, 256 sites were sampled in spring and summer seasons. These sites include 107 streams/rivers, 85 wetlands and 64 lakes. The water quality measurements include, total nitrogen (TN), total phosphorus (TP), trophic state index based on phosphorus (TSIP), trophic state Index based on cholorophyll (TSIchl), trophic state Index based on secchi depth (TSIsd) and specific conductivity.

Biological indicators such as total insect population, sensitive insect population (EPT taxa), species richness of Algae, zooplankton, invertebrates, fish, mussels and plants were also collected at various sampling locations that were re-visited every year (2001-2003). Each water quality measurement had geographic co-ordinates (WGS 84 datum) recorded by a GPS at the time of collection. The water quality measurements were entered in MS Excel worksheets and saved either as .txt files or .dbf files. The water quality measurements were then imported into ESRI Arc Map using the add XY interface. The water quality measurements were mapped in geographic co-ordinates and then projected into UTM zone 16 co-ordinates (datum WGS 84) so that they could be overlaid on the land use / land cover layer.

### 3.2.5 Proximity to LULC types

The proximity to urban and agricultural land use was calculated using the Euclidean distance function in ARCGRID (ESRI, 1994). Binary raster layers that represented urban areas and agricultural land use were used as source Grids The Euclidean distance surfaces this obtained were overlaid with total nitrogen / total phosphorus sampling points to obtain the distance to urban areas and agricultural land use for each sampling point.

### 3.2.6 Scale issues in relating LULC to water quality

In this study, two spatial scales were considered. One was the sub-watershed level study where the LULC in hectares was aggregated to the sub-catchment level and the other being at the level of the first order watersheds. This methodology was based on the premise that hydrological units, namely the sub-watersheds and the first order watersheds would delimit sampling locations and define the areas that influence the measurements taken at those locations.

In addition, the immediate vicinity around the sampling points was correlated with water quality indices through the creation of variable buffers of arbitrary width. This approach did not yield significant results when the width of the buffers was small. Increasing the buffer size beyond 500m–1km meant that areas upstream as well downstream would be assumed to influence the nutrient concentration at the sampling point. This methodology was discontinued since areas that were 500m–1 km downstream of the sampling location would not have a significant impact on the nutrient concentration.

There have been many studies with opposing views on the scale at which land-use and water interactions should be studied. Some suggest that TP concentrations in stream water could be explained by LULC patterns within 100m of a stream while other studies (Omernik et al., 1981) demonstrated that the more distant LULC types at the catchment extent had an impact on water quality along with riparian land use in the immediate vicinity. Comparisons between different case studies are complicated owing to differences between nutrient concentrations, the size of the watershed, and the temporal nature of the measurements (Turner et al., 2001).

## CHAPTER 4 RESULTS AND DISCUSSION

### 4.1 LULC and water quality in the MRW

#### 4.1.1 Specific conductivity and percentage of LULC

Specific conductivity is the measure of the ability of any solution to carry an electric current and increases with increasing mineral content (Prevost et al., 2000). Spearman's rank correlation was used to test for correlations between the percentages of land use and specific conductivity. Specific conductivity within the 40 sub-watersheds was assumed to be positively correlated with the percentages of urban land use, agricultural land, combined percentages of urban and agricultural land as well as the percentage of human use index (combined percentages of urban, agricultural and bare land). It was also assumed that forest cover and specific conductivity measurements were negatively correlated.

| Conductivity vs. LULC | Correlation | *t* value | *t* critical, $\alpha$ = 0.05 |
|---|---|---|---|
| Conductivity vs. urban | 0.606 | 4.5069 | 1.695 |
| Conductivity vs. agriculture | 0.533 | 3.7267 | 1.695 |
| Conductivity vs. urban + agriculture | 0.725 | 6.2274 | 1.695 |
| Conductivity vs. Human Use Index | 0.720 | 6.1379 | 1.695 |
| Conductivity vs. forest | -0.358 | -2.2683 | -1.695 |

Table 4-1. Correlation between specific conductivity (µ siemens) and LULC types

As hypothesis, it was assumed that specific conductivity would increase as the percentage of anthropogenically modified LULC within the sub-watershed and first-watersheds increases. The null hypothesis was that there is no correlation between the two variables ($t \leq$t-critical at $\alpha$ = 0.05). As alternative hypothesis, it was assumed that there is a positive correlation between specific conductivity and the percentage of land use ($t >$ t-critical at $\alpha$ = 0.05). The t critical value = 1.695 at $\alpha$ = 0.05 (95% probability) and df = 34.

$H_o$: Null hypothesis: there is no correlation between specific conductivity and the percentage of land use

$H_1$: Alternative hypothesis: there is a correlation between specific conductivity and the percentage of land use

The null hypothesis was rejected as the *t* values for the different land use / land cover types (table 4-1) were greater than t critical value = 1.695. The correlation between urban, agricultural LULC types as well as the percentage of combined land use and specific conductivity, which ranged between 0.60 and 0.72 was statistically significant at the 95 % confidence level. This could be attributed to the percentage of urban land use greater than 1% in some of the watersheds (Table 4-2).

| Sub-watershed | City/Town |
|---|---|
| 1 | Higgins lake/I -127 |
| 3 | Houghton/I -127 |
| 12 | Cadillac |
| 14 | Lake city |
| 23 | Reed city/Evart |
| 24 | Big rapids/US 31 |
| 25 | Big rapids/US 31 |
| 31 | Howard city |
| 35 | Newaygo (Brooks & Hess lakes) |
| 36 | Freemont |
| 38 | Muskegon wastewater ponds |
| 39 | Muskegon city |
| 40 | Muskegon city |

Table 4-2. Urban land use in sub-watersheds greater than 1%

| | Model summary | | | | | Parameter estimates | |
|---|---|---|---|---|---|---|---|
| Equation | R Square | F | df1 | df2 | Sig. | Constant | b1 |
| Linear | .205 | 8.789 | 1 | 34 | .006 | 329.099 | 11.040 |

Table 4-3. Correlation between specific conductivity (μ siemens) and urban land use within sub-watershed

A simple linear regression characterized a positive linear association between the percentage of urban land use (independent variable) and specific conductivity, the dependent variable (Table 4-3). The model explained 20 % of the dependent variable (specific conductivity) at $\alpha = 0.05$.

### 4.1.2 TN concentration and percentage of LULC

Total nitrogen concentrations (measured in parts per billion) within the sub-watershed were assumed to be positively correlated (using Spearman's rank correlation) with the percentages of urban land use, agricultural land, combined percentage of urban and agricultural land as well as percentage of human use index (combined percentages of urban, agricultural and bare land). It was also assumed that the percentage of forest cover and TN concentrations were negatively correlated.

| TN vs. LULC | Correlation | *t* value | *t* critical, $\alpha = 0.05$ |
|---|---|---|---|
| TN vs. urban | 0.189 | 1.1386 | 1.695 |
| TN vs. agriculture | 0.4 | 2.5819 | 1.695 |
| TN vs. urban + agriculture | 0.454 | 3.0144 | 1.695 |
| TN vs. human use index | 0.464 | 3.0988 | 1.695 |
| TN vs. forest | -0.47 | -3.1501 | -1.695 |

**Table 4-4. Correlation between TN concentrations and LULC within sub-watershed**

| TN vs. LULC | Correlation | *t* value | *t* critical, $\alpha = 0.05$ |
|---|---|---|---|
| TN vs. urban | 0.169 | 1.014 | 1.695 |
| TN vs. agriculture | 0.364 | 2.2833 | 1.695 |
| TN vs. urban + agriculture | 0.351 | 3.0228 | 1.695 |
| TN vs. human use index | 0.360 | 2.9811 | 1.695 |
| TN vs. forest | -0.437 | -3.4339 | -1.695 |

Table 4-5. Correlation between TN concentrations and LULC within first order watershed

In addition, TN concentrations within the first order watersheds were assumed to be positively correlated with the percentages of urban and agricultural LULC types as well as their combined percentages. It was also assumed that the TN concentrations were negatively correlated with the percentage of forests within the first order watersheds.

As hypothesis, it was assumed that TN concentrations increase as the percentage of anthropogenically modified land within the sub watershed and first-watersheds increases. The null hypothesis was that there is no correlation between the two variables ($t \leq$t-critical at $\alpha = 0.05$). As alternative hypothesis, it was assumed that there is a positive correlation between TN concentrations and the percentage of land use ($t >$ t-critical at $\alpha = 0.05$). The t critical value = 1.695 at $\alpha = 0.05$ (95% probability) and df = 34.

$H_0$ null hypothesis: there is no correlation between TN concentrations and the percentage of land use

$H_1$ alternative hypothesis: there is a correlation between between TN concentrations and the percentage of land use

The null hypothesis was rejected as the *t* values for the different land use / land cover types (table 4-4) were greater than t critical value = 1.695. However, the correlation between the percentage of urban land (within the sub-watershed as well the first order watershed) and increasing TN concentrations were not statistically significant at the 95 % confidence level.

Logarithmic regression was undertaken to explain the relationship and variance in a regression model between the percentage of LULC (dependent variable) and TN concentrations (dependent variable). A non-linear regression was considered, as a linear least square regression did not explain any variance. A logarithmic model was fitted to TN concentrations and the percentage of urban use within the sub-watersheds which yielded an $R^2$ = 0.003 (Table 4-6). The model could only explain 0.03% of the dependent variable (TN concentrations) and was also not statistically significant at the 95 % confidence level.

F critical value = 4.13 at $\alpha$ = 0.003 (95 % probability) and $df$ = 35

F statistic = 0.091, therefore 0.091 < 4.13 or F < F critical

| Independent | Logarithmic Model for total nitrogen (dependent) | | | | | Parameter estimates | |
|---|---|---|---|---|---|---|---|
| % LU | R Square | F | df1 | df2 | Sig. | Constant | b1 |
| urban | .003 | .091 | 1 | 34 | .764 | 25.006 | 6.740 |
| agriculture | .024 | 0.847 | 1 | 34 | .364 | 1351.463 | 253.085 |
| urb. + ag. | .017 | 0.584 | 1 | 34 | 0.584 | 916.198 | 358.346 |
| H.U. Index | .016 | 0.538 | 1 | 34 | .468 | 944.906 | 345.990 |
| forest | .024 | 0.851 | 1 | 34 | .363 | 6527.997 | 1264.905 |

Table 4-6. Percentage of LULC (within sub-watersheds) vs. TN concentrations

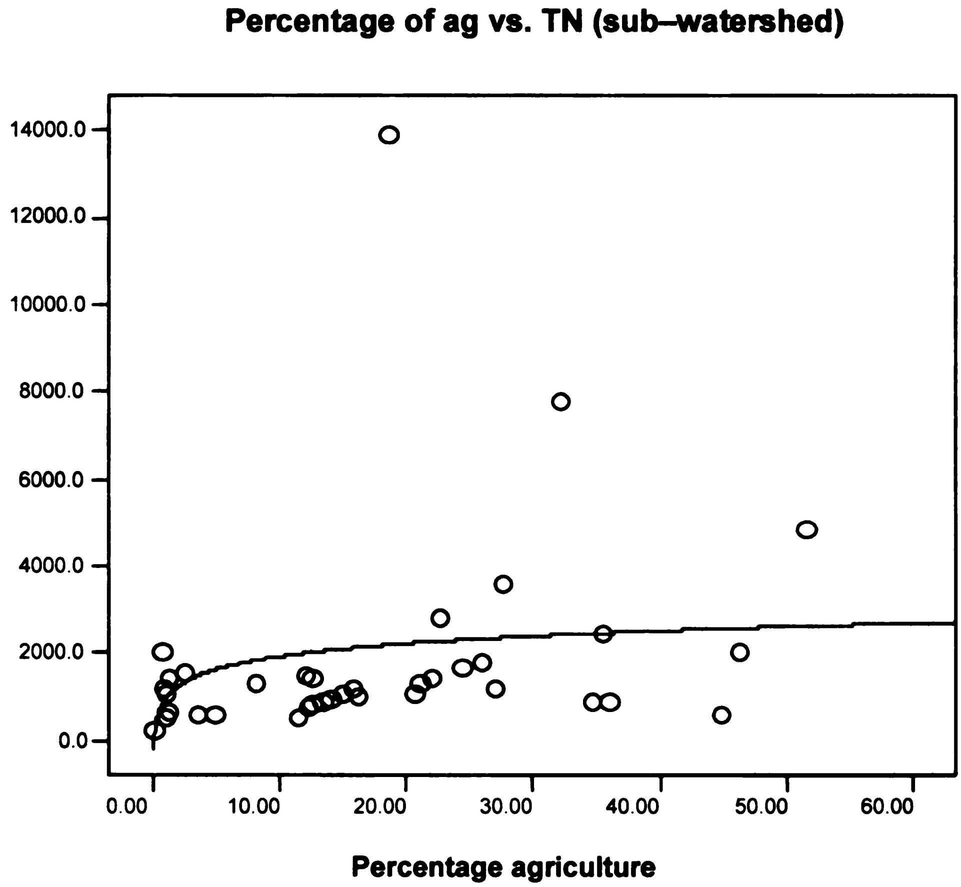


Figure 4-1. Percentage of agricultural land use (within sub-watersheds) vs. TN concentrations

Then, a logarithmic model fitted to TN concentrations and the percentage of agricultural land use within the sub-watersheds, yielded an $R^2 = 0.024$ (Figure 4.1). The model explained only 2.4 % of the dependent variable (total nitrogen) and was not statistically significant at $\alpha = 0.05$ (Table 4-6).

F critical value = 4.13 at $\alpha = 0.05$ (95 % probability) and *df* = 35

F statistic = 0.847, therefore 0.847 < 4.13 or F statistic < F critical

When the percentages of urban and agricultural land use (Figure 4-2) were summed and regressed with the TN concentrations in the sub watersheds, the model explained 1.7 % of the dependent variable (total nitrogen) and was not statistically significant (Table 4-6).

