Use of GIS Modeling Techniques as a Planning Tool for Establishment of Wetlands as Nitrate and Pesticide Removal Facilities

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ABSTRACT

Wetland vegetation and denitrification processes may offer solutions to the problem of the increasing nitrate and pesticide contamination of aquifers underlying irrigated agricultural fields. These technologies: remote sensing, geographic information systems, nitrate leaching simulation modeling, and wetland identification and analysis, can be incorporated into a unified approach to evaluate non-point source treatment by wetland-based biological and physical processes and the possible effects of these processes on the regional aquifer system.

Tests from irrigation wells throughout the South Platte River basin for nitrates indicate there are "hot spots" where nitrates are above the drinking water standards of 10 ppm (parts per million) allowable for human consumption. These excessive concentrations call for research on causes and possible methods for reducing ground water nitrate concentrations for future water uses. The focus of this study is to identify methodologies to identify the sources of nitrates (from fertilizers and feedlot manures); the transport routing and mechanisms; and the potential effect of biological-chemical-physical treatment and remediation processes associated with existing or planned wetlands in agricultural regions. The study area covers 13,761 ha (34,005 acres), and is located approximately 10 km (6 miles) southwest of Greeley, Colorado. The study area contains a 160 ha (400 acre) oxbow wetland.

Satellite and rasterized NAPP aerial photography were classified and used as landcover base mapping layers of a GIS. Digital elevation models (DEM) and well sampling data were used as additional GIS inputs for modeling the aquifer and wetland. For definition of the wetland service area, a groundwater flow and particle tracking model was used to define the area contributing flow into the aquifer.

Assuming an average annual nitrate leaching rate of 100 kg/ha (89.2 lb/acre) for all irrigated crops in the study area (cataloged in Table 2), the total nitrogen loading rate for the study area is 865,000 kg (as nitrogen) (953 tons) from agricultural sources. Similarly calculated loading rates for the wetland service area are 71,200 kg/yr of nitrogen in 2.2 million cubic meters (1,800 ac-ft) of water. Preliminary sampling shows a trend toward lower concentrations of nitrates in water exiting the wetland compared to the groundwater aquifers surrounding and feeding the wetland.

INTRODUCTION

This study describes technologies that can be used to model the potential effect of wetland denitrification processes in regional hydrologic systems. Wetland vegetation and denitrification processes may offer solutions to the problem of increasing nitrate and pesticide contamination of alluvial aquifers underlying irrigated agricultural fields. These technologies; remote sensing, geographic information systems, nitrate leaching simulation modeling, and wetland identification and analysis, can be incorporated into a unified approach to evaluate non-point source treatment through wetland-based biological and physical processes. Identification of methodologies to identify the sources of nitrates (from fertilizers, feedlot manures), pesticides and herbicides (from agricultural and urban sources), the transport routing and mechanisms, and the potential role of biological-chemical-physical treatment and remediation processes is the focus of this study. Figure 1 shows the location of the study area and the wetland selected for detailed study.

The literature documents that denitrification and attenuation of pesticides takes place in natural or environmental conditions. The wetlands environment may provide conditions conducive to removal of nitrates. This study does not investigate nor document the wetland mechanism responsible for the actual removal of the contaminant, but 1) identifies some of the possible steps to be taken in acquiring the necessary data, 2) the procedures for processing the data, and 3) the hardware/software and manpower necessary to conduct a study of wetland uptake and non-point treatment of the contaminants. The steps necessary to determine the nitrate and pesticide loading to the wetlands either through surface or through the leaching/transport through the alluvial aquifer forms a large part of the study effort.

Wetlands have been shown to uptake nitrates in both the benthic sediments and the vegetation. The effect is quantitatively uncertain to date. However, many riverine and perennial wetlands exist in nitrate affected systems, such as the Platte River floodplain discussed here, due to natural alluvial geomorphology, gravel mining activities, and irrigation surface runoff.
These many wetlands, if managed correctly, could be a valuable asset for minimizing impacts of nitrate loading.

RESEARCH PROBLEM AND OBJECTIVES

The research problem is *to identify methodologies to evaluate the potential of wetlands as a treatment device for reduction of nitrate non-point source pollution effects.*

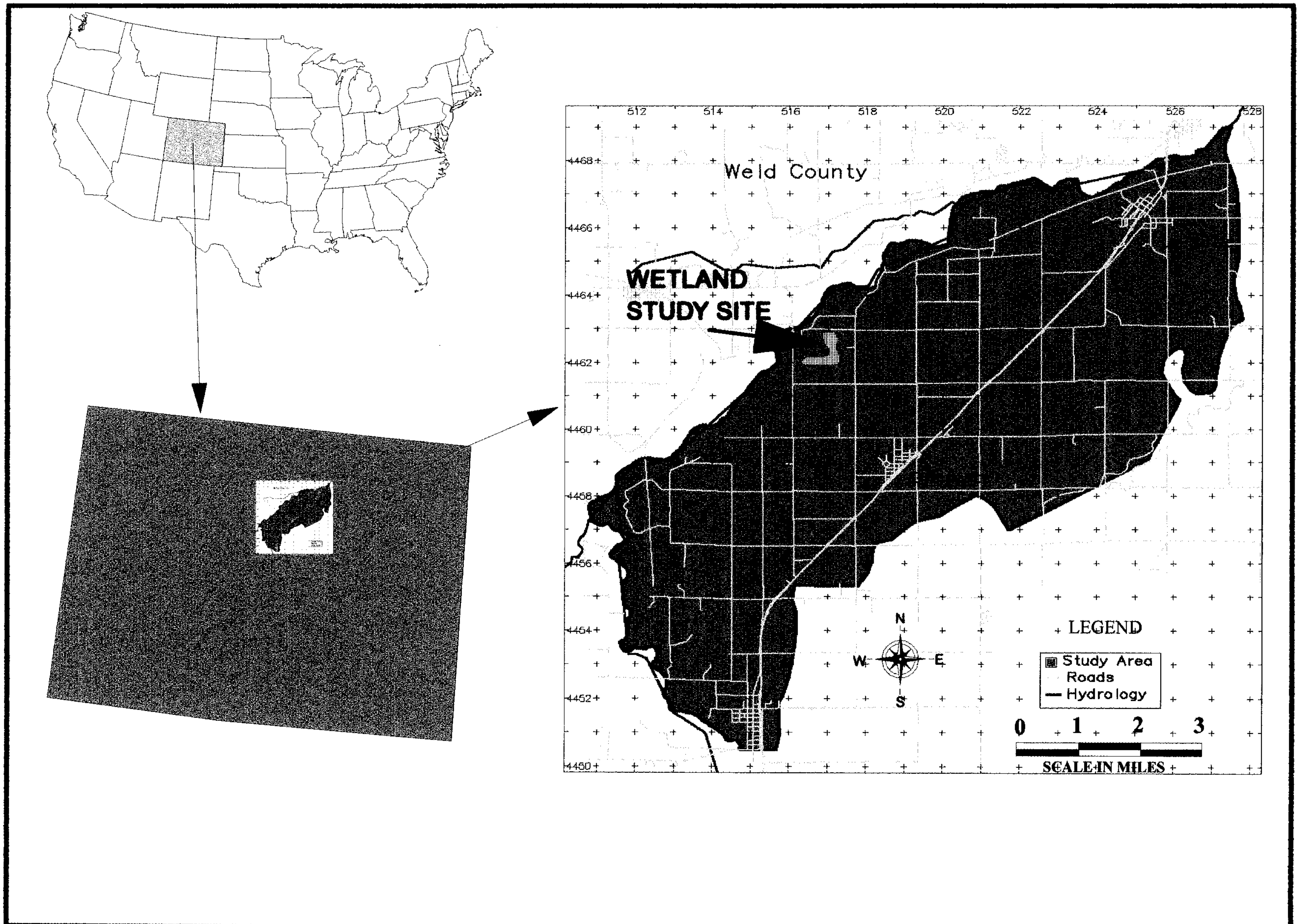


Figure 1. Study area.

The objectives of this research are:

1) To develop a methodology for evaluating management plans to use on-farm and riverine wetlands to control or reduce nitrates and pesticides and test the methodology by developing a model for a selected wetland.

2) To use remotely sensed data derived from aerial photography and satellite imagery and automated computer classification techniques (ACCT) to identify off-stream and riverine wetlands in a portion of the Poudre River and South Platte Basin.

3) To use geographic information systems (GIS) for modeling and to quantify the following:

a. The spatial distribution and area of wetlands in the river basins,
b. the vegetation types and physical characteristics of the wetland areas,
c. the irrigated agricultural areas serviced by the wetlands,
d. the amount of consumptive water use attributed to the wetlands, and,
e. the nitrate loading to the identified wetland.

BACKGROUND

Non-point source contamination of ground water aquifers from irrigated agriculture varies significantly as a result of differing management styles, soils, and climate. A large portion of the irrigated agricultural lands in Colorado produce nitrogen leaching that contributes to eutrophication and increased salinity of the Platte, Arkansas and Colorado rivers and their tributaries. For example, significant portions of the South Platte River and its alluvial aquifer have nitrate levels in excess of drinking water standards (North Front Range Water Quality Planning Association, 1990). A significant portion of the surface and ground water nitrate levels come from non-point source agricultural fertilizer inputs, and municipal waste treatment facilities in the Colorado Front Range contribute point-source nitrates to surface waters. As a result, use of alluvial aquifers for livestock and domestic consumption is either threatened or contaminated (Wylie *et al.*, 1993).

GEOLOGY AND GEOGRAPHY

The South Platte River originates in the Rocky Mountains southwest of Denver Colorado and enters the Colorado Piedmont section of the Great Plains physiographic province about 25 miles southwest of Denver. The drainage basin of the South Platte is characterized by stream-valley lowlands separated by gently rolling uplands (Smith *et al.*, 1964.) The South Platte then flows north-northeastward then turns eastward at Greeley, Colorado about 50 miles north of Denver. According to Smith *et al.* (1964), the flood plain of the South Platte averages about a mile in width and is characterized by an irregular surface of abandoned meander scars, swamps, oxbow lakes and low indistinct terraces. Two distinct terraces separated by well defined escarpments on both sides of the river extend well into the major tributaries which include Clear Creek, St. Vrain Creek, the Big Thompson River and the Cache la Poudre River. The Kuner Terrace and the Kersey Terrace are approximately 10-15 feet and 20-40 feet above the river surface respectively. These terraces play an important role in the irrigated agriculture of the South Platte Basin.

The South Platte River valley has moderately cold winters and warm summers with an average of about 12.5 - 13.5 inches annual precipitation. May has the highest monthly precipitation, averaging 2.0 - 2.5 inches. A majority of the annual precipitation falls as rain during the spring and summer months. The alluvial valley floor is intensely cultivated and irrigated with water diverted from the South Platte and its tributaries and from irrigation wells supplied from the alluvial aquifer.

The bedrock is composed of consolidated sedimentary rocks overlying crystalline Precambrian rock. Volcanic ash from extinct volcanos of the Rocky Mountains are found occasionally in the sedimentary layers of the valley. Unconsolidated deposits of valley fill, dune sand and slope-wash characterize the valley aquifers (Smith *et al.*, 1964). Smith *et al.* (1964) report from studies conducted in the 1950's that the depth to water table on the flood plain of the South Platte is generally less than 10 feet with the capillary fringe extending to the root zone or to the land surface. Due to the confining of the impervious bedrock underlying the unconsolidated valley floor sediments, any change in the volume of water stored in the valley floor aquifers is characterized by a change in the water table.

The present-day water table levels are quite different than during the early years of the settlement of the South Platte basin. Edwin James, the official botanist for the Steven H. Long scientific exploration of the South Platte in 1819-1820, wrote that the South Platte was about 900 yards wide, flowing but very shoaly, and nearly destitute of vegetation with areas of effloresced salt. When the expedition reached a point two to three days journey from the mountains, they found a "narrow strip of timber, extending along the immediate banks of the river, never occupying the space of half a mile in width." (Silkensen, 1993)

Fremont, in an exploration of the mid reaches of the South Platte in July 1842, found the South Platte completely dry. Even with a dry river, Fremont noticed "tolerably large groves of timber." On June 30, 1843 (the following year), coming back across the watershed from the Republican River (author's note: perhaps in the region of Sterling to Julesberg), Fremont commented that they observed a large valley with the South Platte "rolling magnificently along, swollen with the water of the melting snows" (Silkensen, 1993).

In an 1858 guide book, *The New Gold Mines of Western Kansas*, written by William B. Parsons, the South Platte was described: "There is no wood of any account between O'Fallon's Bluff [near the confluence of the North Platte and South Platte] and Fort St. Vrain. Water will be found without difficulty the whole way". Another traveler's commentary in 1862 mentioned that there was no timber in the South Platte Valley, but the islands in the South Platte were "covered with high grass and all have trees, some evergreen.... Came to Bijou Creek [author's note: perhaps in the vicinity of Wiggins, Colorado] and found the first timber for 8 days." (Silkensen, 1993)

Irrigation developed in the South Platte basin in the 1870's. Silkensen (1993) describes the first diversion of water in Clear Creek in 1859, and in 1860 on the Cache la Poudre, St. Vrain and Left Hand Creeks. After a few years of irrigating, farmers noticed an increase in the volume of water in the channels below the irrigated sections. By 1883, the State Engineer's report noted that the flow of water in the Platte River was much more uniform. Parshall documented in a 1922 paper that the river's discharge had increased and had a much more uniform flow. He commented that the storage reservoirs, built to store a portion of the spring runoff "contributed approximately 20 percent to the return flow to the river by seepage" (Silkensen, 1993).

Silkensen concludes that river conditions have changed significantly over the past 130 years, with wetter soil conditions prevailing at present. Riparian vegetation has increased and now constrains the river channel so that the river is no longer free to scour the channel and distribute the sediment load over the flood plain.

The point in presenting the above description of the South Platte River environment is to document the extreme differences between the South Platte River of 1865 and the South Platte River of 1995. The modern river is a product of man's influences over 130 years and the riparian zones of 1995 are quite different than the broad, meandering, torrential flooding South Platte of 1865. The wetlands described in this report are the result of man's influences in converting an ephemeral river to a perennial river with continuous flow and perennial wetlands. It is these wetlands that are the focus of this research.

NITROGEN LEACHING "HOT SPOTS"

Urban municipal waste water treatment facilities, agricultural fertilizer, and other agricultural non-point sources contribute to the nitrate nitrogen concentrations in ground water and in the alluvial aquifers fed by and connected to the South Platte River and its tributaries. Because of the leaching of fertilizers during periods of high precipitation or during the irrigation season, the alluvial aquifer has increased concentrations of pesticides, industrial chemicals and nitrates. This report will focus only on nitrates supplied to the aquifers from agricultural inputs.

Tests from irrigation wells throughout the South Platte River basin for nitrates indicate there are "hot spots" where nitrates are above the drinking water standards of 10 ppm (parts per million) allowable for human consumption. It is these excessive concentrations that focus the research on causes and possible methods for reducing ground water nitrate concentrations for future water uses. Use of alluvial aquifer water supplies for livestock watering as well as the expense of providing higher quality water from distant sources for human consumption are concerns for area residents. Figure 2 shows a map of nitrate concentrations in the South Platte River alluvial aquifers derived from well sampling and nitrate analysis.

Wylie *et al.*, (1993) used NLEAP (Nitrate Leaching and Economic Analysis Package) and GIS techniques to estimate the potential and amount of nitrates leached from areas north and south of Greeley, Colorado to correlate estimates from the NLEAP computer model with actual ground water nitrate concentrations (Wylie *et al.*, 1993). Two outputs of the NLEAP program are the nitrate leached (NL) and the annual leaching risk potential indices (ALRP). The research objectives of the Wylie study were to use crop maps generated from computer processing of satellite data sets of the study area and information on crop rotations, cropping patterns, soils characteristics, climate and other GIS data layers to estimate the two indices identified above. The goal was to use computer simulations within a GIS to predict the locations of high nitrate concentrations or "nitrate hot spots," that could occur in ground water aquifers under certain agricultural practices. The proof of concept would be the matching of the computer predicted "hot spots" with actual hot spots determined from sampling irrigation wells in the study area. Statistical methods were used within the context of the GIS map outputs to determine if there was a correlation between irrigation methods, soil types, fertilizer application practices, and adjacency of feedlots to the "nitrate hot spots" in the aquifer. The report concluded that location relative to feedlots and soil types gave the strongest correlation but singly or combined, these two factors only accounted for up to 41 percent of the variability in ground water nitrate concentrations.

NITROGEN PROCESSES

This research is premised on the existence of elevated nitrate nitrogen concentrations in the alluvial aquifer and the need to reduce the nitrate concentrations over time to improve the quality of the aquifer waters and return flows to the South Platte. Denitrification, or biochemical conversion of nitrate nitrogen to gaseous nitrogen, is not well understood due to the difficulty in replication of the various denitrification processes in a variety of saturated and unsaturated environments. A very brief review of nitrogen chemistry in saturated conditions found often in wetland and riparian environments is provided, with an emphasis placed on the chemistry and biochemistry of denitrification. The inclusion of denitrification processes in this section of the report is directed toward an understanding of the processes so that any management process can be directed to the exploitation of N cycles as a sink for N in ground water.

Two denitrification environments are of interest here: 1) aquifers, and 2) wetland and riparian zones where aquifer water is in transition to surface waters or the reverse (wetlands receiving the water and recharging the nitrate laden waters back into the aquifer).