F critical value = 4.13 at $\alpha = 0.05$ (95 % probability) and *df* = 35

F statistic = 0.584, therefore 0.584 < 4.13 or F statistic < F critical

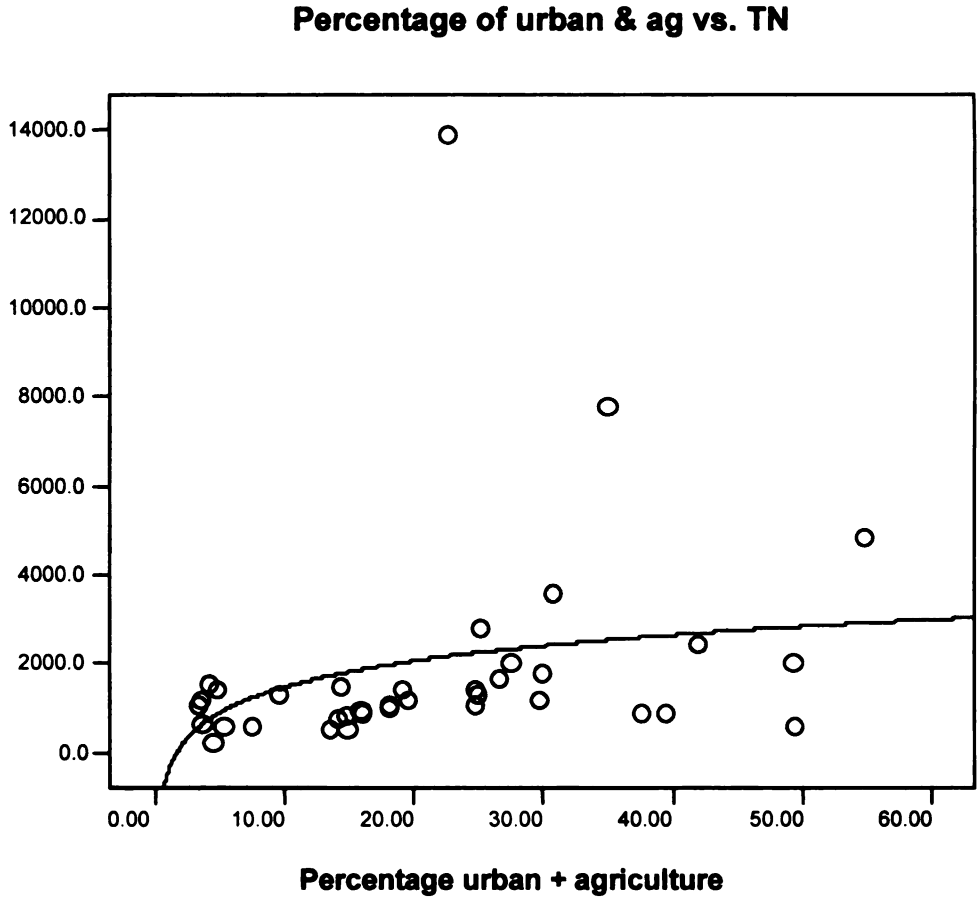


Figure 4-2. Percentage of agricultural & urban land use (within sub-watersheds) vs. TN concentrations

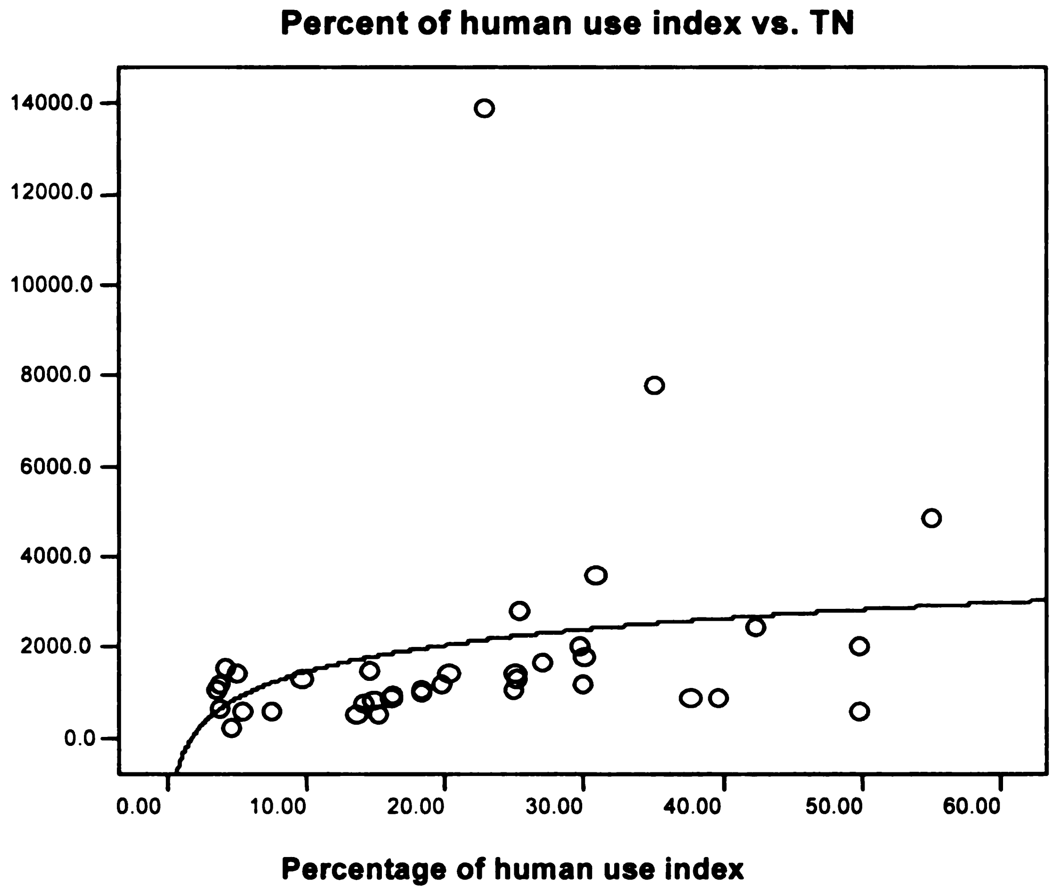


Figure 4-3. Percentage of human use index (urban + agriculture + bare soil within sub-watersheds) vs. TN concentrations

The percentage of human use index, i.e., the combined percentages of urban, agricultural and bare-soil, was regressed with TN concentration (Figure 4-3). However, the model did not significantly explain 1.6 % of the dependent variable, total nitrogen (Table 4-6).

F critical value = 4.13 at $\alpha = 0.05$ (95 % probability) and *df* = 35

F statistic = 0.538, therefore $0.538 < 4.13$ or F statistic < F critical

Logarithmic regression was also undertaken to explain the relationship and variance in a regression model between the percentages of land use (independent variable) and TN concentrations (dependent variable) aggregated to the first order watershed level. The percentages of land use types include the percentages of urban land use, agricultural land, the combined percentage of urban and agricultural land as well as the percentage of human use index (combined percentages of urban, agricultural and bare land).

| Independent | Logarithmic model for total nitrogen (dependent) | | | | | Parameter estimates | |
|---|---|---|---|---|---|---|---|
| % LU-first | R square | F | df1 | df2 | Sig. | Constant | b1 |
| urban | .000 | .000 | 1 | 35 | .990 | 1895.525 | 9.586 |
| agriculture | .175 | 7.223 | 1 | 34 | .011 | 11.760 | 8.444 |
| urban + ag. | .071 | 2.675 | 1 | 35 | .111 | -380.810 | 790.953 |
| h.u. index | .071 | 2.666 | 1 | 35 | .111 | -431.927 | 803.948 |
| Forest | .049 | 1.791 | 1 | 35 | .189 | 8007.71 | -1645.04 |

Table 4-7. Percentage of LULC (within first order watersheds) vs. TN concentrations

A logarithmic model (Table 4-7) fitted to TN concentrations and the percentage of urban land use within the first order-watersheds yielded an $R^2$ = 0.000. The model could not explain any variance in the dependent variable (TN concentration) and was not statistically significant at the 95 % confidence level.

F critical value = 4.13 at $\alpha$ = 0.003 (95 % probability) and *df* = 35

F statistic = 0.000, therefore 0.000 < 4.13 or F < F critical

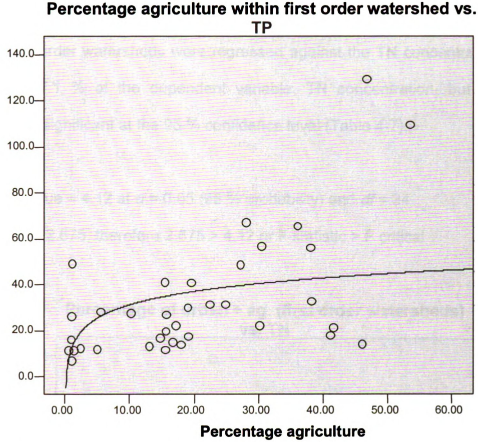


Figure 4-4. Percentage of agricultural land use (within first order-watersheds) vs. TN concentrations

F critical value = 4.13 at $\alpha = 0.05$ (95 % probability) and $df = 35$

F statistic = 7.223, therefore 7.223 > 4.13 or F statistic > F critical

Another logarithmic model fitted to TN concentrations and the percentage of agricultural land use within the first order-watersheds yielded an $R^2$ of 0.175 (Figure 4-4). The model explained 17% of the dependent variable, TN concentration, but was not statistically significant at the 95 % confidence level (Table 4-7).

The combined percentages of urban and agricultural land use (Figure 4-5) in the first order watersheds were regressed against the TN concentrations and explained 7.1 % of the dependent variable, TN concentration, but was not statistically significant at the 95 % confidence level (Table 4-7).

F critical value = 4.12 at $\alpha$ = 0.05 (95 % probability) and *df* = 34

F statistic = 2.675, therefore 2.675 > 4.12 or F statistic > F critical

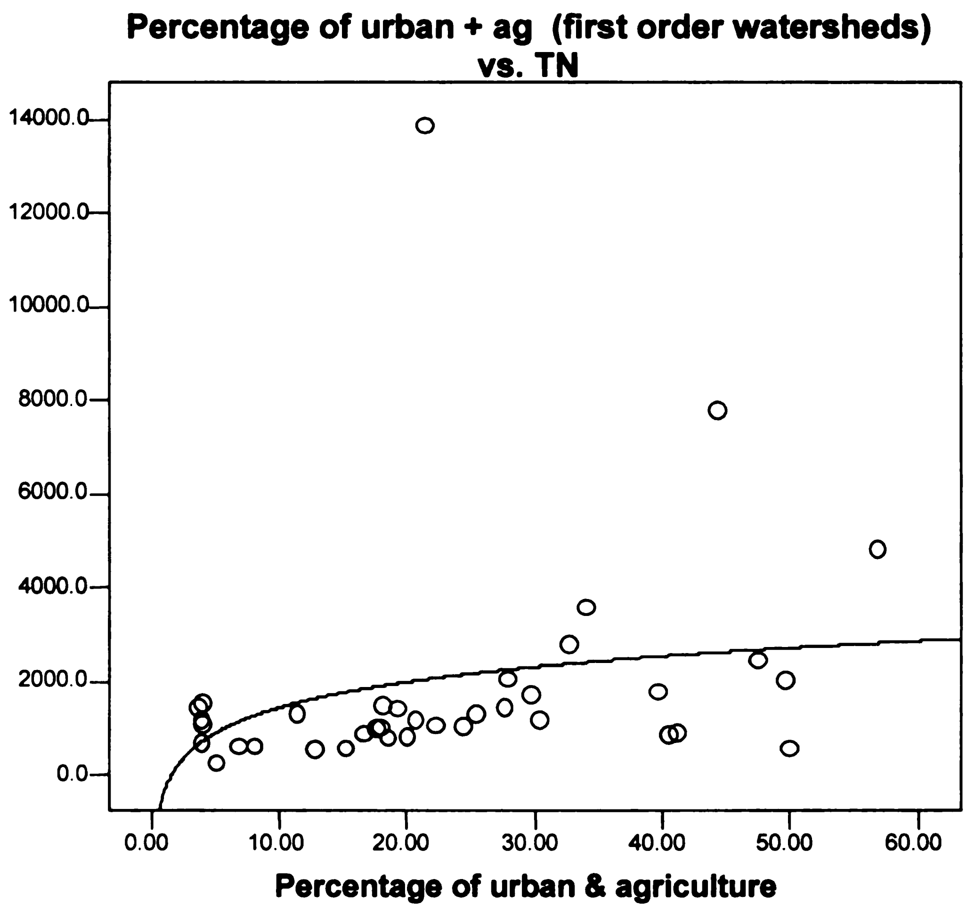


Figure 4-5. Percentage of agricultural & urban land use (within first order watersheds) vs. TN concentrations

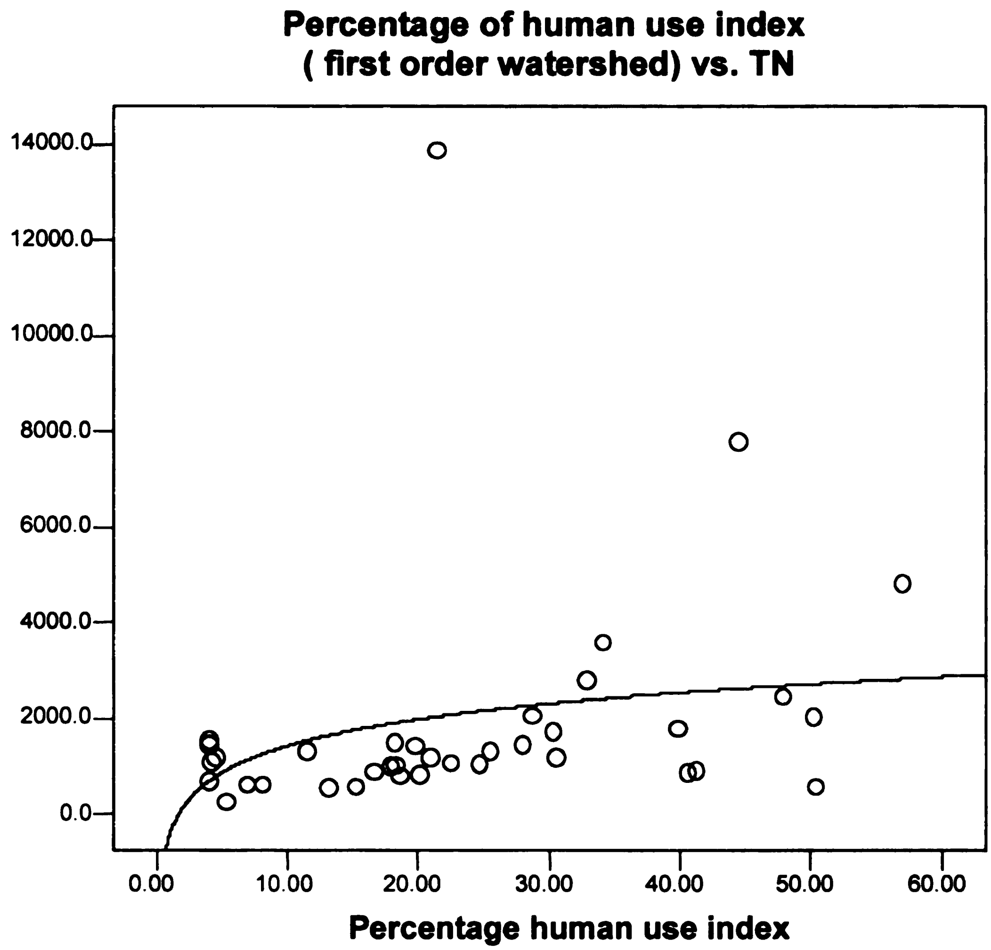


Figure 4-6. Percentage of human use index (urban + agriculture + bare within first order watersheds) vs. TN concentrations

The combined percentages of urban and agricultural land uses as well as bare soil were regressed with TN concentrations within the first order watersheds (Figure 4-6). The logarithmic model explained 7.1 % of the dependent variable, total nitrogen, but was not statistically significant at the 95 % confidence level (Table 4-7).