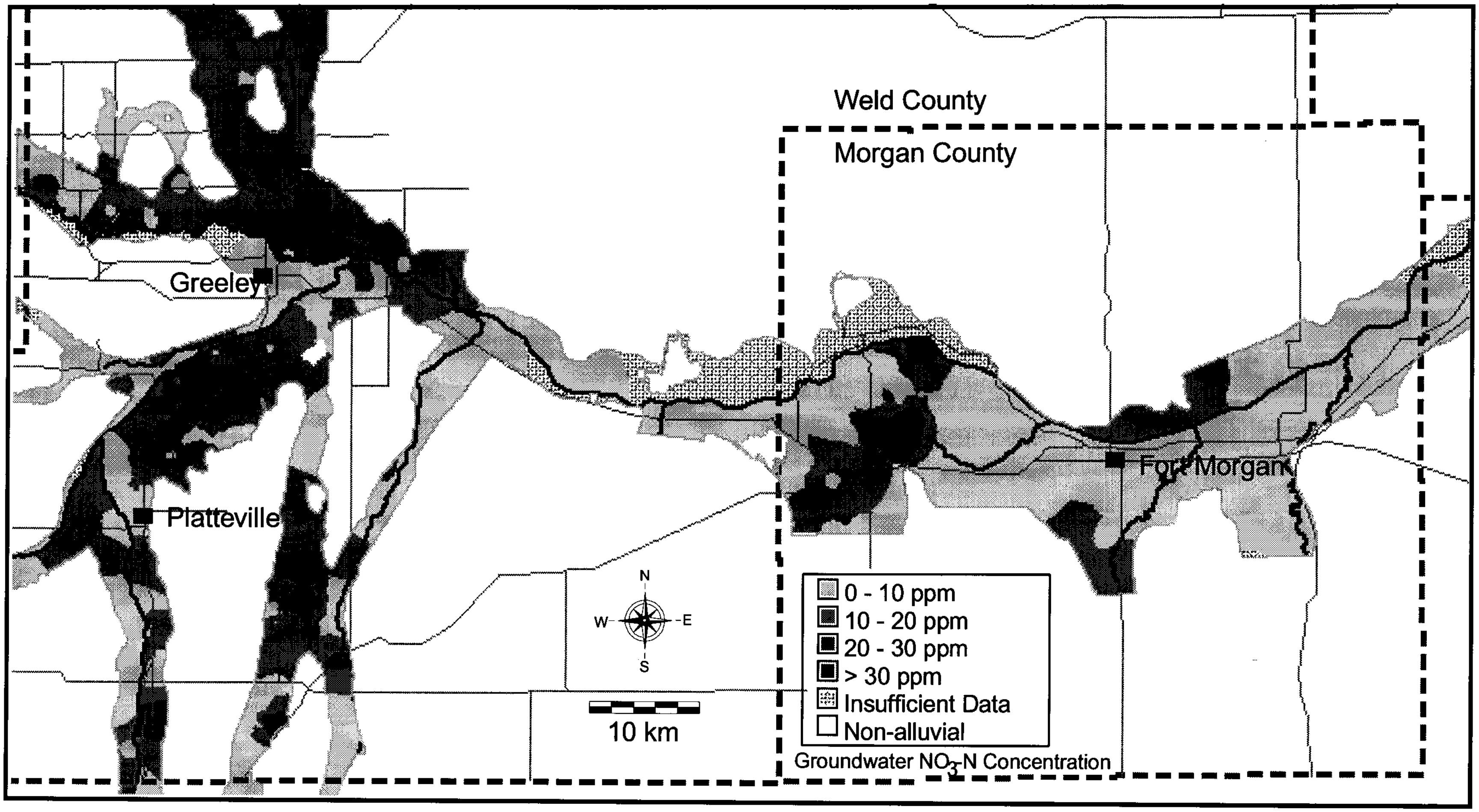


Figure 2. Groundwater nitrate concentration map: South Platte River Basin.

Aquifer Denitrification. Denitrification in aquifers or saturated zones is characterized by autotrophic or heterotrophic processes. An example of a heterotrophic denitrification process follows (Kölle et al., 1983):

$$5C + 4NO_3^- + 2H_2O = 2N_2 + 4HCO_3^- + CO_2 \tag{1}$$

In the above expression, C represents an arbitrary organic compound with an oxidation state of zero. Naturally occurring organic compounds are varied (Thurman, 1985) and the C atoms may have different oxidation states (Stumm and Morgan, 1981). Trudell *et al.* (1986) used a geochemical model to compare the production of HCO_3^- from glucose as the organic e- donor and found that the production of HCO_3^- from glucose in heterotrophic denitrification did agree with the model. The model was represented by:

$$4NO_3^- + 5/6C_6H_{12}O_6 + 5CaCO_3 + 4H^+ = 2N_2 + 10HCO_3^- + 5Ca^{2+} + 2H_2O \tag{2}$$

Starr and Gilliam (1989) performed a similar test in an agricultural site and found no denitrification in spite of the presence of nitrate and denitrification bacteria. The lack of denitrification was due to the lack of organic carbon (OC). Their explanation was that deep water table conditions prevented OC from reaching the saturated zone through the unsaturated zone.

Aquifers containing Mn^{2+}, Fe^{2+}, and HS^- can serve as e^- donors with NO_3^- acting as an e^- acceptor in autotrophic denitrification in reducing zones in aquifers. Lind (1983) reported that ground water containing Fe^{2+} never contained NO_2^- but did not report whether the Fe^{2+} abiotically reduced the NO_2^- or whether denitrifying bacteria were the cause of the removal of the NO_2^-. Gouy *et al.* (1984) reported that *Gallionella ferruginea* reduces NO_3^- to NO_2^- abiotically in an OC-poor environment. The reduction of NO_2^- is to a gaseous N compound occurs by one of several possible reactions including:

$$\begin{gathered} HNO_2 + Fe^{2+} + H^+ = NO + Fe^{3+} + H_2O \text{ and} \\ 2NO + 2Fe^{2+} + 2H^+ = N_2O + 2Fe^{3+} + H_2O \end{gathered} \tag{3}$$

or

$$2HNO_2 + 6Fe^{2+} + 6H^+ = N_2 + 6Fe^{3+} + 4H_2O \tag{4}$$

Sulfur compounds may also act as an e^- donor with NO_3^- as an e^- acceptor. Strebel and Böttcher (1989) reported inputs of large quantities of NO_2^- to aquifers under agricultural lands (30 mg/l), but the concentration of NO_3^- from wells ranged from 0.05 to 0.3 mg/l NO_3^-. The reaction:

$$5FeS_2 + 14NO_3^- + 4H^+ = 7N_2 + 10SO_4^{2-} + 5Fe^{2+} + 2H_2O, \tag{5}$$

is mediated by *Thiobacillus denitrificans*. (Kölle *et al.*, 1983; Kölle *et al.*, 1985)

Denitrification rates reported by the researchers [Starr and Gilliam (1989); Trudell *et al.* (1986)] range from 0.19 to 0.58 mg N L^{-1} d^{-1}. Kölle *et al.* (1985) reported a first order reaction with a half life of 1.2 to 2.4 years.

Denitrification in aquifers seems to be dependent either on the amount of organic carbon sources for heterotrophic denitrification or the presence of iron, manganese or sulfur for autotrophic denitrification. A survey of the literature does not indicate a clear way to determine the difference between autotrophic and heterotrophic processes, although both processes may proceed concurrently.

Wetlands as Treatment Devices for Nitrates. The discussion of nitrification and denitrification must take place in the context of the research problem. The South Platte aquifers have excessive concentrations of nitrate-N. The nitrogen limited flooded wetland soils reported by (Gambrell and Patrick 1978; Sullivan and Daiber, 1974; Valiela and Teal, 1974; Klopatek, 1978; and Simpson *et al.*, 1978) are not the norm for the study area.

Figure 3 shows a schematic of the nitrogen processes taking place within flooded wetland soils. The primary form of mineralized nitrogen in flooded wetland soils is the ammonium ion (NH_4^+). Much of the nitrogen can also be tied up in the form of organic nitrogen compounds in highly organic soils. Mineralization of nitrogen results from the biological transformation of organically combined nitrogen to ammonium nitrogen during biological degradation of organic matter. This transformation can occur under both anaerobic and also aerobic conditions and is called ammonification (Gambrell and Patrick, 1978). The transformation for urea, an organic compound of nitrogen, is shown as follows:

$$NH_2{\cdot}CO{\cdot}NH_2 + H_2O \rightarrow 2NH_3 + CO_2 \qquad (6)$$

The ammonium ion can be absorbed by the plant root systems, or converted back to organic matter by anaerobic bacteria. It can also be immobilized through ion exchange onto negatively charged soil particles (Mitsch and Gosselink, 1993). Because of the thin oxidized layer at the surface of many wetland soils, the ammonium ion does not build to excessive levels due to oxidization at this thin surface layer. The ammonium ion is oxidized through the process of nitrification by two bacterium, Nitrosomonas sp. and Nitrobacter sp. in the following transformations:

$$\text{Nitrosomonas sp.} \quad 2NH_4^+ + 3O_2 \rightarrow 2NO_2^- + 2H_2O + 4H^+ + \text{energy} \qquad (8)$$

$$\text{Nitrobacter sp.} \quad 2NO_2^- + O_2 \rightarrow 2NO_3^- + \text{energy} \qquad (9)$$

The oxidized rhizosphere of plants can oxidize ammonium ions in the presence of sufficient oxygen (Reddy and Graetz, 1988).

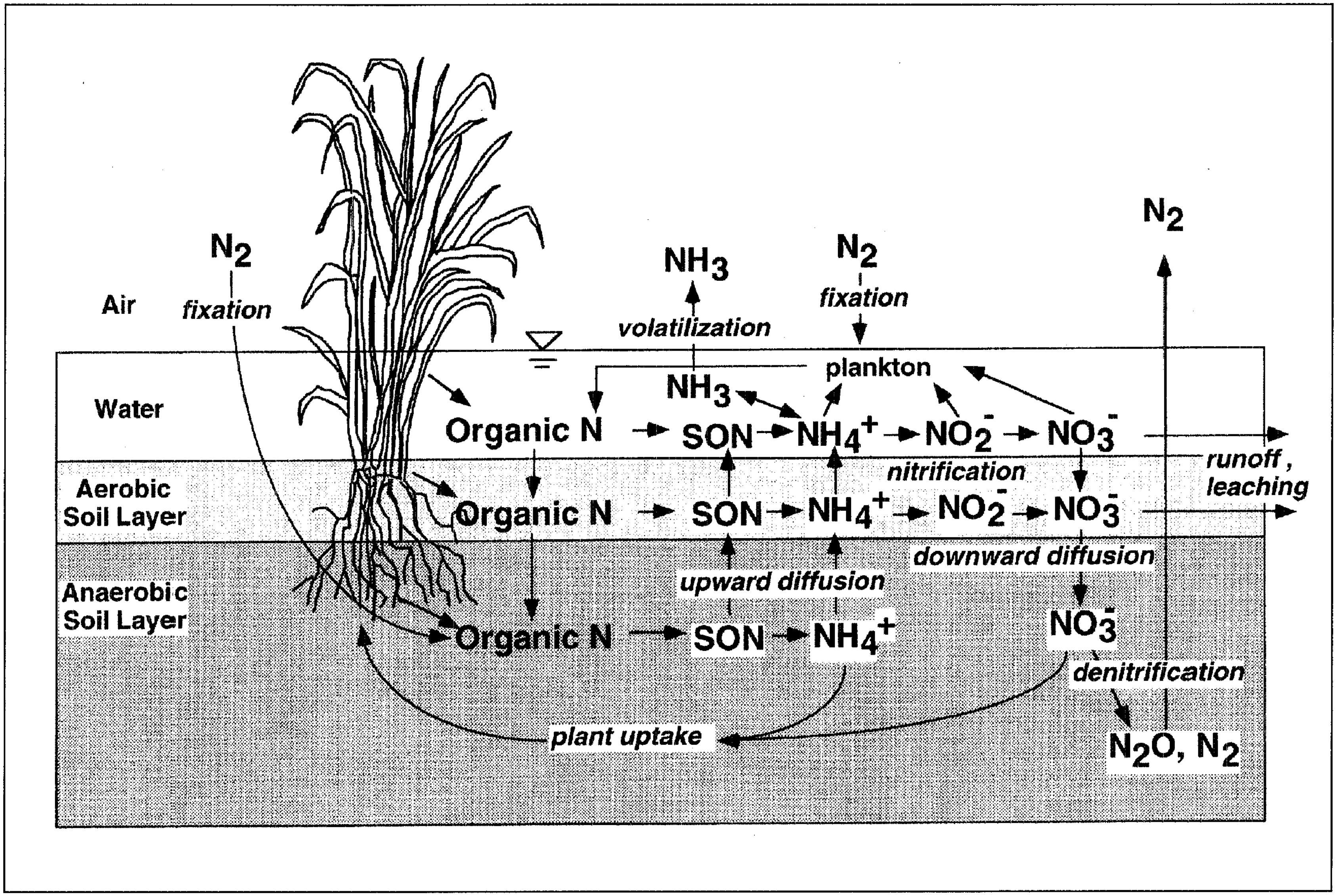


Figure 3. Nitrogen transformations in wetlands. SON indicates soluble organic nitrogen. (from Mitch and Gosselink, 1993)

Denitrification, represented also in equation (2) above, is carried out by microorganisms under anaerobic conditions, where nitrate acts as a terminal electron acceptor and is converted to gaseous nitrous oxide (N_2O) and molecular nitrogen (N_2) as follows:

Denitrification in wetlands may be significant. A significant loss of nitrogen in salt marshes through denitrification has been documented by Kaplan *et al.* (1979) and Whitney *et al.* (1981). Rice culture denitrification has been documented by Patrick and Reddy (1976) and Mohanty and Dash (1982). The process of denitrification occurs in the following sequence: the ammonium-nitrogen diffuses to the aerobic soil layer; nitrification occurs; nitrate-nitrogen diffuses back into the anaerobic layer and then denitrification occurs in the anaerobic layer (Mitsch and Gosselink, 1993). Reddy and Gratz (1988) and Reddy and Patrick (1984) reported that nitrate-nitrogen diffusion is seven times faster than ammonium-nitrogen diffusion in wetland soils, therefore the entire process of denitrification is limited by ammonium and nitrate nitrogen diffusion processes. Denitrification processes are inhibited in acidic soils and peat (Etherington, 1983).

In contrast, nitrogen fixation may be a significant source of organic nitrogen in some wetlands (Mitsch and Gosselink, 1993). The conversion of N_2 gas to organic nitrogen is facilitated through microorganisms in the presence of the enzyme nitrogenase and is favored in zones of low oxygen tension (Etherington, 1983) because nitrogenase activity is inhibited by high oxygen. Reddy and Graetz (1988) reported that nitrogen fixation may occur in wetlands in overlying waters, in the aerobic soil layer, in the anaerobic soil layer, in the oxidized rhizosphere of the plants, and on the stem and leaf surfaces of the plants. Whitney *et al.* (1975, 1981) and Teal *et al.* (1979) documented that bacterial fixation is the most significant pathway for nitrogen compounds in salt marsh soils.

Denitrification in western riparian wetlands is highly influenced by the presence of flooding and waterlogging of soils, thus creating aerobic and anaerobic zones conducive to the denitrification processes described above. The riparian zones of the South Platte are not subject to extensive flooding and waterlogging due to the controlled flow of the river from irrigation diversions. Most of the riparian zones on the flood plain of the South Platte River are populated with perennial grasses and used for pasture.

Plant uptake of nitrogen is based on the nutrient needs of the plant. Unless the wetland vegetation is harvested and removed from the wetland system, the growth and senescence of plants keep organic plant materials in the wetland system, so that organic matter becomes part of the nitrogen cycles. The harvesting of wetland vegetation and management of South Platte River wetlands is not a part of this research.

MATERIALS AND METHODS

INTRODUCTION

The research problem, as stated in the goals and objective on page 2 is *to identify methodologies to evaluate the potential of wetlands as a treatment device for reduction of nitrate non-point source pollution effects.*

The methodology is divided into two sections: 1) development and acquisition of mapping data and tabular data for building the necessary GIS and the ground water aquifer model; and 2) running the ground water model to identify the service areas of the selected wetland for evaluation of the wetland modeling technique within and without the GIS. Figure 4 illustrates the sources of mapping data and the generation of the map layers for the GIS. Figure 5 illustrates the steps involved in running the wetland ground water model and the GIS.

DATA ACQUISITION AND BUILDING THE GIS

Spatial data layers needed for a development of a regional GIS/ground water modeling system include:

1) physical aquifer dimensions including lateral extent, bottom confining layer, important boundary conditions such as lakes and rivers, and some estimate of the existing water table location;
2) wetland locations;
3) stresses to the aquifer, mainly recharge and pumping, or a means of estimating these stresses from other information; and
4) aquifer hydraulic conductivity or transmissivity.

General aquifer dimensions and properties for the study area were obtained primarily from a series of reports published by the U.S. Geological Survey (Hurr *et al.*, 1972a; Hurr *et al.*, 1972b). Aquifer boundaries, contours of the bedrock surface, transmissivity of aquifer materials, and contours of the water table surface were digitized from these reports. The raster surfaces corresponding to these digitized contours were interpolated using a spline with tension interpolator in GRASS (U.S. Army Corps of Engineers, 1993; Mitsova and Mitas, 1994; Mitsova and Hofierka, 1994). The availability of the prepared maps of aquifer properties and dimensions for this area greatly simplified the process of aquifer definition for the ground water simulation. However, aquifer dimensions and estimations of aquifer properties can often be derived from well logs and pumping yield information which is usually available from the responsible state agency (for Colorado this is the Office of the State Engineer). Figure 6 is a three dimensional graphic of the surface topography, aquifer volumetric characterization and bedrock surface.