F critical value = 4.13 at $\alpha = 0.05$ (95 % probability) and $df = 35$

F statistic = 2.666, therefore 2.666 < 4.13 or F statistic < F critical

The TN concentrations could be influenced by land use types such as urban, agricultural and bare soil. However, this is to be expected as the three LULC types are all created and modified by humans. Therefore, we would expect to find an inverse or negative correlation between natural cover such as forests and nitrogen concentrations to prove that nutrient loading in the MRW is a function of land use. A logarithmic model (Figure 4-7) in which the percentage of forest cover was regressed with TN concentrations within the sub-watersheds resulted in $R^2$ = 0.024, explaining 2.4 % of the dependent variable (TN concentration), but was not statistically significant at $\alpha = 0.05$ (Table 4-6).

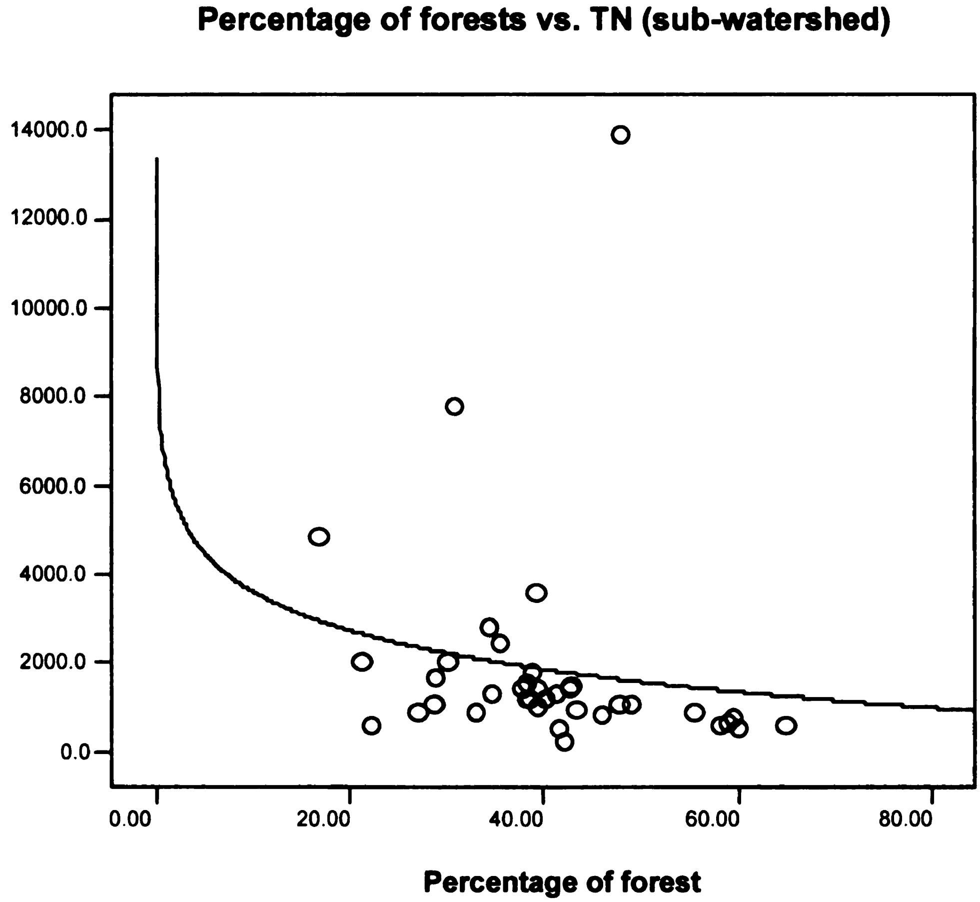


Figure 4-7. Percentage of forests (within sub-watersheds) vs. TN concentrations

F critical value = 4.13 at $\alpha = 0.05$ (95 % probability) and *df* = 35

F statistic = 0.851, therefore, 0.851< 4.13 or F statistic < F critical

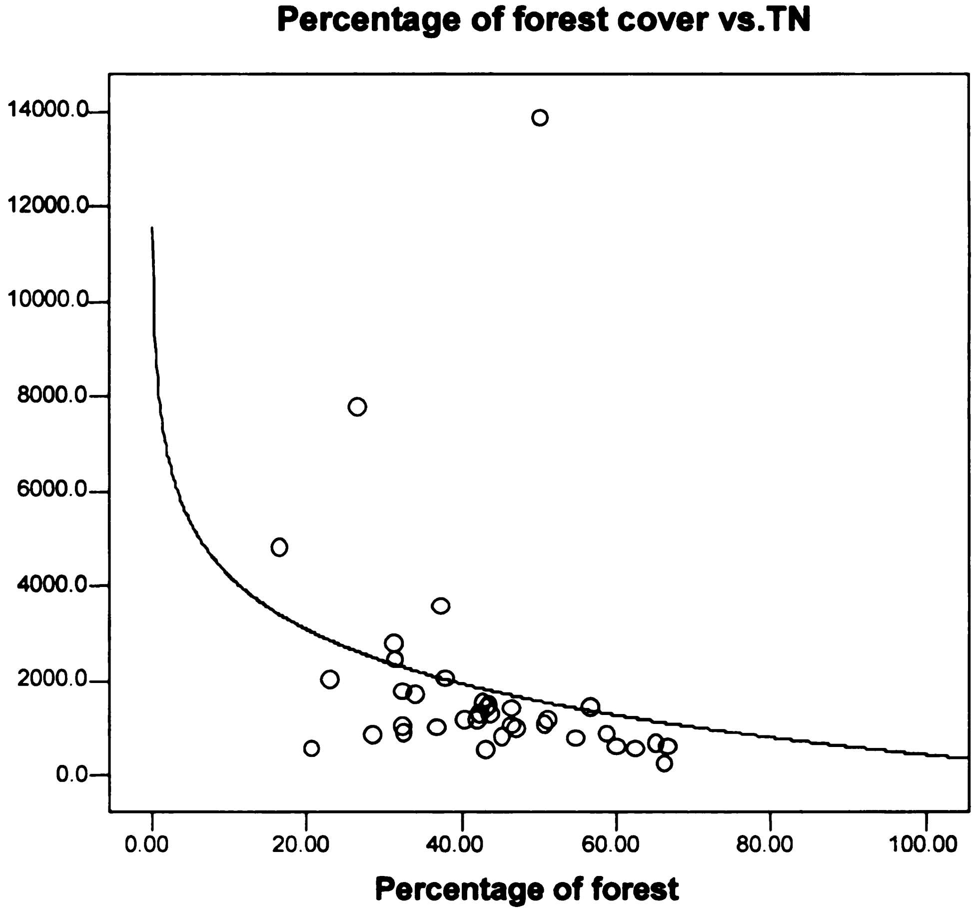


Figure 4-8. Percentage of forests (within first order watersheds) vs. TN concentrations

Another logarithmic model (Figure 4-8) in which the percentage of forest cover was regressed with TN concentrations within the first order watersheds resulted in $R^2 = 0.049$ explaining 4.9 % of the dependent variable, TN concentration, but was not statistically significant at $\alpha = 0.05$ (Table 4-7).

critical value = 4.13 at $\alpha = 0.05$ (95 % probability) and $df = 35$

F statistic = 0.851, therefore, 0.851< 4.13 or F statistic < F critical

The proportion of urban land use within a watershed gives us an indication of urban development. However, a more accurate picture of the effects of anthropogenic modification can be realized through the use of a human use index. The human use index combines the proportion of land use types such as urban, agriculture and bare soil. The index allows us to identify areas that have been converted from their natural state (O'Neil et al., 1988) instead of considering individual LULC categories. The logarithmic regressions between total nitrogen distributions and percentages of LULC indicate an initial increase in TN concentrations with an increase in the percentages of agricultural and urban land use and then remain unchanged regardless of any increase in LULC percentages.

Denitrification, sedimentation and uptake by aquatic biotic communities would account for some of the variability in the regression model (Hill, 1981; Burns 1998). In addition to the above, nutrient transport within a watershed is a function of stream discharge and travel time. Travel time is a measure of stream length divided by stream velocity. Stream velocity itself is directly proportional to stream discharge. As stream discharge drops, velocity also drops resulting to an increase in travel times from the source to the stream system. As a result, a lesser volume of water entering the watershed system would mean a greater loss of nutrients such as nitrogen and phosphorus. Total nitrogen distribution, depicted by graded symbols across the watershed was overlaid on a GIS layer depicting the percentage of agricultural land using ESRI Arc Map. The resultant map shows a strong spatial correlation between increasing total nitrogen concentrations and the percentage of agricultural land use (Figure 4-9). It is interesting to note that the upper Midwest has the highest application rates of fertilizer (greater than 7 tons per square mile) when compared to the rest of the Nation (USGS circular, 1999).

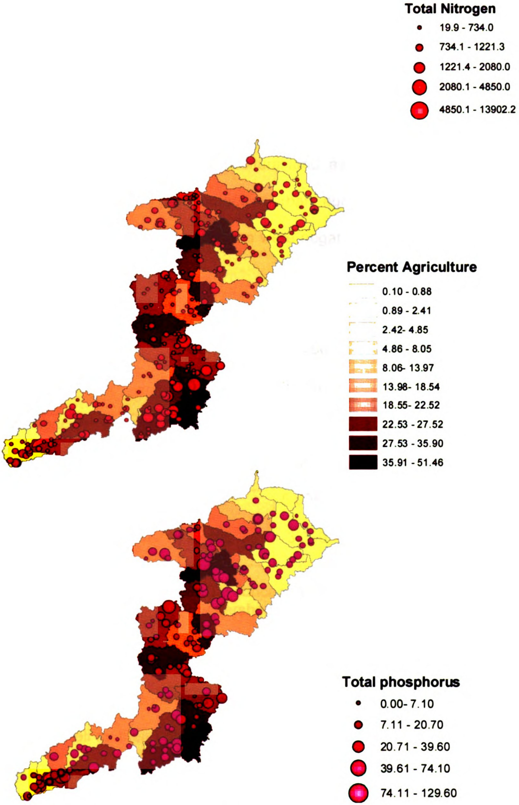


Figure 4-9. TN and TP concentration distribution overlaid on % agricultural land use

### 4.1.3 TP concentration and percentage of LULC

Total phosphorus concentrations (measured in parts per billion) within the sub-watershed were assumed to be positively correlated (using Spearman's rank correlation) with the percentages of urban land use, agricultural land, the combined percentage of urban and agricultural land, as well as the percentage of human use index (combined percentages of urban, agricultural and bare land). It was also assumed that TP concentrations were negatively correlated with forest cover.

| TP vs. LULC | Correlation | *t* value | *t* critical, $\alpha = 0.05$ |
|---|---|---|---|
| TP vs. urban | 0.384 | 2.425 | 1.695 |
| TP vs. agriculture | 0.564 | 3.982 | 1.695 |
| TP vs. urban + agriculture | 0.651 | 5.000 | 1.695 |
| TP vs. urban + ag + bare | 0.658 | 5.095 | 1.695 |
| TP vs. forest | -0.496 | -3.330 | -1.695 |

Table 4-8. Correlation between TP concentrations and LULC within sub-watersheds

In addition, TP concentrations within the first order watersheds were assumed to be positively correlated with percentages of urban and agricultural LULC types as well as their combined percentages. It was also assumed that the TP concentrations were negatively correlated with the percentage of forests in the first order watersheds.

| TP vs. LULC | Correlation | *t* value | *t* critical, α = 0.05 |
|---|---|---|---|
| TP vs. urban | 0.295 | 1.8002 | 1.695 |
| TP vs. ag | 0.568 | 4.0241 | 1.695 |
| TP vs. urban + agriculture | 0.640 | 4.8567 | 1.695 |
| TP vs. urban + ag + bare | 0.629 | 4.7178 | 1.695 |
| TP vs. forest | -0.499 | -3.3575 | -1.695 |

Table 4-9. Correlation between TP concentrations and LULC within first order watersheds

As hypothesis, it was assumed that TP concentrations increase when the percentage of anthropogenically modified land within the sub-watersheds and the first order watersheds increases. The null hypothesis was that there is no correlation between the two variables (t ≤ t-critical α = 0.05). As alternative hypothesis, it was assumed that there is a positive correlation between total invertebrate taxa and percentage of urban land use (t > t-critical at α = 0.05). The t critical value was 1.695 at α = 0.05 (95% probability) and df = 34.

$H_o$ Null hypothesis: there is no correlation between TP concentrations and percentage of land use

$H_1$ Alternative hypothesis: there is a correlation between TP concentrations and percentage of land use

The null hypothesis was rejected as the *t* values for the different cases (Table 4-8) were greater than the t critical value = 1.695.

Logarithmic regression analysis was also undertaken to explain the variance in a regression model between percentage of LULC (dependent variable) and TP concentration in the sub-watershed as well as within the first order watersheds. A non-linear regression was considered, as a linear least square regression did not explain any variance in the independent variable. A logarithmic model (Table 4-10) fitted to TP concentrations and the percentage of urban land use within the sub-watersheds which yielded an $R^2$ = 0.020. The model explained 2.0% of the dependent variable, TP concentrations and was not statistically significant at $\alpha$ = 0.05.