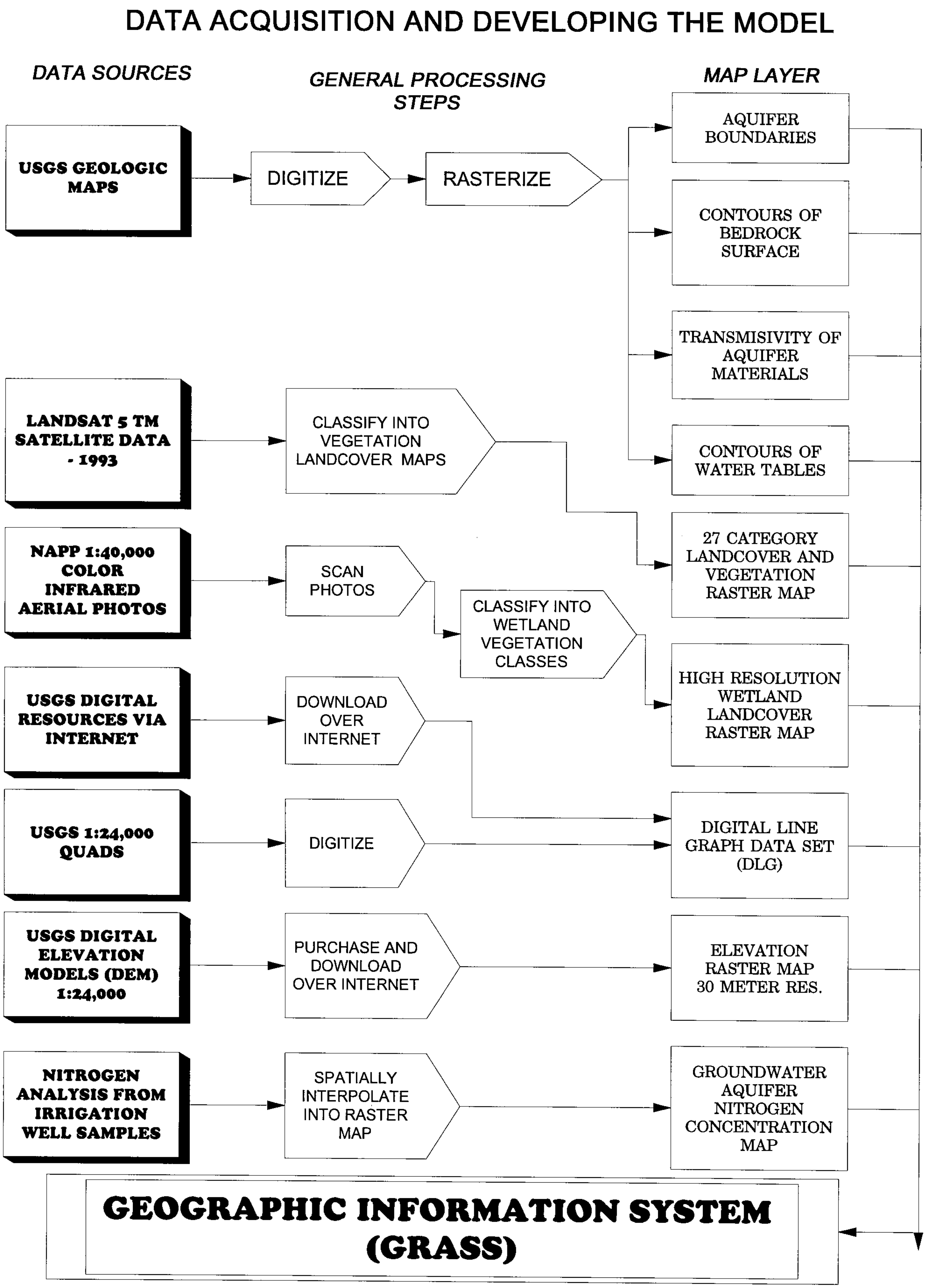


Figure 4. General procedures for generating GIS data layers

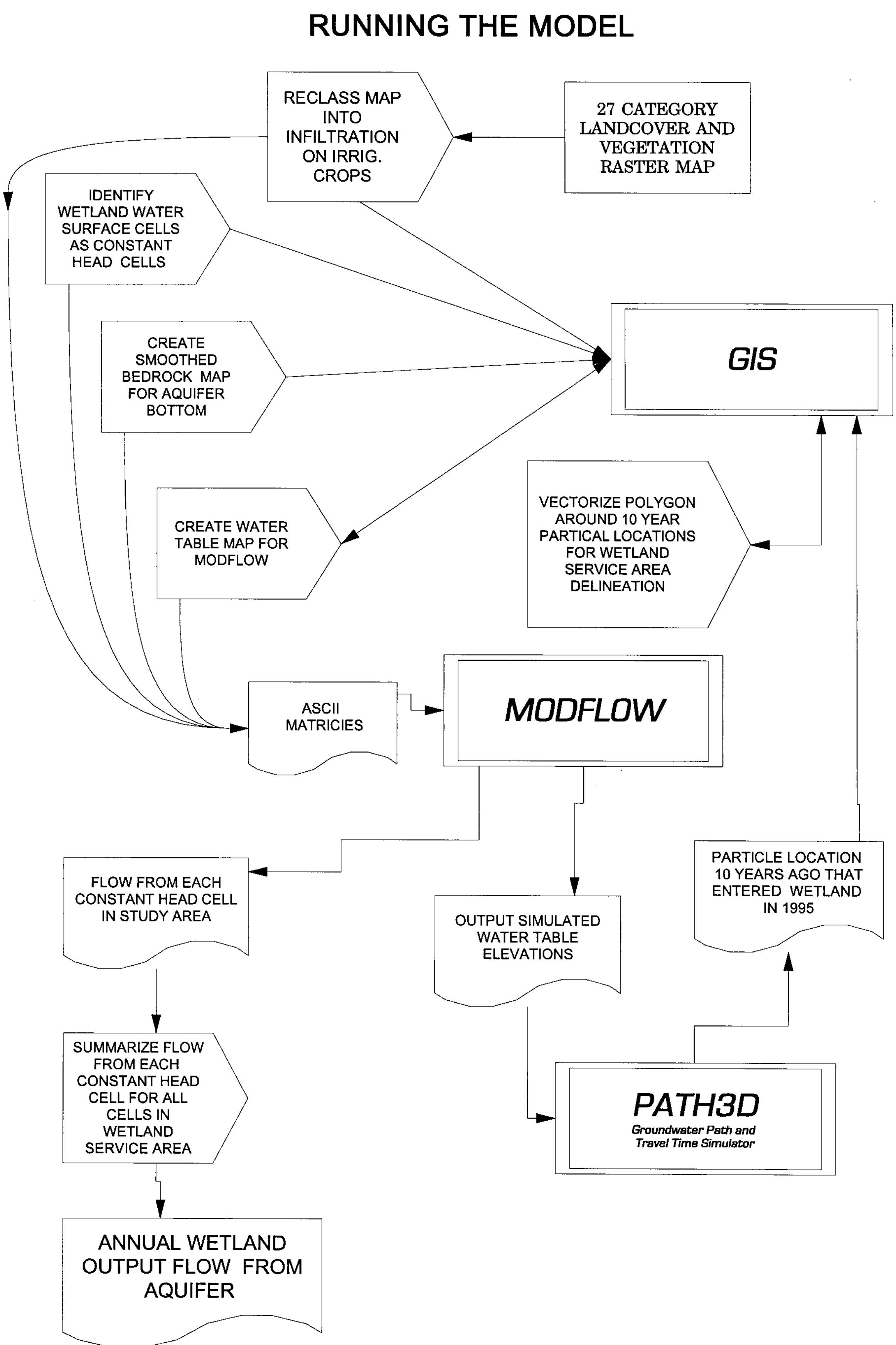


Figure 5. General procedures for running the GIS model

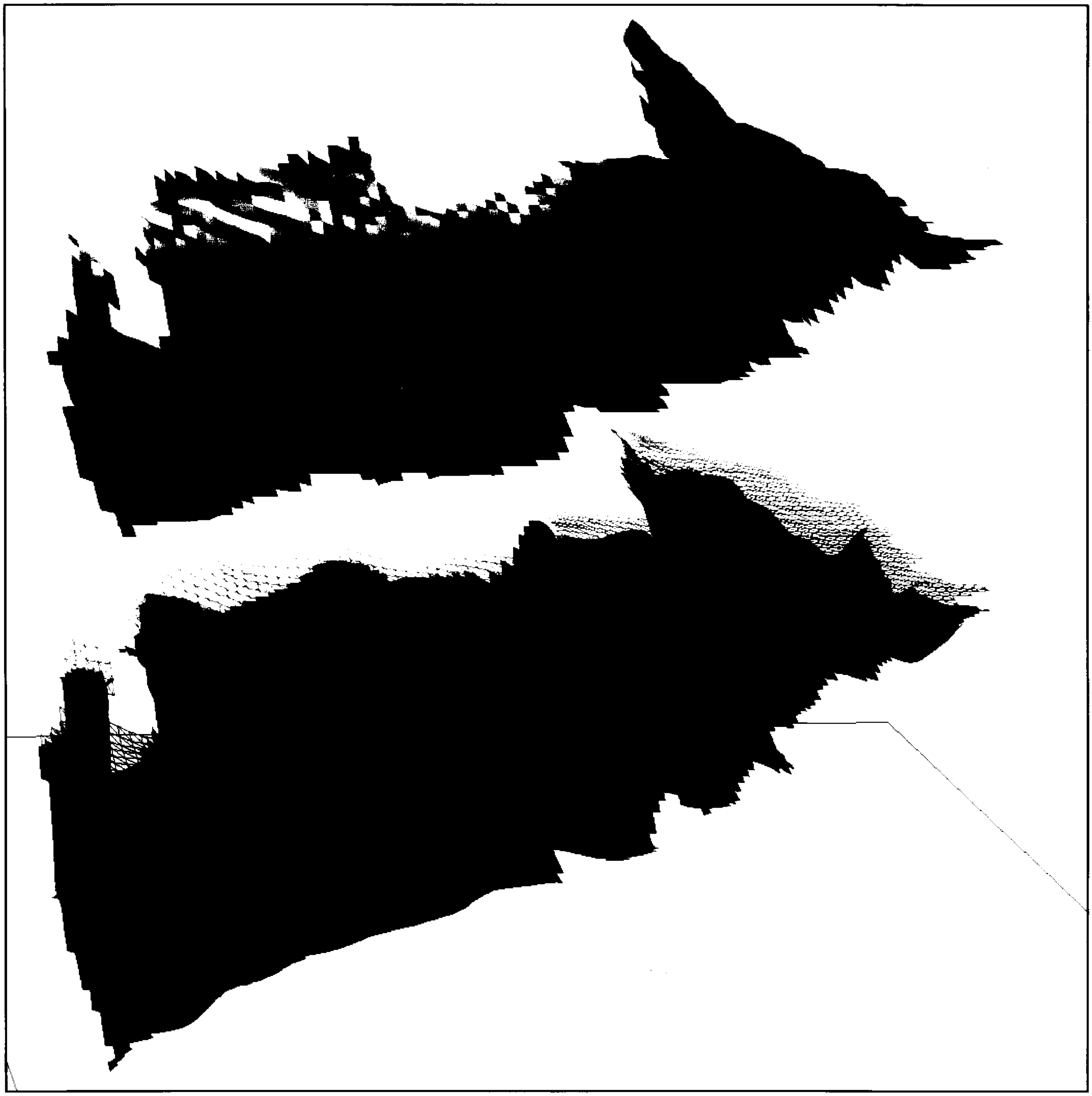


Figure 6. Three-dimensional characterization of land surface, aquifer, and bedrock.

Other GIS data layers were generated from remotely sensed data as described below. The location of the target wetland was identified by reclassification of the vegetation coverage. The location of the river was obtained from U.S.G.S. digital line graph (dlg) files for the Milliken, LaSalle, and Platteville 7.5 minute quadrangles. The line (vector) representing the river location was then converted to a raster, yielding a series of raster cells along the river.

The Landsat 5 Thematic Mapper (TM) data set was acquired for May 30, July 1, and Sept. 3, 1993. The three data sets were combined into a 21-band multitemporal-multispectral data set. The complete scene was subset into a section covering the south half of Weld County and the subset was computer classified using unsupervised classification techniques to generated a 27 class land cover-land use map (Wagner, 1994). The study site, Figure 7, shows the land cover-land use map extracted from the South Weld County classified vegetation map.

USGS National Aerial Photography Program (NAPP) Color IR transparencies (9"x9") of the study area, at 1:40,000 scale, were acquired and videorasterized for the wetland site. The resulting three-band (RGB) data set of the wetland, shown in Figure 8, was combined with a subset of Landsat 5 Thematic Mapper (TM) multitemporal data sets of the study area for a 24 band multispectral data set. The 24 band data set was computer classified into 27 categories including water, riparian and perennial grasses. These three major vegetation types are common in the wetland area. Figures 9 and 10 show photos of the wetland site indicating the cattails and open water zones, the perennial grasses and bare ground of the approximately 160 ha (400 acres) wetland site.

An alternative less timely method for locating wetlands is to use the digital or hardcopy wetland inventory maps developed by the U.S. Fish and Wildlife Service (U.S. Dept. of Interior, F&WS, 1975). Figure 11 shows the locations and types of wetlands in the study area according to this source. The wetland digital map was digitized from hardcopy wetland maps at 1:24,000 scale.

DEVELOPING THE GROUND WATER ANALYSIS PROCEDURES AND RUNNING THE MODEL

Nitrogen Loading to the Aquifer. While the target wetland may receive some surface runoff related to extreme rainfall events, most of the wetland's supply is from the inflow of ground water. The unconsolidated alluvial deposits along the South Platte River receive excess water from extensive irrigation of the overlying lands which elevates the surrounding water table and normally provides perennial flow to the wetland and the nearby South Platte. Surveys of the surrounding ground water quality have found nitrate concentrations ranging from 19 to 45 mg/L as nitrogen in ground water immediately up-gradient from the wetland (NFRWQPA, 1991). Leaching from irrigated agriculture has been identified as the primary source of nitrate to the aquifer with upstream municipal discharges, septic systems, and

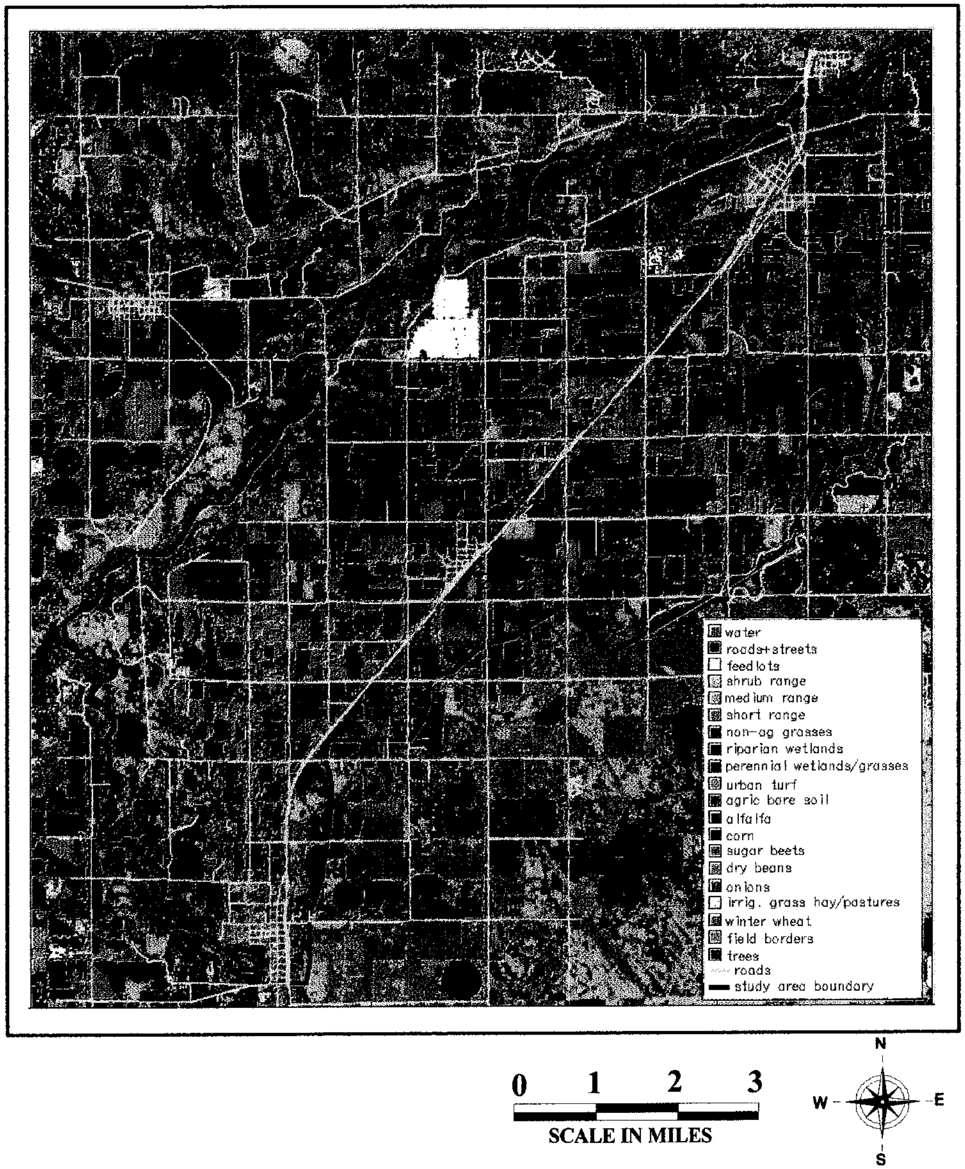


Figure 7. Landuse / landcover map derived from Landsat 5 TM satellite data set.

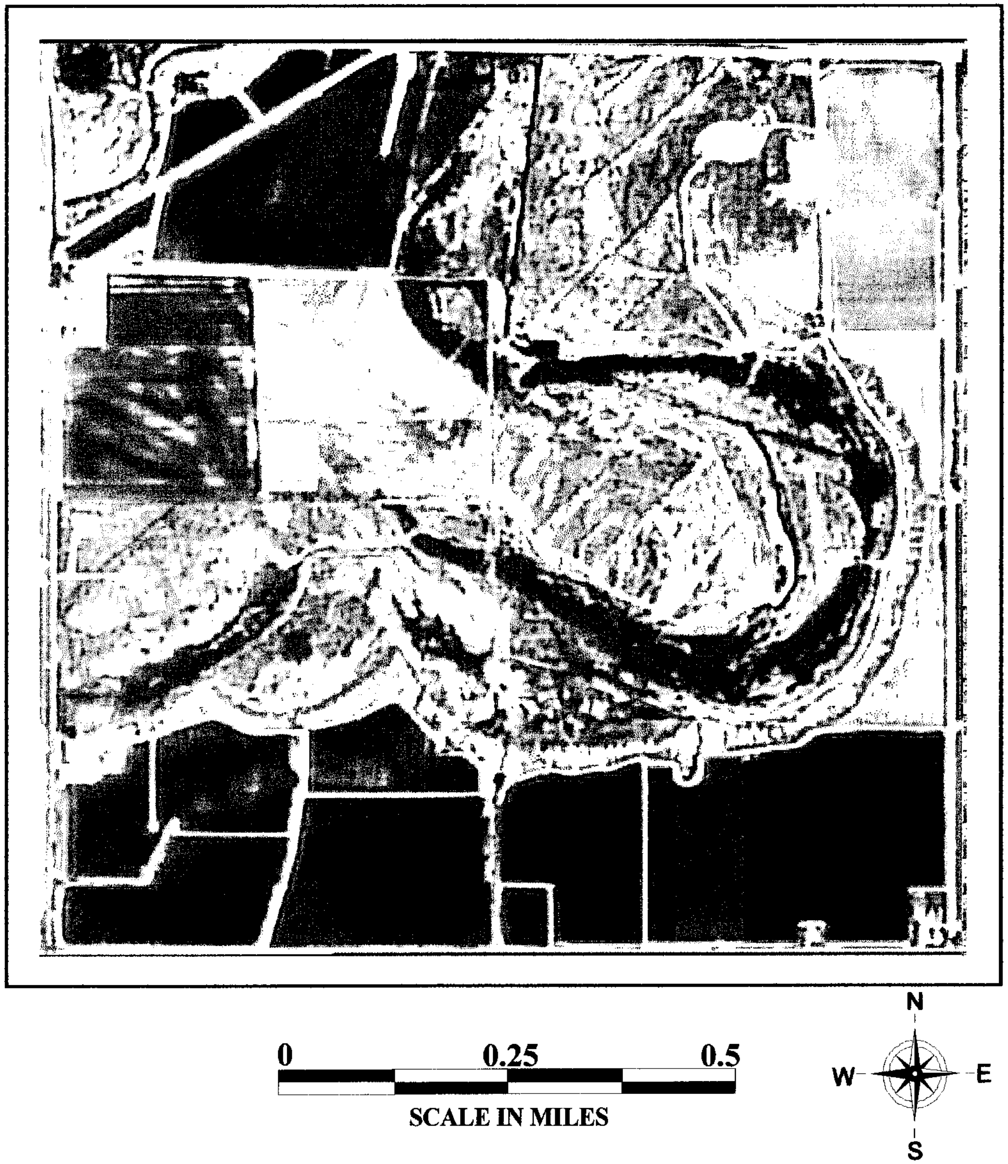
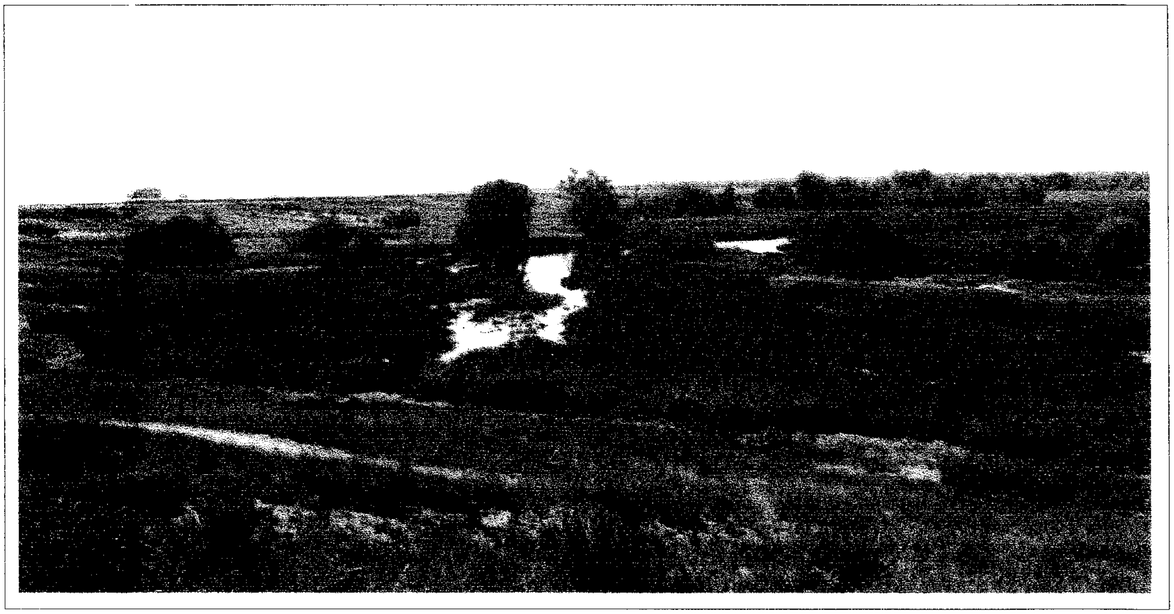