F critical value = 4.12 at $\alpha$ = 0.05 (95 % probability) and *df* = 34

F statistic = 0.701 therefore 0.701 < 4.12 or F statistic < F critical

| Independent | Logarithmic model for total phosphorus (dependent) | | | | | Parameter estimates | |
|---|---|---|---|---|---|---|---|
| % LU-sub | R Square | F | df1 | df2 | Sig. | Constant | b1 |
| urban | .020 | 0.701 | 1 | 34 | .408 | 25.006 | 6.740 |
| agriculture | .187 | 7.836 | 1 | 34 | .000 | 10.884 | 9.163 |
| urb.+ag. | .307 | 15.053 | 1 | 34 | .000 | -20.436 | 18.713 |
| h.u. index | .309 | 15.206 | 1 | 34 | .000 | -21.401 | 18.948 |
| forest | .404 | 23.012 | 1 | 34 | .000 | 238.851 | -56.44 |

Table 4-10. Percentage of LULC (within sub-watersheds) vs. TP concentrations

A logarithmic model (Figure 4-10) fitted to TP concentrations and the percentage of agricultural land use within the sub-watersheds yielded an $R^2$ = 0.187 and significantly explained 18.7 % of the dependent variable, TP concentrations (Table 4-10).

F critical value = 4.12 at $\alpha = 0.05$ (95 % probability) and *df* = 34

F statistic = 7.836

Therefore 7.836 4.12 or F statistic > F critical

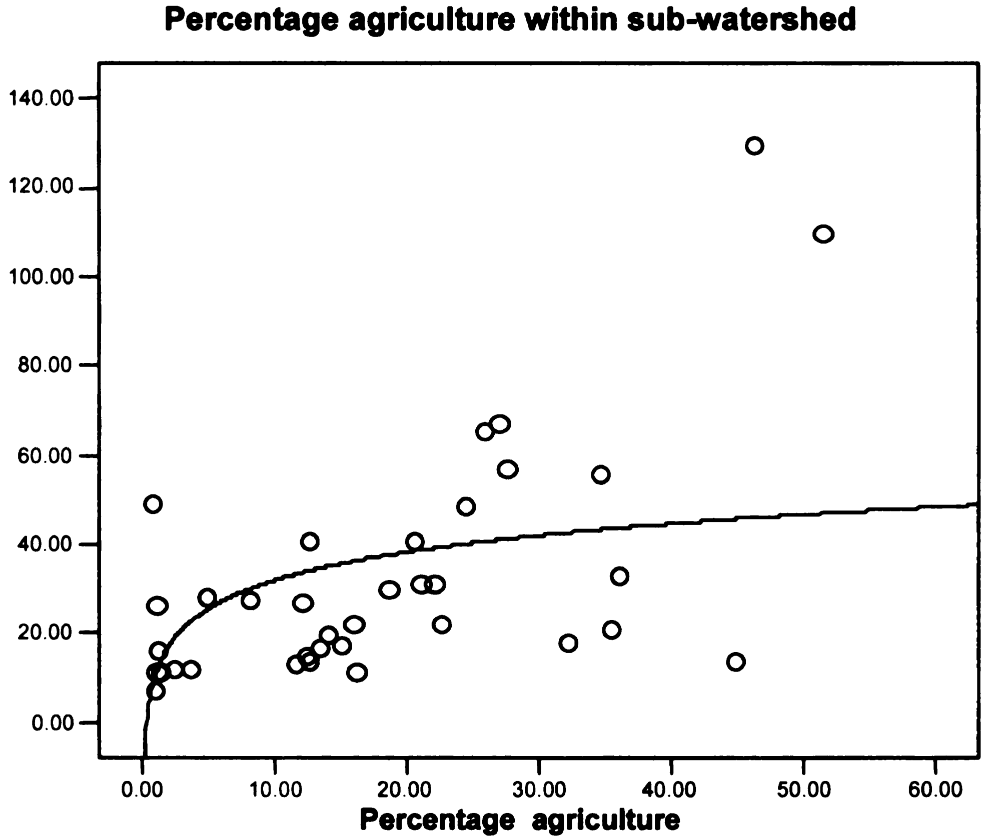


Figure 4-10. Percentage of agricultural land (within sub-watersheds) vs. TP concentrations

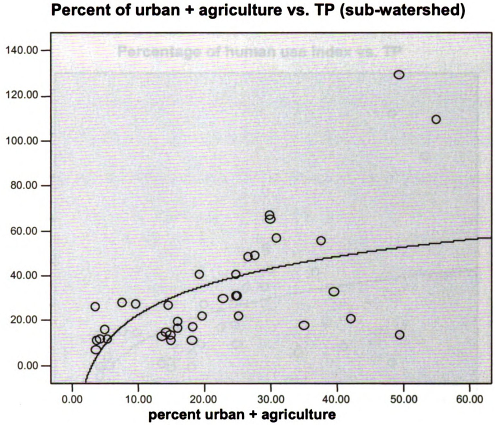


Figure 4-11. Percentage of urban & agricultural land (within sub-watersheds) vs. TP concentrations

A logarithmic model (Figure 4-11) fitted to TP concentrations and the combined percentages of urban and agricultural land use raised the $R^2$ to 0.307, significantly explaining 30.7 % of the dependent variable, TP concentrations (Table 4-10).

F critical value = 4.12 at $\alpha$ = 0.05 (95 % probability) and *df* = 34

F statistic = 15.053 therefore 15.053 > 4.12 or F statistic > F critical

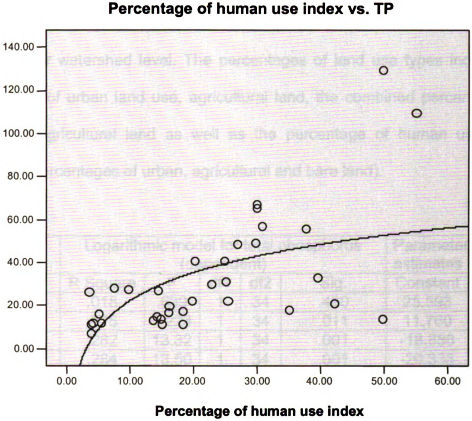


Figure 4-12. Percentage of human use index within sub-watersheds vs. TP concentrations

A logarithmic model (Figure 4-12) in which the combined percentages of urban land use, agricultural land use, and bare soil were regressed with TP concentrations within the sub-watersheds resulted in $R^2$ = 0.309, significantly explaining 30.9 % of the dependent variable, TP concentrations (Table 4-10).

F critical value = 4.12 at $\alpha$ = 0.05 (95 % probability) and $df$ = 34

F statistic = 15.206, therefore 15.206 > 4.12 or F statistic > F critical

Logarithmic regression was also undertaken to explain the relationship and variance in a regression model between the percentages of LULC (dependent variable) and TP concentrations (dependent variable) aggregated to the first order watershed level. The percentages of land use types include the percentages of urban land use, agricultural land, the combined percentages of urban and agricultural land as well as the percentage of human use index (combined percentages of urban, agricultural and bare land).

| Independent | Logarithmic model for total phosphorus (dependent) | | | | | Parameter estimates | |
|---|---|---|---|---|---|---|---|
| % LU-first | R Square | F | df1 | df2 | Sig. | Constant | b1 |
| urban | .018 | .639 | 1 | 34 | .430 | 25.593 | 6.612 |
| agriculture | .175 | 7.223 | 1 | 34 | .011 | 11.760 | 8.444 |
| urban+ag. | .282 | 13.32 | 1 | 34 | .001 | -18.886 | 17.636 |
| human index | .284 | 13.50 | 1 | 34 | .001 | -20.333 | 18.031 |
| forest | .367 | 19.69 | 1 | 34 | .000 | 218.885 | -50.37 |

Table 4-11. Percentage of LULC (within first order watersheds) vs. TP concentrations

A logarithmic model (Table 4-11) fitted to TP concentrations and the percentage of urban land use within the first order -watersheds yielded an $R^2$ = 1.8. The model explained 1.8% of the dependent variable (TP concentration) and was not statistically significant at $\alpha = 0.05$.

F critical value = 4.12 at $\alpha = 0.05$ (95 % probability) and $df = 34$

F statistic = 0.639 therefore 0.639 < 4.12 or F statistic < F critical

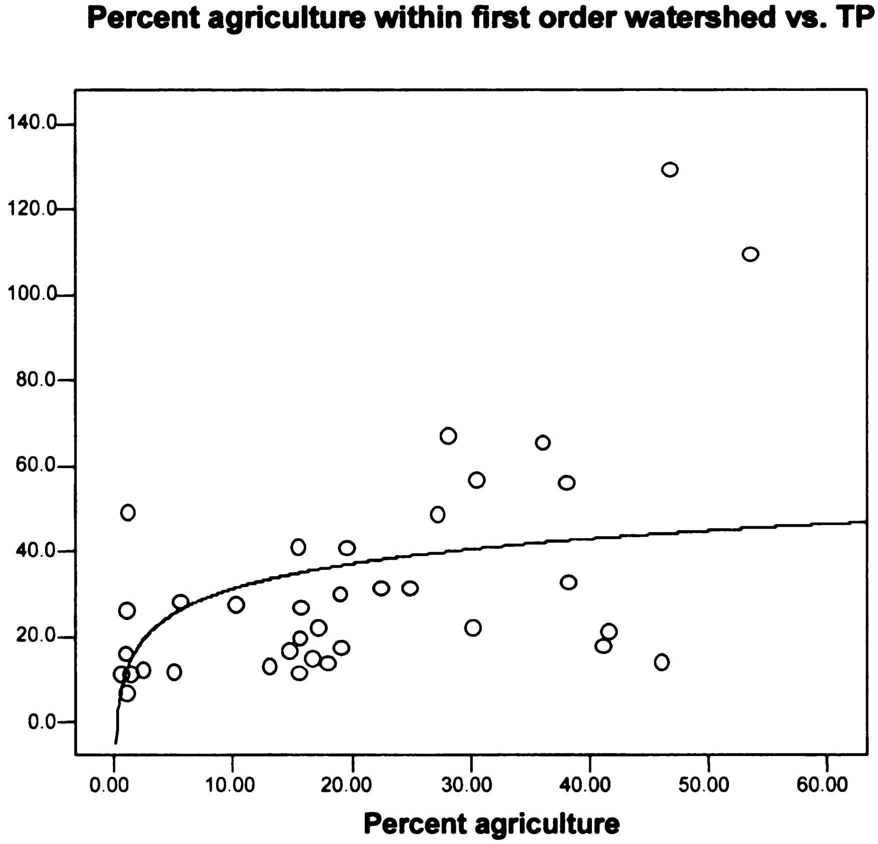


Figure 4-13. Percentage of agriculture (within first order -watersheds) vs. TP concentrations

A logarithmic model (Figure 4-13) fitted to TP concentrations and the percentage of agricultural land use within the first order-watersheds yielded an $R^2$ = 0.175 and significantly explained 17.5 % of the dependent variable, TP concentrations (Table 4-11).

F critical value = 4.12 at $\alpha$ = 0.05 (95 % probability) and *df* = 34

F statistic = 7.223

Therefore 7.223 4.12 or F statistic > F critical

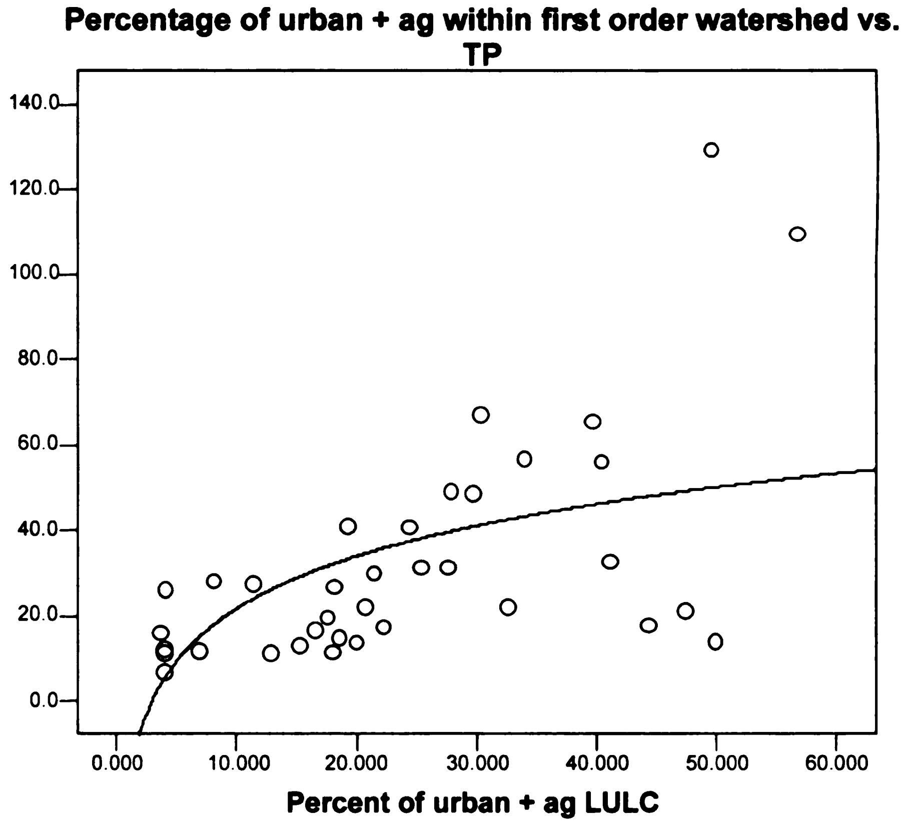


Figure 4-14. Percentage of urban & agriculture (within first order-watersheds) vs. TP concentrations

A logarithmic model (Figure 4-14) fitted to TP concentrations and the combined percentages of urban and agricultural land use resulted in an $R^2$ of 0.282, significantly explaining 28.2% of the dependent variable, TP concentrations (Table 4-11).