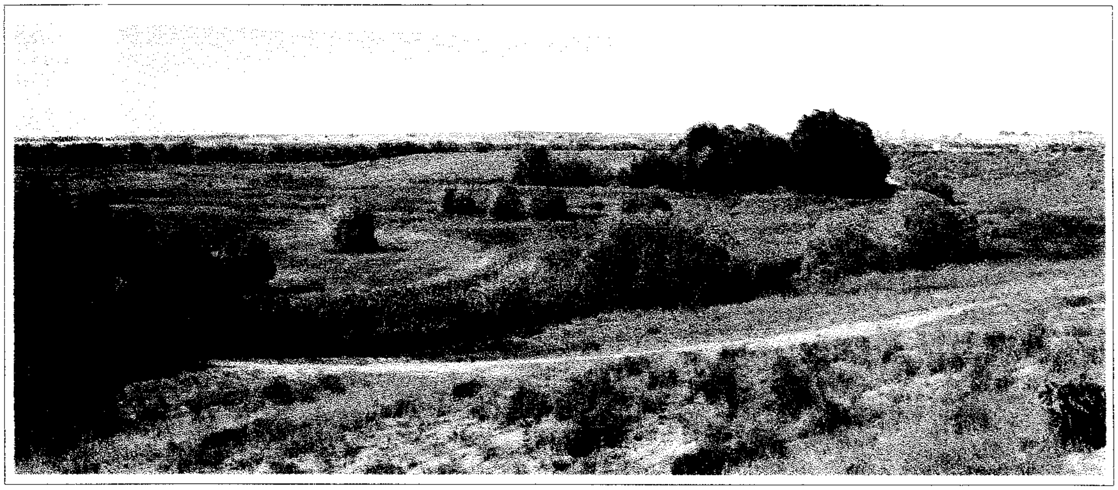
Figure 8. Wetland site extracted from NAPP aerial photograph at 1:40,000 scale.



View Looking to Northwest

View Looking to North

Figure 9. Wetland Study Area.



View of Wetland Study Area Looking North

Open Water Area in Center of Wetland

Figure 10. Wetland Study Area

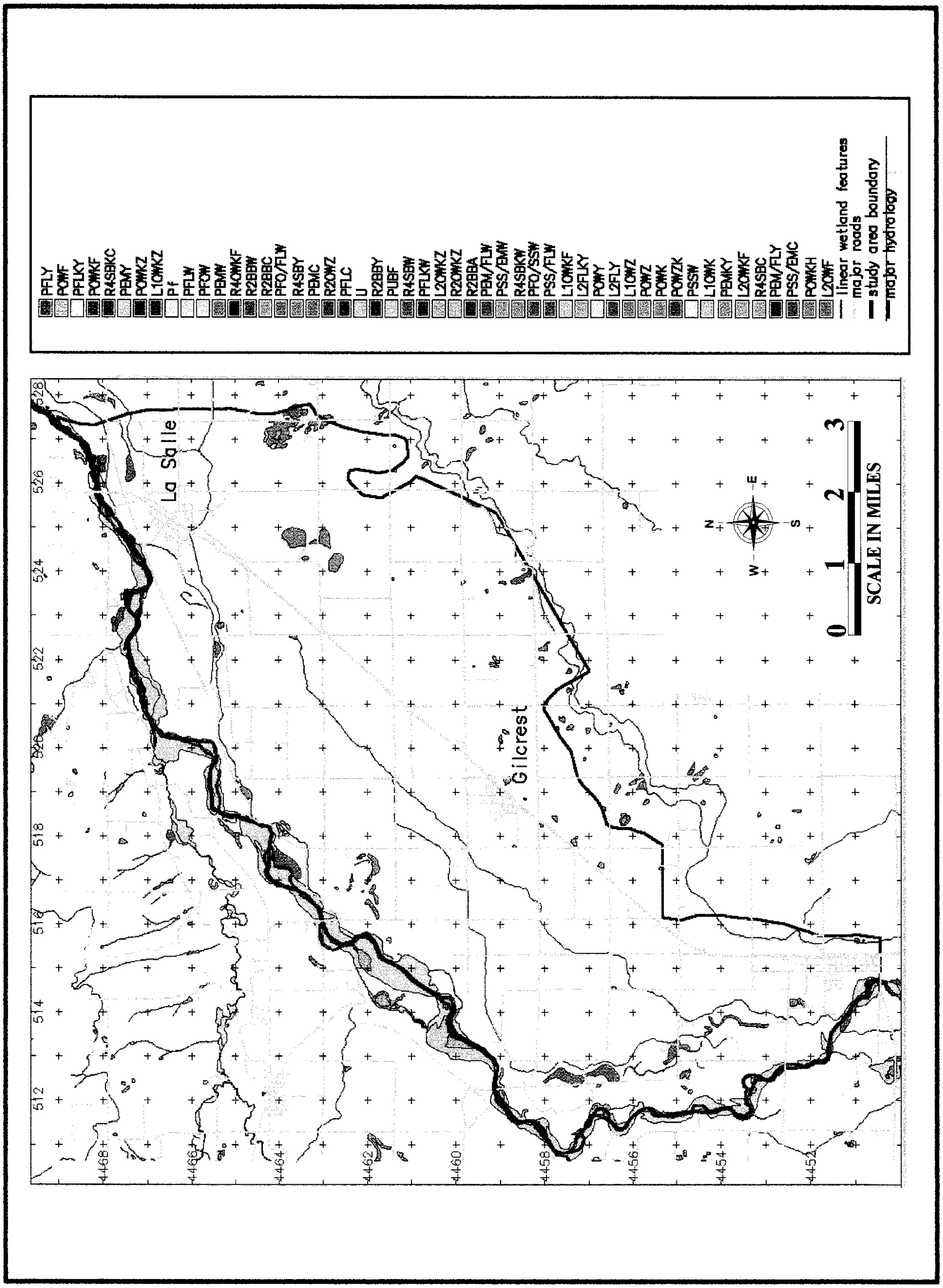


Figure 11. South Platte wetlands from National Wetlands Inventory maps.

other sources making less significant contributions (McMahon *et al.*, 1993; Wylie *et al.*, 1994).

Efforts have been made to estimate the rate of nitrate leaching to the ground water under various conditions. The Northern Colorado Water Conservancy District has closely monitored numerous irrigated fields in the region since 1989 as part of their Irrigation Management Program (NCWCD, 1989-1994). These field studies estimated nitrate leaching rates of 0 to 180 kg/ha (160 lb/ac) of nitrogen, mostly on grain corn and sugar beets. The NCWCD field studies included fertilizer and irrigation practices designed to minimize nitrate leaching, and other studies have suggested much higher nitrate leaching rates. Wylie *et al.* (1994) described a methodology for using a nitrogen management simulation model in combination with a GIS to estimate leaching rates over a large area using soil characteristics and general assumptions for management practices based on farmer and extension personnel surveys.

Estimates of nitrate loading to the aquifer over any defined region could be made using detailed simulation or approximate values (or a range of approximate values) from related field reports such as the NCWCD studies. The disadvantage of the detailed simulation route is the large amount of site specific data required and the difficulties of defining "normal" management practices for a diverse agricultural area. The disadvantage of assuming an approximate value for a region is the lack of spatial variability considerations. If one's object is to develop or locate wetlands with potential to treat ground water nitrate, the lack of spatial considerations is a considerable drawback.

Aquifer Definition and Analysis For wetlands receiving their water supply from ground water, a definition and analysis of the aquifer flow is critical in determining which portions of the aquifer may be made available for potential treatment by a particular wetland. Definition of ground water flow may also be helpful in estimating the response of the wetland to changes in external stresses such as drought or land use change on surrounding uplands. Most wetland/aquifer interactions involve unconfined (water table) aquifers that receive spatially distributed recharge from the ground surface. However, special conditions may exist where wetlands are supplied from outcropping of confined aquifers. This discussion focuses on the case of unconfined aquifers.

In relatively simple cases, graphical and/or analytical methods may be sufficient to answer the questions of approximate service area of the wetland, anticipated flow into the wetland, etc. (See Demenico and Scwartz, 1990; McWhorter and Sunada, 1977; Bear, 1979). For regional analyses, where wetlands and external system stress variables are numerous and spatially variable, numerical modeling of the study area is preferred. According to the exact needs of the study and the availability of data, the level of detail required in the ground water simulation will vary. For details on the theory and practice of ground water modeling the reader is referred to a few of the many excellent references on the subject (Bear and Verruijt, 1987; Anderson and Woessner, 1992; Kinzelbach, 1986).