F critical value = 4.12 at $\alpha = 0.05$ (95 % probability) and $df = 34$

F statistic = 13.328 therefore 13.328 > 4.12 or F statistic > F critical

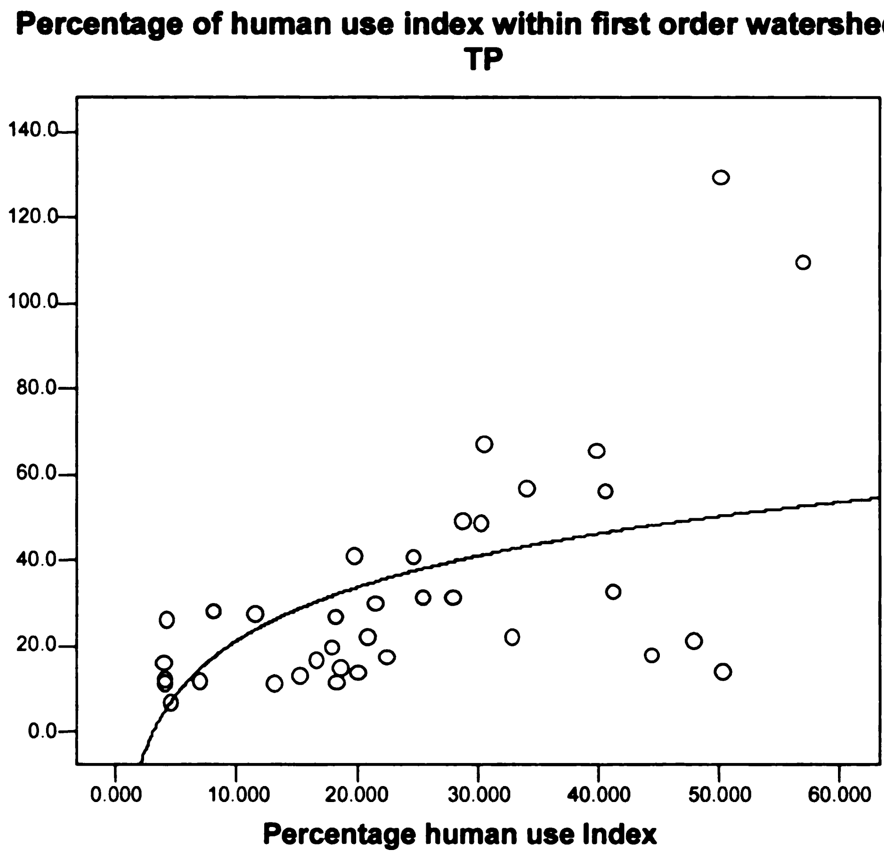


Figure 4-15. Percentage of human use index (urban + agriculture + bare within first order watersheds) vs. TP concentrations

Another logarithmic model (Figure 4-15) in which the combined percentages of urban, agricultural and bare soil were regressed with TP concentrations within the first order watersheds resulted in $R^2$ = 0.284, significantly explaining 28.4 % of the dependent variable, TP concentrations (Table 4-11).

F critical value = 4.12 at $\alpha$ = 0.05 (95 % probability) and *df* = 34

F statistic = 13.500, therefore 13.500> 4.12 or F statistic > F critical

The amount of TP concentrations increased with the percentages of urban & agricultural land use and then remained the same irrespective of any increase when bare soil cover was added. This might be accounted for by mitigation of TP concentrations due to sedimentation, and stream flow. The TP concentrations were positively correlated with the land use types such as urban, agricultural and bare soil. However, this is to be expected as the three LULC types are all created and modified by humans. Therefore, we would expect to find an inverse or negative correlation between natural cover such as forests and phosphorus concentrations to prove that nutrient loading in a watershed is a function of land use.

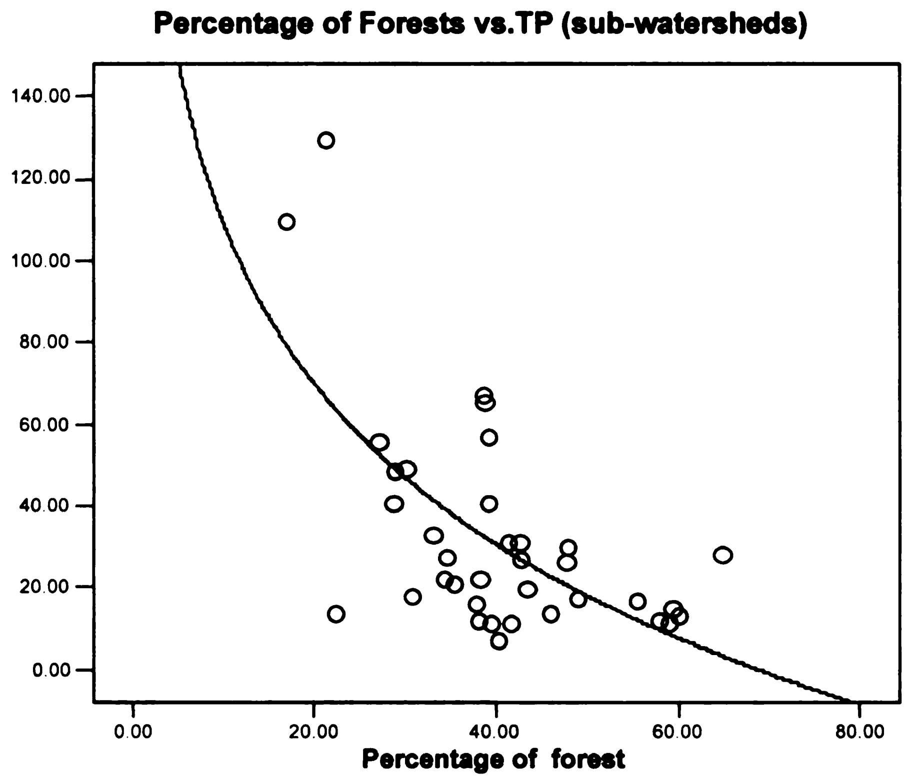


Figure 4-16. Percentage of forests vs. TP concentrations within sub-watersheds

A logarithmic model (Figure 4-16) in which the percentage of forest cover was regressed with TP concentrations within the sub-watersheds resulted in $R^2 = 0.404$, significantly explaining 40.4% of the dependent variable, TP concentrations (Table 4-11).

F critical value = 4.13 at $\alpha = 0.05$ (95 % probability) and $df = 34$

F statistic = 23.012, therefore 23.012 > 4.13 or F statistic > F critical

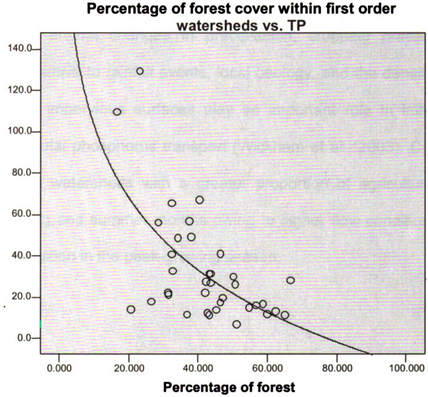


Figure 4-17. Percentage of forests vs. TP concentrations within sub-watersheds

Another logarithmic model (Figure 4-17) in which the percentage of forest cover was regressed with TP concentrations within the first order-watersheds resulted in $R^2 = 0.367$, significantly explaining 36.7% of the dependent variable, TP concentrations (Table 4-11).

The regression between total phosphorus and forest cover within the sub-watershed as well as the first order watersheds confirmed the negative correlation between the variables. This negative relationship confirms the premise that land use patterns influence TP concentrations within the MRW.

The nutrient export within the watershed as downstream of the headwaters is a function of various physical and anthropogenic factors. Factors such as inter-annual changes in precipitation, cropping practices, fertilizer applications relative to rainfall events, local geology, and the density and spatial distribution of impervious surfaces play an important role in influencing total nitrogen and total phosphorus transport (Wickham et al., 2003). Concentrations of nutrients in watersheds with a greater proportion of agricultural areas are higher in spring and summer months owing to higher flow conditions as well as fertilizer application in the peak-growing season.

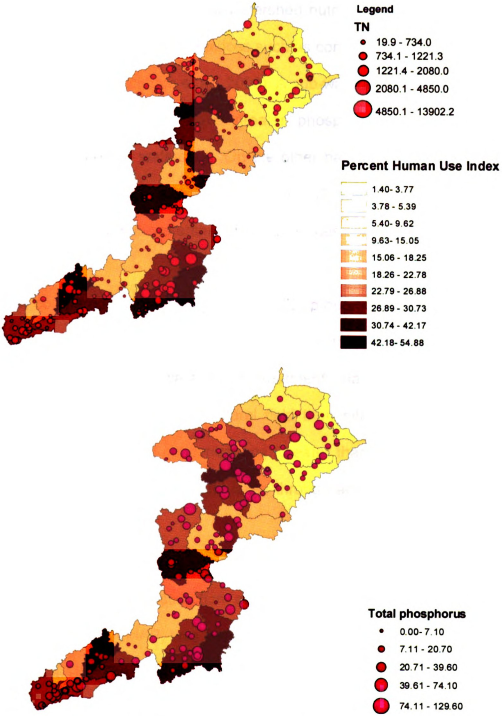


Figure 4-18. TN and TP concentrations overlaid on % human use index

The differences between sub-watershed nutrient concentrations could be explained by different cropping practices such as conservation tillage versus non-conservation tillage (Wickham et al., 2003). A sub-watershed where conservation tillage was practiced would have greater phosphorus concentrations than compared to a forested watershed. On the other hand, a sub-watershed where non-conservation tillage was practiced would have greater phosphorus concentrations than a sub-watershed which practices conservation tillage.

As compared to total nitrogen, a smaller proportion of phosphorus is lost to nutrient decay. Total phosphorus originates from livestock manure and agricultural fertilizer and wastewater treatment plants. Even though less phosphorus is leached from agricultural field than nitrogen, there is a higher probability that it will reach concentrations (greater than 0.1 mg/liter) enough to cause eutrophication in streams and lakes through aquatic plant growth (USGS circular, 1999).

Total phosphorus distribution depicted by graded symbols across the watershed in ESRI ArcMAP, was overlaid on the percentage of agricultural land use (Figure 4-18). The total phosphorus distribution was also mapped on the total proportion of the landscape modified by human activities, ie, the human use index (urban + agriculture + bare). Again, a strong spatial correlation was seen between the two variables. A visual assessment indicates that total phosphorus concentrations that are downstream of sub-watersheds with greater percentage of agricultural land use have higher values (Figure 4-18).

### 4.1.4 Total invertebrate taxa and percentage of urban land use

Total invertebrate taxa within the sub-watershed were negatively correlated with the percentage of urban land use (Spearman's rank correlation: -0.416, $\alpha$ = 0.05). The null hypothesis was that there is no correlation between the two variables (t $\leq$t-critical at $\alpha$ = 0.05). As alternative hypothesis, it was assumed that there is a negative correlation between total invertebrate taxa and the percentage of urban land use (t > t-critical at $\alpha$ = 0.05). The t critical value = -1.695 at $\alpha$ = 0.05 (95% probability) and df = 34.

$H_0$ Null hypothesis: there is no correlation between total invertebrate taxa and the percentage of urban land use

$H_1$ Alternative hypothesis: there is a correlation between total invertebrate taxa and the percentage of urban land use

The null hypothesis was rejected as the t = -2.675 and was greater than –1.695.

| Total invertebrate taxa & LULC | Correlation | *t* value | *t* critical, $\alpha$ = 0.05 |
|---|---|---|---|
| Total invertebrate taxa & % urban | -0.416 | -2.675 | -1.695 |

Table 4-12. Percentage of urban land use vs. total invertebrate population within sub-watersheds

Linear regression of the total invertebrate taxa and the percentage of urban land use failed to explain the variance in the data and so a log-transform was used on the percentage of urban land use to linearize the model.

$$y'i = \log\left(N * y + \frac{0.5}{N}(1-y) + \frac{1}{2}\right) \tag{4-1}$$

where y = % urban land use

The log-transformed model (Bailey and Gatrell, 1995) significantly described 22% of the variance ($p < 0.003$) between total invertebrate taxa and the percentage of urban land use. The linear regression performed after the log transform showed a negative trend between the diversity of invertebrate taxa and the percentage of urban land use (constant = 5.5661 and slope = -0.022).

### 4.1.5 Total EPT taxa and percentage of urban land use

Total EPT taxa within the sub-watershed were negatively correlated with the percentage of urban land use (Spearman's rank correlation: -0.381, $\alpha = 0.05$). The null hypothesis was that there is no correlation between the two variables ($t \leq$ t-critical, $\alpha = 0.05$). As alternative hypothesis, it was assumed that there is a negative correlation between total EPT and the percentage of urban land use ($t >$ t-critical at $\alpha = 0.05$). The t critical value = -1.695 at $\alpha = 0.05$ (95% probability) and df = 34.

$H_0$ Null hypothesis: there is no correlation between total EPT taxa and the percentage of urban land use

$H_1$ Alternative hypothesis: there is a correlation between total EPT taxa and the percentage of urban land use

The null hypothesis was rejected as the t = -2.393 and was greater than t-critical.

| Sensitive insect taxa & LULC | Correlation | *t* value | *t* critical, $\alpha = 0.05$ |
|---|---|---|---|
| EPT taxa & % urban | -0.381 | -2.393 | -1.695 |

Table 4-13. Percentage of urban land use correlated with sensitive insect population within sub-watersheds

EPT taxa within the sub-watershed were regressed with the percentage of urban use. As in the case of total invertebrate taxa, a linear regression failed to explain the variance in the data and so the percentage of LULC was log-transformed in an attempt to linearize the model. A linear regression of the log-transformed urban land use yielded an $R^2 = 0.15$ ($p < 0.011$) and shows that there is a slight decrease in EPT taxa with increase in percentage of urban land use (constant = 5.264 and slope = -0.036).

This negative correlation can be explained by the fact that the mayflies, caddisflies and stoneflies that make up the EPT taxa evolved in oxygen rich, cool waters. Depleted oxygen levels as well as increasing water temperatures due to increased runoff and short travel times over increasing impervious surfaces might explain the reason in EPT population decline as the percentage of urban land use increases.

### 4.1.6 Proximity to urban areas

A correlation test between total nitrogen concentrations and distance to urban areas found the two variables to be negative correlated (Table 4-14). Total nitrogen concentrations dropped with increasing distance from urban areas.

| TN concentration & distance to LULC | Correlation, $\alpha = 0.05$ |
|---|---|
| TN concentration & Urban | -0.137 |

Table 4-14. Proximity to urban areas & TN concentrations within MRW

The null hypothesis was that there is no correlation between the two variables (t $\leq$t-critical at $\alpha = 0.05$). As alternative hypothesis, it was assumed that there is a negative correlation between proximity to urban areas and TN concentrations within the watershed (t > t-critical at $\alpha = 0.05$). The t critical value = -1.645 at $\alpha = 0.05$ (95% probability) and df = 217.

$H_o$ Null hypothesis: there is no correlation between proximity to urban areas and TN concentrations

$H_1$ Alternative hypothesis: there is a correlation between proximity to urban areas and TN concentrations

The null hypothesis was rejected as the t = -2.0375 and was greater than t-critical. A correlation test was also carried out between total phosphorus concentrations and distance to urban areas found the two variables to be positively correlated but with a very low correlation co-efficient, which was not statistically significant (at $\alpha$ = 0.05).

| TP concentration & distance to LULC | Correlation, $\alpha$ = 0.05 |
|---|---|
| TP concentration & urban | 0.057 |

Table 4-15. Proximity to urban areas & TP concentrations within MRW

The null hypothesis was that there is no correlation between the two variables (t $\leq$t-critical at $\alpha$ = 0.05). As alternative hypothesis, it was assumed that there is a negative correlation between proximity to urban areas and total phosphorus concentrations within the watershed (t > t-critical at $\alpha$ = 0.05). The t critical value = 1.645 at $\alpha$ = 0.05 (95% probability) and df = 127.

$H_o$ Null hypothesis: there is no correlation between proximity to urban areas and TP concentrations

$H_1$ Alternative hypothesis: there is a correlation between proximity to urban areas and TP concentrations

The alternative hypothesis was rejected as the t = 0.6434 and was less than t-critical. Regression analysis characterized a positive linear association between the dependent variable (total nitrogen concentration) and the independent variable, distance to urban areas resulted in a negative relationship (Table 4-16). However, it was not statistically significant at $\alpha$ = 0.05.

| Equation | Model summary | | | | | Parameter estimates | |
|---|---|---|---|---|---|---|---|
| | R Square | F | df1 | df2 | Sig. | Constant | b1 |
| Linear | .008 | 1.735 | 1 | 217 | .189 | 1043.300 | -.397 |

Table 4-16. Proximity to urban areas vs. TN concentrations within MRW

### 4.1.7 Proximity to agricultural areas

A correlation test carried out between total nitrogen concentrations and distance to agricultural areas found the two variables to be negatively correlated, but with a very low correlation co-efficient, which was not statistically significant at $\alpha$ = 0.05 (Table 4-17).

| TN concentration & distance to LULC | Correlation |
|---|---|
| TN concentration & agriculture | -0.0124 |

Table 4-17. Proximity to agricultural areas & TN concentrations within MRW

A correlation test between TP concentrations and distance to agricultural areas found the two variables to be negative correlated (Table 4-18). Total phosphorus concentrations dropped with increasing distance from agricultural areas.

| TP concentration & distance to LULC | Correlation, $\alpha = 0.05$ |
| --- | --- |
| TP concentration & agriculture | -0.324 |

Table 4-18. Proximity to agricultural areas & TP concentrations within MRW

The null hypothesis was that there is no correlation between the two variables ($t \leq$t-critical at $\alpha = 0.05$). As alternative hypothesis, it was assumed that there is a negative correlation between proximity to agricultural areas and TP concentrations within the watershed ($t >$ t-critical at $\alpha = 0.05$). The t critical value = 1.645 at $\alpha = 0.05$ (95% probability) and df = 127.

$H_0$ Null hypothesis: there is no correlation between proximity to agricultural areas and TP concentrations

$H_1$ Alternative hypothesis: there is a correlation between proximity to agricultural areas and TP concentrations

The null hypothesis was rejected as t = -3. 8594 and was greater than t-critical.

# CHAPTER 5 CONCLUSIONS

The objective of this research was to obtain an accurate inventory of LULC within the 40 sub-watersheds of the Muskegon River Watershed and correlate the LULC types with water quality indices such as total nitrogen, total phosphorus, specific conductivity, populations of sensitive insect species (EPT taxa) and total invertebrate taxa. The specific hypothesis was that LULC affects water quality within the MRW.

The results demonstrate that there is a positive correlation between types of land use (urban, agriculture and bare soil) and water quality indicators. This positive correlation was determined by the spatial scale of study. The two spatial scales considered were the sub-watershed level study and the first order watershed level. In addition, negative correlations were found between forest cover and water quality indicators at the sub-watershed level. This confirms the fact that it is land use within the watershed that affects the water quality and that forest cover within catchments is beneficial.

The sub-hypothesis conclusion is that the LULC within the MRW was accurately mapped through an unsupervised classification of ETM+ imagery (Landsat-7) and with an overall classification accuracy of 83%. In addition, water quality indicators measured at 256 sampling locations were aggregated to the sub-watershed level and correlated with LULC types within the 40 sub-watersheds.

Statistical relationships were established between water quality measures and percentage of land use / land cover within the watershed. Among the water quality indicators, total phosphorus and total nitrogen distribution as well as biological indicators such as EPT taxa and total invertebrate taxa, were influenced by the percentages of agricultural and urban land uses within the sub-watershed.

Significant positive correlations were found between total nitrogen concentrations and percentage of urban and agricultural land uses (0.18 and 0.4 respectively). The correlations increased slightly when the combined percentages of urban, agriculture and bare soil were considered. Positive correlations were also found between total phosphorus concentrations and percentage of urban and agricultural land uses (0.38 and 0.65 respectively). These correlations are important as the phosphorus concentrations cause eutrophication in lakes and steams within the watershed. The sources of the nutrients might be from fertilizer application on farms but might also be from confined animal feeding operations (CAFO's) or wastewater treatment plants. An inverse or negative correlation was found between natural cover such as forests and nitrogen/phosphorus concentrations (0.40 and 0.49 respectively) to prove that nutrient loading in a watershed is a function of land use.

The variance in total phosphorus concentrations could be explained more by the percentage of land use within the sub-watersheds as compared to the variance in total nitrogen concentrations. Though the statistical relationships between the dependent variables (water quality indices) and the independent variables (LULC types) were significant at the 95% confidence level, the predictive power of the logarithmic models was very low with $R^2$ ranging from 0.018 to 0.40.

The low predictive power of the logarithmic regression models for nitrogen concentrations can be attributed to various physical and anthropogenic factors. For instance, the variability in total nitrogen concentrations could be attributed to denitrification, sedimentation and uptake by aquatic biotic communities. The transport of nutrients in a watershed is also influenced by stream velocity and travel time. The variability of the nutrient concentrations within the sub-watersheds could also be governed by factors such as changes in precipitation, cropping practices, fertilizer applications relative to rainfall events, local geology, the density and the spatial distribution of impervious surfaces. Total nitrogen concentrations were influenced more by distance to urban areas as opposed to total phosphorus concentrations, which were influenced by distance to agricultural areas.

Specific conductivity was positively correlated with individual land use types such as urban and agricultural land use or the combined percentages of human modified LULC types with correlations ranging from 0.60 and 0.72. This could be attributed to a greater proportion of urban land use in some of the sub–watersheds. Also, the percentage of forest cover was negatively correlated with specific conductivity. This negative correlation makes a strong case for the influence of urban and agricultural land use on water quality. In addition, sensitive insect populations

(EPT taxa) as well as the total invertebrate taxa were negatively correlated with urban land use (0.38 and 0.41 respectively). These correlations were significant at the 95 % confidence level.

My research studies the land-water interactions at two spatial scales, i.e., the sub-watershed level and the first order watershed level. It is based in part on Omernik's study (1981) that demonstrated that land use in the distant uplands had as much effect on water quality as land use in the vicinity. In addition, my research considers the proportions of land use within hydrologically distinct units such as sub-watersheds as opposed to the immediate vicinity, which is the case in most ecological studies. These studies suggest that water quality is influenced by the land use within a buffer zone that is approximately 30-100m from the water's edge.

The beautiful natural surrounding and easy access to recreation have resulted in an increase in urban development around the lakes in Michigan. In recent times, land value has gravitated towards residential development rather than agricultural land use. Also, it is important to note that the upper Midwest has the highest application rates of fertilizers on agricultural land when compared to the rest of the nation. Agricultural land use still plays a major role in affecting water quality. This research is important as it documents the correlation between urban and agricultural land use and water quality of Michigan's second largest watershed. The statistical analysis provides proof that sub-watersheds with a greater proportion of natural cover such as forest cover are beneficial and negatively correlated to nutrient concentrations as compared to sub-catchments with a greater proportion of urban / agricultural land use. Local decision makers could be empowered to act in their own interests and conserve the natural surrounding which in turn would ensure qood water quality. Sub-watersheds that are predominantly urban or agricultural in nature could mitigate their impact on the water quality through the maintenance of riparian buffer strips. In addition to acting as filters and reducing nutrient concentration within lakes and streams, riparian buffers prevent stream bank erosion. Ecologists who believe that landscape patterns influence water quality often prescibe riparian buffers as a conservation measure. Future research that concerns land-water interactions in the MRW might consider correlating landscape metrics (e.g., contagion, dominance) that describe the spatial configuration of the landscape with water quality indicators.

# APPENDICES

## APPENDIX-A: Figures used in the study

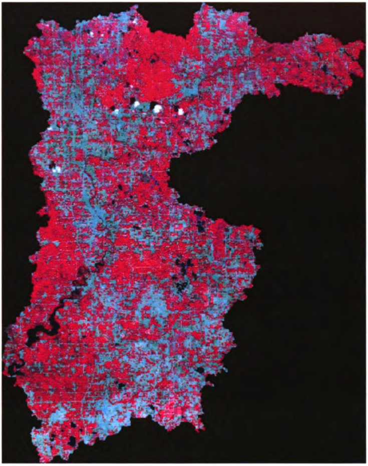

Figure 5-1. Imagery from WRS Path 21 Row 30 Landsat 7 (ETM+)

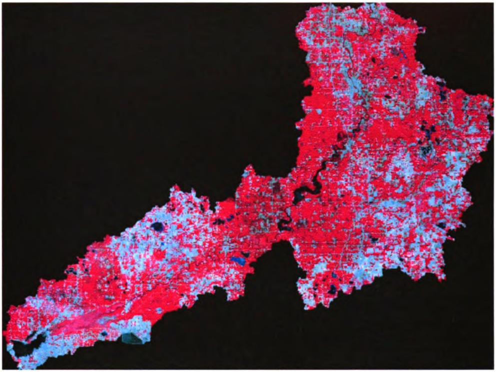

Figure 5-2. Imagery from WRS Path 22 Row 30 Landsat 7 (ETM+)

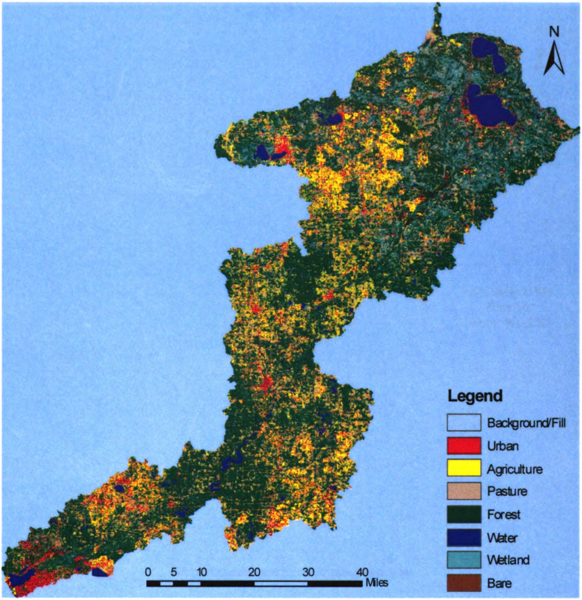


Figure 5-3. MRW–LULC map derived through unsupervised classification

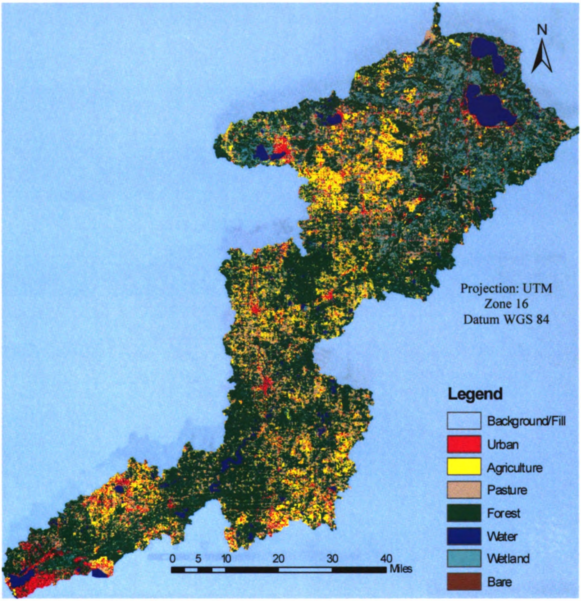


Figure 5-4. MRW–LULC map derived through unsupervised classification and smoothed with 3x3 majority filter

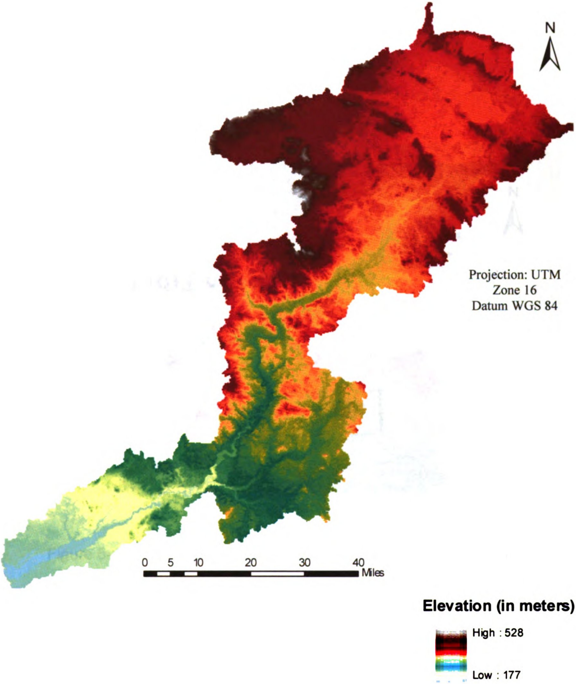


Figure 5-5. SRTM derived Digital Elevation Model (DEM)

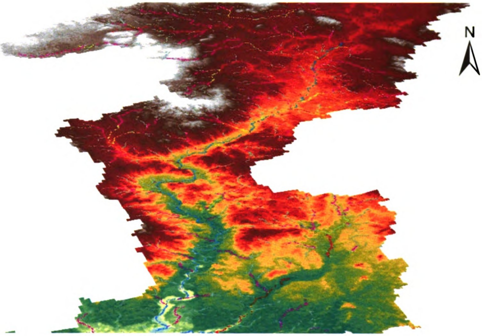


Figure 5-6. SRTM derived DEM overlaid with Strahler stream order

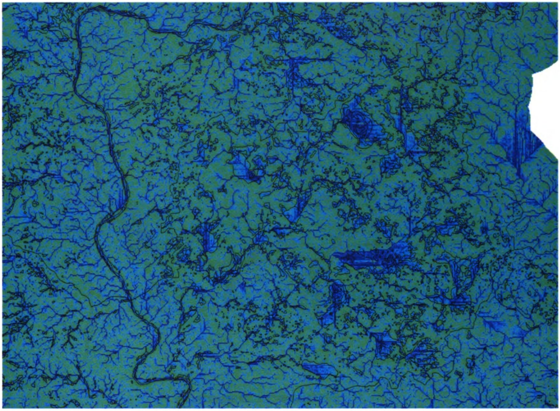

Figure 5-7. Wetness index derived from SRTM DEM overlaid with MRW wetlands coverage

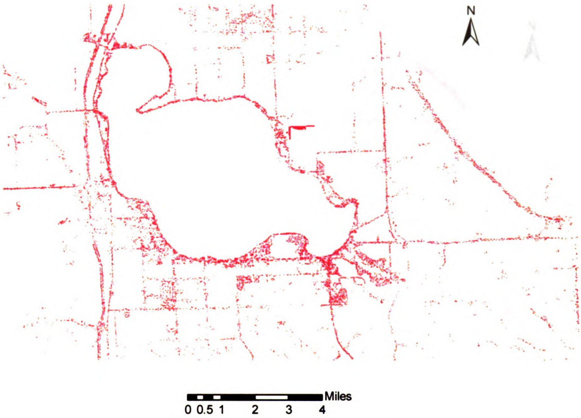


Figure 5-8. Urban development around Lake Houghton (2001)

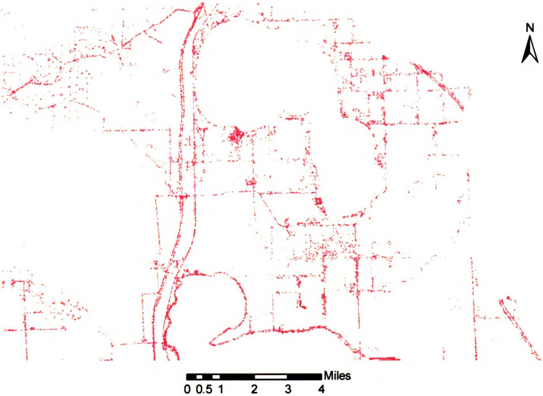


Figure 5-9. Urban development around Lake Higgins (2001)

## APPENDIX B: Data used in statistical analysis

| Sub-watersheds | TN | TP |
|---|---|---|
| 1 | 270.80 | no data |
| 2 | 1204.20 | 7.10 |
| 3 | 1460.00 | 16.17 |
| 4 | 1549.70 | 12.40 |
| 5 | 1320.00 | 27.83 |
| 6 | 1006.20 | 19.80 |
| 7 | no data | no data |
| 8 | 1721.40 | 48.70 |
| 9 | 1118.78 | 26.38 |
| 10 | 695.92 | 11.50 |
| 11 | 637.04 | 11.91 |
| 12 | 1447.30 | 41.10 |
| 13 | 1478.90 | 31.50 |
| 14 | 1200.00 | 22.30 |
| 15 | no data | no data |
| 16 | 2060.10 | 129.60 |
| 17 | 1046.20 | 11.70 |
| 18 | 848.60 | 14.00 |
| 19 | 1193.00 | 67.20 |
| 20 | 1503.10 | 27.10 |
| 21 | 890.70 | 56.20 |
| 22 | 906.60 | 16.91 |
| 23 | 1325.40 | 31.47 |
| 24 | 937.90 | 32.90 |
| 25 | 13902.20 | 30.13 |
| 26 | 586.40 | 13.30 |
| 27 | 2829.40 | 22.23 |
| 28 | 7811.00 | 18.00 |
| 29 | 3595.40 | 57.00 |
| 30 | 4850.00 | 109.80 |
| 31 | 2483.40 | 21.30 |
| 32 | no data | no data |
| 33 | 641.30 | 28.30 |
| 34 | 809.30 | 15.00 |
| 35 | 1808.70 | 65.70 |
| 36 | 593.00 | 14.20 |
| 37 | 1084.40 | 17.60 |
| 38 | 1070.00 | 40.95 |
| 39 | 555.60 | 11.60 |
| 40 | 2080.00 | 49.30 |

Table 5-1. Water quality indicators within MRW sub-watersheds

TN- total nitrogen, TP- total phosphorus (measured in parts per billion)

| Sub-watersheds | Conductivity | EPT taxa | Tot invert. taxa |
|---|---|---|---|
| 1 | no data | no data | no data |
| 2 | 236.0 | 16.0 | 43.0 |
| 3 | 268.5 | 10.0 | 34.0 |
| 4 | 224.0 | 9.0 | 42.0 |
| 5 | 294.0 | 23.0 | 66.0 |
| 6 | 268.8 | 12.0 | 36.0 |
| 7 | no data | no data | no data |
| 8 | 292.5 | 4.0 | 22.0 |
| 9 | 221.0 | 11.0 | 42.0 |
| 10 | 181.0 | 20.0 | 50.0 |
| 11 | 324.0 | 6.0 | 26.0 |
| 12 | no data | no data | no data |
| 13 | 338.8 | 13.0 | 43.0 |
| 14 | 355.9 | 15.0 | 39.0 |
| 15 | no data | no data | no data |
| 16 | 398.1 | 15.0 | 41.0 |
| 17 | 349.3 | 17.0 | 47.0 |
| 18 | 290.8 | 25.0 | 58.0 |
| 19 | 359.3 | 11.0 | 41.0 |
| 20 | 344.4 | 15.0 | 49.0 |
| 21 | 371.6 | 14.0 | 42.0 |
| 22 | 521.0 | 22.0 | 48.0 |
| 23 | 414.3 | 13.0 | 32.0 |
| 24 | 415.7 | 11.0 | 37.0 |
| 25 | 455.0 | 8.0 | 24.0 |
| 26 | 355.4 | 17.0 | 38.0 |
| 27 | 330.0 | 7.0 | 33.0 |
| 28 | 519.3 | 16.0 | 41.0 |
| 29 | 472.2 | 14.0 | 43.0 |
| 30 | 514.0 | 1.0 | 9.0 |
| 31 | 525.3 | 5.0 | 30.0 |
| 32 | 496.5 | 6.0 | 23.0 |
| 33 | 325.0 | 2.0 | 26.0 |
| 34 | 287.8 | 18.0 | 42.0 |
| 35 | 521.3 | 27.0 | 57.0 |
| 36 | 496.5 | 6.0 | 23.0 |
| 37 | 273.5 | 9.0 | 34.0 |
| 38 | 444.9 | 12.0 | 49.0 |
| 39 | 286.5 | 6.0 | 28.0 |
| 40 | 631.0 | 2.0 | 16.0 |

**Table 5-2. Water quality indicators within MRW sub-watersheds**

**Conductivity- specific conductivity, EPT taxa- ephemeroptera, plecoptera & trichoptera, tot invert.taxa – total invertebrate taxa**

| sub-watersheds | urban | agriculture | urban+ agric. | bare | ag.+urban+bare | urban (log transformed) |
|---|---|---|---|---|---|---|
| 1 | 4.33% | 0.11% | 4.44% | 0.12% | 4.56% | 0.808 |
| 2 | 2.62% | 0.88% | 3.50% | 0.28% | 3.78% | 0.443 |
| 3 | 3.60% | 1.18% | 4.78% | 0.19% | 4.97% | 0.667 |
| 4 | 1.64% | 2.42% | 4.06% | 0.05% | 4.11% | 0.156 |
| 5 | 1.49% | 8.05% | 9.54% | 0.09% | 9.63% | 0.101 |
| 6 | 1.87% | 13.98% | 15.85% | 0.31% | 16.15% | 0.231 |
| 7 | 0.80% | 0.60% | 1.40% | 0.00% | 1.40% | -0.183 |
| 8 | 2.12% | 24.32% | 26.44% | 0.45% | 26.89% | 0.306 |
| 9 | 2.30% | 1.04% | 3.34% | 0.11% | 3.45% | 0.359 |
| 10 | 2.34% | 1.28% | 3.62% | 0.08% | 3.71% | 0.370 |
| 11 | 1.71% | 3.56% | 5.27% | 0.10% | 5.37% | 0.180 |
| 12 | 6.56% | 12.55% | 19.11% | 1.05% | 20.16% | 1.142 |
| 13 | 2.65% | 21.99% | 24.64% | 0.29% | 24.93% | 0.452 |
| 14 | 3.67% | 15.75% | 19.43% | 0.23% | 19.66% | 0.683 |
| 15 | 3.35% | 42.68% | 46.03% | 0.45% | 46.47% | 0.616 |
| 16 | 2.95% | 46.24% | 49.18% | 0.53% | 49.71% | 0.525 |
| 17 | 1.85% | 16.13% | 17.97% | 0.29% | 18.26% | 0.223 |
| 18 | 1.00% | 2.00% | 3.00% | 0.10% | 3.10% | -0.091 |
| 19 | 2.69% | 26.93% | 29.62% | 0.23% | 29.84% | 0.461 |
| 20 | 2.40% | 11.95% | 14.35% | 0.06% | 14.41% | 0.387 |
| 21 | 2.86% | 34.61% | 37.46% | 0.13% | 37.59% | 0.503 |
| 22 | 2.56% | 13.32% | 15.88% | 0.13% | 16.01% | 0.429 |
| 23 | 3.83% | 21.00% | 24.83% | 0.15% | 24.99% | 0.715 |
| 24 | 3.45% | 35.90% | 39.35% | 0.15% | 39.49% | 0.637 |
| 25 | 4.07% | 18.55% | 22.62% | 0.16% | 22.78% | 0.761 |
| 26 | 2.07% | 11.43% | 13.50% | 0.03% | 13.52% | 0.291 |
| 27 | 2.50% | 22.53% | 25.02% | 0.27% | 25.30% | 0.412 |
| 28 | 2.84% | 32.07% | 34.91% | 0.14% | 35.05% | 0.500 |
| 29 | 3.15% | 27.53% | 30.68% | 0.05% | 30.73% | 0.572 |
| 30 | 3.25% | 51.47% | 54.72% | 0.17% | 54.89% | 0.594 |
| 31 | 6.50% | 35.37% | 41.87% | 0.31% | 42.17% | 1.135 |
| 32 | 5.22% | 0.17% | 5.39% | 0.00% | 5.39% | 0.955 |
| 33 | 2.57% | 4.85% | 7.42% | 0.01% | 7.44% | 0.432 |
| 34 | 1.89% | 12.18% | 14.07% | 0.07% | 14.13% | 0.236 |
| 35 | 3.98% | 25.82% | 29.80% | 0.13% | 29.94% | 0.744 |
| 36 | 1.00% | 2.00% | 3.00% | 0.43% | 3.43% | -0.091 |
| 37 | 3.10% | 14.90% | 18.00% | 0.20% | 18.21% | 0.561 |
| 38 | 3.98% | 20.62% | 24.60% | 0.25% | 24.84% | 0.743 |
| 39 | 13.75% | 1.02% | 14.77% | 0.28% | 15.06% | 1.793 |
| 40 | 26.74% | 0.69% | 27.42% | 2.25% | 29.67% | 2.416 |

**Table 5-3. Percentages of LULC types within sub-watersheds**

## Bibliography


Allan, J. D., Erickson, D. L., and Fay, J. (1997). The influence of catchment land use on stream integrity across multiple scales. *Freshwater Biol.* 37:149–161.

Alexander, G. R., Fenske, J. L., and Smith, D. W. (1995). A fisheries management guide to stream protection and restoration. Michigan Department of Natural Resources, Fisheries Division. Special Report Number 15, Ann Arbor, Michigan.

Anderson, J. R., Hardy, E. E., Roach, J. T., and Witmer, W. E. (1976). A land use and land-cover classification system for use with remote sensor data. USGS Professional Paper 964. Reston, VA: U.S. Geological Survey.

Bailey, T.C., and Gatrell, A.C. (1995) *Interactive Spatial Data Analysis*. Longman.

Bannerman, R., Owens, D., Dodds, R., and Hornewer, N. (1993). Sources of Pollutants in Wisconsin Stormwater. *Water Science and Technology*. 28(3-5): 241-259.

Beach, D. (2002). Coastal Sprawl: The effects of urban design on aquatic ecosystems of the United States, Pew Oceans Commission, Arlington, Va.

Booth, D. B. (1991). Urbanization and the natural drainage system-impacts, solutions and prognoses. *The Northwest Environmental Journal* 7:93-118.

Bruneau, P., Gascuel-oduux, C., Robin, P., Merot, P., Beven, K. (1995). Sensitivity to Space and Time resolution of a hydrological model using Digital Elevation Data. *Hydrological Processes*, 9, 69-81.

Burns, D. A., (1998). Retention of NO−3 in an upland stream environment: a mass balance approach. *Biogeochemistry* 40: 73–96.

Burt, T. P., and Butcher, D. P. (1985). Topographic controls of soil moisture distributions, *J of Soil sci.,* 36, 469-486.

Carpenter, S. R., Caraco, N. F., Corell, D. L., Howarth, R. W., Sharpley, A. N., Smith, V. H. (1998). Nonpoint pollution of suface waters with phosphorus and nitrogen. *Ecological Applications* 8:559-568.

Carpenter, S., Frost, T., Persson, L., Power, M., and Soto, D. (1995). Freshwater ecosystems: linkages of complexity and processes. In H.A. mooney, ed. *Functional Roles of Bio-Diversity: A Global Perspective*, Chapter 12. John Wiley & Sons, New York, New York, USA.

Changnon, S. A. (1992). Inadvertent weather modification in urban areas: Lessons for global climate change. *Bull.Am.Meteorol.Soc.,* 73, 619-627.

Christopherson, R. W. (1995). Elemental Geosystems: A Foundation in Physical Geography, Prentice Hall .

Cirmo, C. P., and Mc Donell, J. J. (1997). Linking the hydrologic and biogeochemical controls of nitrogen transport in near-stream zones of temperate forested catchments. A review. *Journal of Hydrology* 199:88-120.

Congalton, R. G. (1988). Using spatial autocorrelation analysis to explore the errors in maps generated from remotely sensed data. *Photogrammetric Engineering and Remote Sensing*, 54, 587.

Congalton, R. G. (1991). A review of assessing the accuracy of classifications of remotely sensed data. *Remote Sensing of the Environment*, Vol. 37, pp. 35-46.

Cowardin, L. M., Carter, V., Golet, F. C., LaRoe, E. T. (1979). Classification of Wetlands and Deepwater Habitat of the United States, Fish and Wildlife Service, U.S. Department of the Interior, Washington, D.C.

Detenbeck, N., Johnston, C. A., and Niemi, G. (1993). Wetland effects on lake water quality in the Minneapolis/St Paul metropolitan area. *Landscape Ecology* 8:39-61.

Engman, E. T. (1997). Soil moisture, the hydrologic interface between surface and ground waters. Remote Sensing and Geographic Information Systems for Design and Operation of Water Resources Systems (Proceedings of Rabat Symposium S3), 242, 129– 140.

ERDAS Field Guide (2004), Leica Geosystems GIS & Mapping.

ESRI (1994), Cell Based Modeling with GRID.

Fairfield, J., and Leymarie, P. (1991). Drainage networks from grid digital elevation models. *Water Resources,* 27(5), 109– 717.

Fitzgerald, R. W., and Lees, B. G. (1994). Assessing the classification accuracy of multisource remote sensing data. *Remote Sensing of the Environment*, Vol. 47, pp. 362-368.

Gaufin, A. R., and Tarzwell, C. M. (1952). Aquatic invertebrates as indicators of stream pollution. Public Health Reports 67: 57-64.

Gomi, T., Sidle, R, C., Richardson, J. S. (2002). Understanding Processes and downstream Linkages of Headwater Systems. *Bioscience*, 52, (10), 905-916.

Goudie, A. (1990). The human impact on the natural environment, 3rd Ed. The MIT Press, Cambridge, Massachusetts.

Goward, S. N., Masek, J. G., Williams, D. L., Irons, J. R., and Thompson, R. J. (2001). The Landsat 7 mission: Terrestrial research and applications for the 21st century. *Remote Sensing of Environment,* 78, 3-12.

Halwas, K. L, Church, M. (2002). Channel units in small, high gradient streams on Vancouver Island, British Columbia. *Geomorphology* 43: 243–256.

Harding, J. S., Young, R. G., Hayes, J. W., Shearer, K. A., and Stark, J. D. (1999). Changes in agricultural intensity and river health along a river continuum. *Freshwater Biol.* 42:345–357.

Horton, R. E. (1933), The role of infiltration in the hydrologic cycle. *Trans. Amer. Geophys. Union* 14:446-460.

Hill, A. R. (1981). Nitrate-nitrogen flux and utilization in a stream ecosystem during low summer flows. *Canadian Geographer* 35: 225–239.

Hunsaker, C. T., and Levine, D. A. (1995). Hierarchical approaches to the study of water quality in rivers. *Bioscience*, 45(3), 183–192.

Jenson, S., K., and Domingue, J., O. (1988). Extracting topographic structure from digital elevation data for geographic information system analysis. *Photogrammetric Engineering and Remote Sensing*, 54, 1593-1600.

Johnson, L. B., and Gage, S. H. (1997). Landscape approaches to the analysis of aquatic ecosystems. *Freshwater Biology* 37:113–132.

Johnson, L. B., and Richards, C., Host, G., and Arthur, J. W. (1997). Landscape influences on water chemistry in Midwestern streams. *Freshwater Biology* 37: 193-208.

Kibler, D. F. (ed.), (1982), Urban stormwater hydrology. American Geophysical Union, Washington, DC.

Lammert, M., and Allan, J. D. (1999). Assessing biotic integrity of streams: Effects of scale in measuring the influence of land use/cover and habitat structure on fish and macroinvertebrates. *Environ.Manage.* 23:257–270.

Ligon, F. K., Dietrich, W. E., & Trush, W. J. (1995). Downstream ecological effects of dams: A geomorphic perspective. *Bioscience*, 45, 183–192.

Likens, G. E, Bormann, F. H, Pierce, R. S, Eaton, J.S, Johnson, N. M (1977). *Biogeochemistry of a Forested Ecosystem*. New York: Springer-Verlag.

Machemer, P.L., Kaplowitz, M, D., Edens, T, C. (1999). Managing Growth and Addressing Urban Sprawl, An Overview. 1999, Research Report, Michigan Agricultural Experiment Station, Michigan State University.

Mark, D. M. (1983). Automated detection of drainage networks from digital terrain models, Proc.Auto-Carto 6, Automated Cartography: International Perspectives on Achievements and Challenges. Ottawa, Ontario, 288-298.

Masek, J. G., Honzak, H., Goward, S. N., Liu, P., and Pak, E. (2001). Landsat-7 ETM+ as an observatory for land cover: Initial radiometric and geometric comparisons with Landsat-5 Thematic Mapper. *Remote Sensing of Environment,* 78, 118-130.

Meyer, J. L., Wallace, J. B. (2001). Lost linkages and lotic ecology: Rediscovering small streams. Pages 295–317 in Press MC, Huntly NJ, Levin S, eds. Ecology: Achievement and Challenge. Oxford (United Kingdom): Blackwell Scientific.

Milesi, C., Elveridge, C. D., Nemani, R. R, and Running, S. W. (2003). Assessing the impact of urban land development on net primary productivity in the South eastern United States. *Remote Sensing of the Environment.*, 86, 401-410.

Michigan Land Cover /Use Classification System (2000).

Moore, I. D., Grayson, R. B., Ladson, A. R. (1991). Digital terrain modeling: A review of Hydrological, Geomorphological, and Biological Applications. *Hydrological Processes*, Vol.5, 3-30.

Mueller, D. K and Helsel, D. R. (1996). Nutrients in the Nation's waters- too much of a good thing? U.S. Geological Survey Circular 1136.

Natural Resources Inventory. (1987). Michigan Data. USDA Soil Conservation Service, East Lansing, Michigan.

Neckel, H and D. Labs. (1984). The solar radiation between 33000 and 125000. *Solar Physics*, 90:205-258.

O'Callaghan, J., F., and Mark, D. M. (1984). The extraction of drainage networks from digital elevation data, *Computer Vision, Graphics and Image Processing*, 28, 323-344.

O'Neal, R. P. (1997). Muskegon River Watershed Assessment. Michigan Department of Natural Resources, Fisheries Division. Fisheries Special Report 19. 187 pp.

O'Neill, R. R., Christensen, S. W., Dale, V. H., De Angelis, D. L., Gardner, R. H., Graham, R. L, Jackson, B., Krummel, J. R., Milne, B. T., Sugihara, G., Turner, M. G., Zygmyunt, B. (1988). Indices of Landscape pattern, *Landscape Ecology*, 1(3): 153-162.

Omernick, J. M., Abernathy, A., R., and Male, L. M. (1981). Stream nutrient levels and proximity of agricultural and forest land to streams: some relationships. *Journal of Soil and Water Conservation* 36:227-231.

Osborn, L. L., and Wiley, M. J. (1988). Empirical relationships between land use/land cover and stream quality in an agricultural watershed. *Journal of Environmental Management* 26: 9-27.

Paulsen, S. G., Larsen, D. P., Kaufmann, P. R., Whitier, T. R., Baker, J. R., Peck, D. V., McGue, J., Hughes, R. M., McMullen, D., Stevens, D., Stoddard, J. L., Lazorchak, J., Kinney, W., Selle, A. R., and Hjort, R. (1991). Environmental Monitoring and Assessment Program: EMAP surface waters monitoring and research strategy- fiscal year 1991. EPA/600/3-91/022.United States Environmental Protection Agency. Washington, D. C.

Pijanowski, B. C., Shellito, B., Pithadia, S., Alexandridis, K. (2002). Forecasting and assessing the impact of urban sprawl in coastal watersheds along eastern Lake Michigan, *Lakes & Reservoirs: Research and Management* 7: 000–000 (in press).

Poff, N. L., Allan, J. D., Bain, M. B., Karr, J. R., Prestegaard K. L., Richter, B. D., Sparks, R.E., Stromberg, J.C. (1997). The natural flow regime: A paradigm for conservation and restoration of river ecosytems. *BioScience* 47: 769–784.

Poff, N. L., Hart, D. D. (2002). How dams vary and why it matters for the emerging science of dam removal. *BioScience* 52: 659–668.

Prevost, M., Plamondon, A. P., Belleau, P. (1999). Effects of drainage of a forested peatland on water quality and quantity. *Journal of Hydrology*214: 130-143.

Rabeni, C. F., and Smale, M. A. (1995). Effects of siltation on stream fishes and the potential mitigating role of the buffering riparian zone. *Hydrobiologia* 303:211–219.

Rabus, B., Eineder, M., Roth, A., Bamler, R. (2003). The shuttle radar topography mission—a new class of digital elevation models acquired by spaceborne radar. *ISPRS Journal of Photogrammetry & Remote Sensing,* 57, 241– 262.

Richards, C., Host, G. E., and Arthur, J. W. (1993). Identification of predominant environmental factors structuring stream macroinvertebrate communities within a large agricultural catchment. *Freshwater Biol.* 29:285–294.

Richardson, R. E. (1928). The bottom fauna of the Middle Illinois River, 1913-1925. *Illinois Natural History Survey Bulletin* 17: 387-475.

Sidle, R. C, Tsuboyama, Y., Noguchi, S., Hosoda, I., Fujieda, M., Shimizu, T. (2000). Streamflow generation in steep headwaters: A linked hydro-geomorphic paradigm. *Hydrological Processes* 14: 369–385.

Soranno, P. A., Hubler, S. L., Carpenter, S. R., and Lathrop, R. C. (1996). Phosphorus loads to surface waters: a simple model to account for spatial patterns of land use. *Ecological Applications* 6: 865-878.

Steuer, J., Selbig, W., Hornewer, N., and Prey, J. (1997). Sources of Contamination in an Urban Basin in Marquette, Michigan and an Analysis of Concentrations, Loads, and Data Quality. U.S. Geological Survey, Water-Resources Investigations Report 97-4242.

Strahler, A. N. (1957). Quantitative Analysis of Watershed Analysis of Watershed geomorphology. *Transactions of the American Geophysical Union.*

Sun, G., Ranson, K. J, Kharuk, V. I., Kovacs, K. (2003). Validation of surface height from shuttle radar topography mission using shuttle laser altimeter, *Remote Sensing of Environment,* 88, 401-411.

Teillet, P. M., Barker, J. L., Markham, B. L., Irish, R. R., Fedosejevs, G. and Storey, J. C. (2001). Radiometric cross-calibration of the Landsat-7 ETM+ and Landsat-5 TM sensors based on tandem data sets. *Remote Sensing of Environment,* 78, 39-54.

Tou, J. T., and Gonzalez, R. C. (1974). *Pattern Recognition Principles*. Reading, Massachusetts: Addison-Wesley Publishing Company.

Turner, M. G., Gardner, R. H., O'Neill, R. V. (2001). Landscape Ecology In Theory and Practice: Pattern and Process. New York: Springer Verlag.

Tsukamoto, Y., Ohta, T., Noguchi H. (1982). Hydrological and geomorphological study of debris slides on forested hillslope in Japan, ed. *Recent Developments in the Explanation and Prediction of Erosion and Sediment*

*Yield*. International Association of Hydrological Sciences publication no. 137, 89-98, Walling D E.

U. S. Environmental Protection Agency. (1996). Strategic Plan for the Office of Research and Development. EPA/600/R-96/059. Washington, D. C.

U. S. Environmental Protection Agency. (1997). Environmental Monitoring and Assessment Program (EMAP) (http://www.epa.gov/emap/html/about.html).

U. S. Environmental Protection Agency. (1997). Index of Watershed indicators: A Overview (http://www.epa.gov/iwi/iwi-overview.pdf).

U.S.Geological Survey. (1999). The Quality of Our Nation's Waters- Nutrients and Pesticides: U.S.Geological Survey Circular 1225, 82p.

U.S.Geological Survey. (2000). US GeoData Digital Elevation Models, 040-00 (April 2000) http://erg.usgs.gov/isb/pubs/factsheets/fs04000.html.

Veatch, J. O. (1953). Soil and land of Michigan. Memoir (Michigan State College. Agicultural Experiment Station), no.7.

Vitousek, P. M. and Howarth, R. W. (1991). Nitrogen limitation on land and in the sea: how can it occur? *Biogeochemistry* 13:87-115.

Vogelmann, J. E, Sohl, T., Howard, S. M. (1998). Regional Characterization of Land Cover Using Multiple Sources of Data. *Photogrammetric Engineering & Remote Sensing*, Vol.64, 1, 45-57.

Wallace, J. B., Grubaugh, J. W., and Whiles, M. R. (1996). Biotic indices and stream ecosystem processes: results from an experimental study. *Ecological Applications* 6: 140-151.

Waschbusch, R., Selbig, W., and Bannerman, R. (2000). Sources of phosphorus in stormwater and street dirt from two urban residential basins in Madison, Wisconsin, 1994-1995. In: National Conference on Tools for Urban Water Resource Management and Protection. US EPA February 2000: pp. 15-55.

Whipple, K. X., Hancock, G. S., and Anderson, R. S. (2000). River incision into bedrock: Mechanics and relative efficacy of plucking, abrasion, and cavitation. *Geol. Soc. Am. Bull.*, 112, 490–503.

Wickham, J. D., Wade, T. G., Riitters, K. H., O'Neill, R. V., Smith, J. H., Smith, E. R., Jones, K. B., and Neale, A. C. (2003). Upstream-to-downstream changes in nutrient export risk, *Landscape Ecology*. 18: 195-208.

Wipfli, M. S. (1997). Terrestrial invertebrates as salmonid prey and nitrogen source in streams: Contrasting old-growth and young-growth riparian forests in southeastern Alaska, USA. *Canadian Journal of Fisheries and Aquatic Sciences* 54: 1259–1269.

Wipfli, M. S. and Gregovich, D., P. (2002), Invertebrates and detritus export from fishless headwater streams in southeastern Alaska: Implications for downstream salmonid production. *Freshwater Biology* 47: 957–970.

Wolock, D., M., Hornberger, G. M., and Musgrove, T., J. (1990). Topographic effects on flow path and surface chemistry of the Llyn Brianne catchments in Wales. *J.Hydrol.*, 115, 243-259.

Wolock, D. M. and Price, C. V. (1994). Effects of digital elevation model map scale and data resolution on a topography-based watershed model. Water resources research, vol 30, 11, 3041-3052.

Zimmerman, A. and Church, M. (2001). Channel morphology, gradient profiles and bed stresses during flood in a step-pool channel. *Geomorphology*, 40: 311–327.

Zhang, W. and Montgomery, D. R. (1994). Digital elevation model grid size, landscape representation and hydrologic simulations. *Water Resources Research,* Vol 30, 4, 1019-1028.