The use of a GIS system can greatly simplify definition of a system for ground water simulation. To date, the use of GIS in ground water modeling has been primarily for system characterization, input development, and output analysis and display. Complete integration of ground water simulation within a GIS framework has been widely discussed and somewhat less widely attempted, but such a system that is robust and readily available is yet to be developed. The most convenient combination of GIS with ground water modeling involves the use of raster-based GIS systems with finite-difference ground water models. The uniform rectangular grids employed by both technologies provide ease of data transfer and portability. Special software has been developed for specific GIS systems and groundwater models to aid in data transfer, but because of the similar grid systems, data transfer between most established systems should not be a major obstacle.

The aquifer analysis for the target wetland was carried out using the U.S.G.S. Modular Finite Difference Ground-water Flow Model (MODFLOW) (McDonald and Harbaugh, 1988) and the raster-based GRASS GIS system (U.S. Army Corps of Engineers, 1993). A particle tracking model, PATH3D (Zheng, 1991), was also used to identify flow paths and capture zones from the MODFLOW head solutions. A wide range of software is available that would be adequate for this analysis. These particular programs were chosen based on availability, familiarity of the operators, and flexibility for use in related work. The GIS portion of the analysis was performed on UNIX workstations and the ground water modeling portions were carried out in a PC environment. Data were easily transferred by means of FTP over a local network.

Within the study area, the primary input to the aquifer is recharge from irrigated agriculture. The spatial distribution of the recharge was developed from a reclass of the vegetation coverage made from the Landsat imagery discussed earlier. All areas classified as irrigated crops were reclassed as areas receiving recharge. A single assumption of recharge rate was made based on an approximate value for irrigated corn obtained from an analysis of research reports for the region. The flow of data layers and information for the ground water analysis is shown in Figure 4 (previously described).

To avoid unnecessary complication in the process description due to local specifics, and because of limitations in the data-collection scope of this project, considerable simplification of the flow system was made for these initial, proof-of-concept runs. The system simulation was constructed with the following assumptions:

1) The river and the target wetland were simulated as constant head cells in the ground water simulation. This is equivalent to assuming that the water elevation in the river and the wetland are constant throughout the year. While the river obviously changes levels with variation in flow rate, this

variation and those in the wetland are small in relation to the regional water table gradient. With detailed data collection, variations in levels of these constant head boundaries could be incorporated.

2) Effects of pumping were ignored. This assumption certainly introduces significant error into the simulation as there are more than one hundred wells in the study area. However, in this first attempt, the object of establishing a general ground water flow regime and potential service area of the wetland could still be achieved. Inclusion of pumping effects would be a good first step at improving the simulation, and this modification will be included in future work. However, pumping is not closely controlled in this region, and reasonable estimates of pumping rates are very difficult to obtain.

3) Other secondary sources of recharge, such as natural precipitation and leakage from ditches and small watering ponds, were considered negligible. This, again, is a significant assumption, and consideration of these sources might improve simulation accuracy.

4) The hydraulic conductivity of the aquifer was assumed constant. The head approximations obtained under this assumption were reasonable for most of the study area, but some obvious problem areas on the margins of the study area need more attention. As a refinement of ground water simulation, spacial variability in hydraulic conductivity may be introduced during calibration procedures in the future.

5) The system was simulated as steady state flow. It was assumed that the recharge takes place steadily over the entire calendar year rather than concentrated in the growing season. Once again, considerable improvement in simulation accuracy might be obtained by consideration of seasonal cycles.

Spatial distributions of the following were exported from the GIS as ASCII files:

- active area (0/1 map of the study area to define spatial extent of simulation)
- constant head cells (river and wetland)
- observed water table elevations (used as starting approximations for the flow simulation)
- bedrock elevations (used to define the aquifer bottom)
- recharge areas (active recharge areas are given non-zero values corresponding to recharge rates)
- aquifer hydraulic conductivity (in this case a matrix of constant values)

The above listed files were in the form of rows of values corresponding to the rows of the finite difference modeling grid. In the case of MODFLOW, these files remain as separate files to be called by the main program. With other flow models, the matrices of values may have to be incorporated into a main input file. While most of the initial GIS analysis was carried out at the resolution of the Landsat imagery (28.5m x 28.5 m), the ground water simulations were conducted at 285m x 285m resolution to reduce simulation run time, simplify file transfer and inspection, and allow cell by cell interrogation of the data. The other non-spatial input files for the MODFLOW simulation were prepared according to the chosen assumptions, model grid dimensions, output requirements, and solution methods.

The flow model output consisted of a predicted head (water table elevation) distribution and/or drawdown (difference between simulated steady-state and starting water table elevations). This output was then be transferred back to the GIS for display and analysis.

For definition of the wetland service area, the particle tracking model, PATH3D was used (Zheng, 1991). This model used head distribution output from MODFLOW to estimate flow velocities of imaginary fluid particles placed at strategic locations within the flow simulation grid. For the steady-state simulation, PATH3D is capable of reverse tracking which allows placement of particles at the location of the wetland cells and tracking them back through time to the point where they enter the flow field. This allows definition of the portion of the aquifer which will contribute to the wetland inflow for a given period of years. Similar information can be obtained by locating numerous particles throughout the grid and identifying the location through time of those particles exiting the wetland.

RESULTS AND DISCUSSION

RESULTS

The study area, shown in Figure 1, covers 13,761 ha (34,005 acres). Table 1 provides areas for each of the vegetation and land cover categories in the study area. Table 2, a condensation of Table 1, shows only the area of irrigated crops within the study area.

The wetland capture zone (service area shown in Figure 12) encompasses approximately 712 hectares (1759 acres). The capture zone is defined as the portion of the area contributing recharge that will flow through the wetland within 10 years. The boundary of the capture zone is the origin of water reaching the wetland after 10 years of travel. Table 3 lists the vegetation types and areas lying within the capture zone.

Table 4 shows the ground water nitrate-N concentrations taken from irrigation wells tapping the aquifer surrounding the wetland (NFRWQPA, 1991). Figure 13 shows the location of the sampling wells.

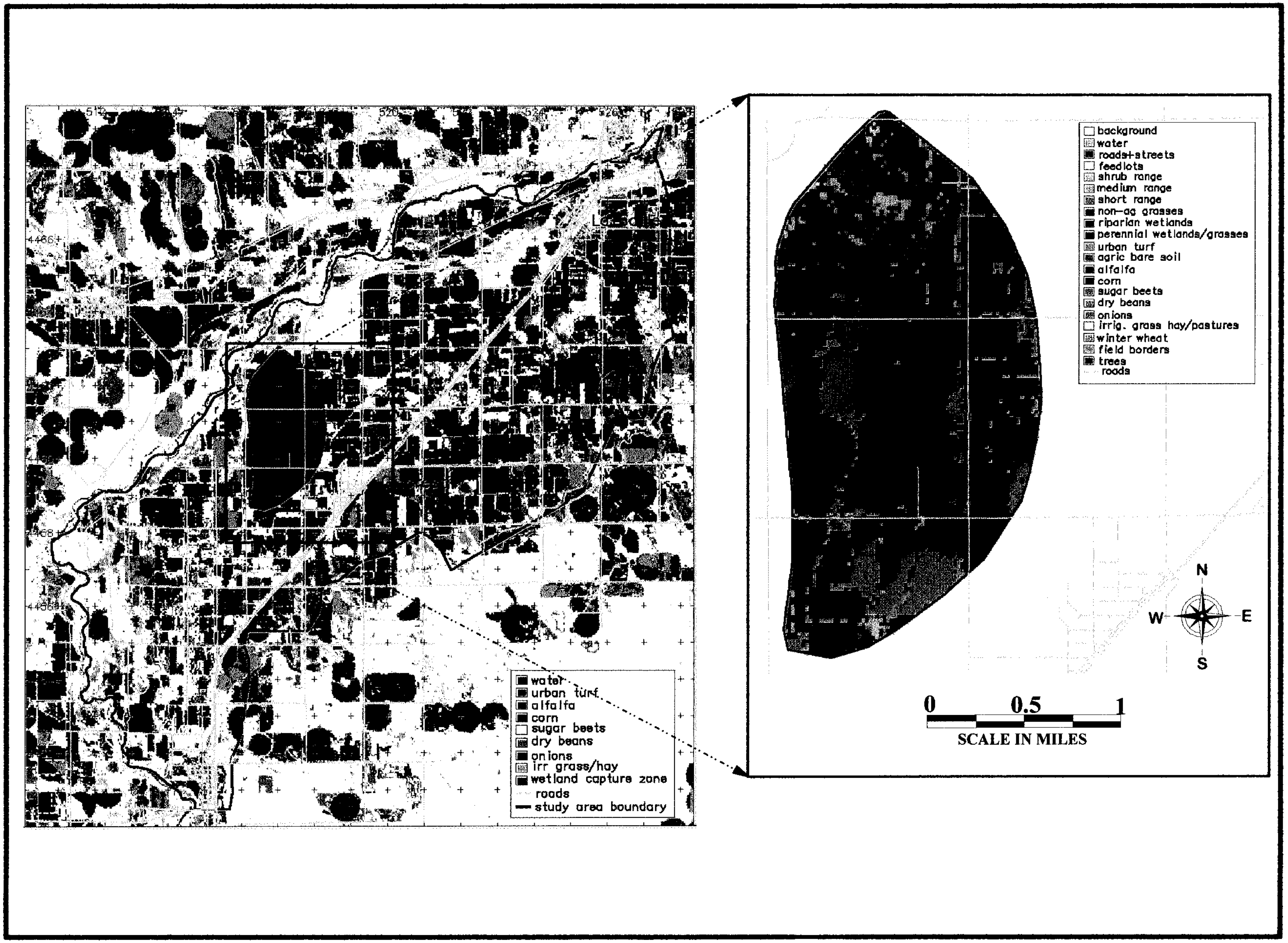


Figure 12. Simulated zone of recharge captured by wetland within 10 years.

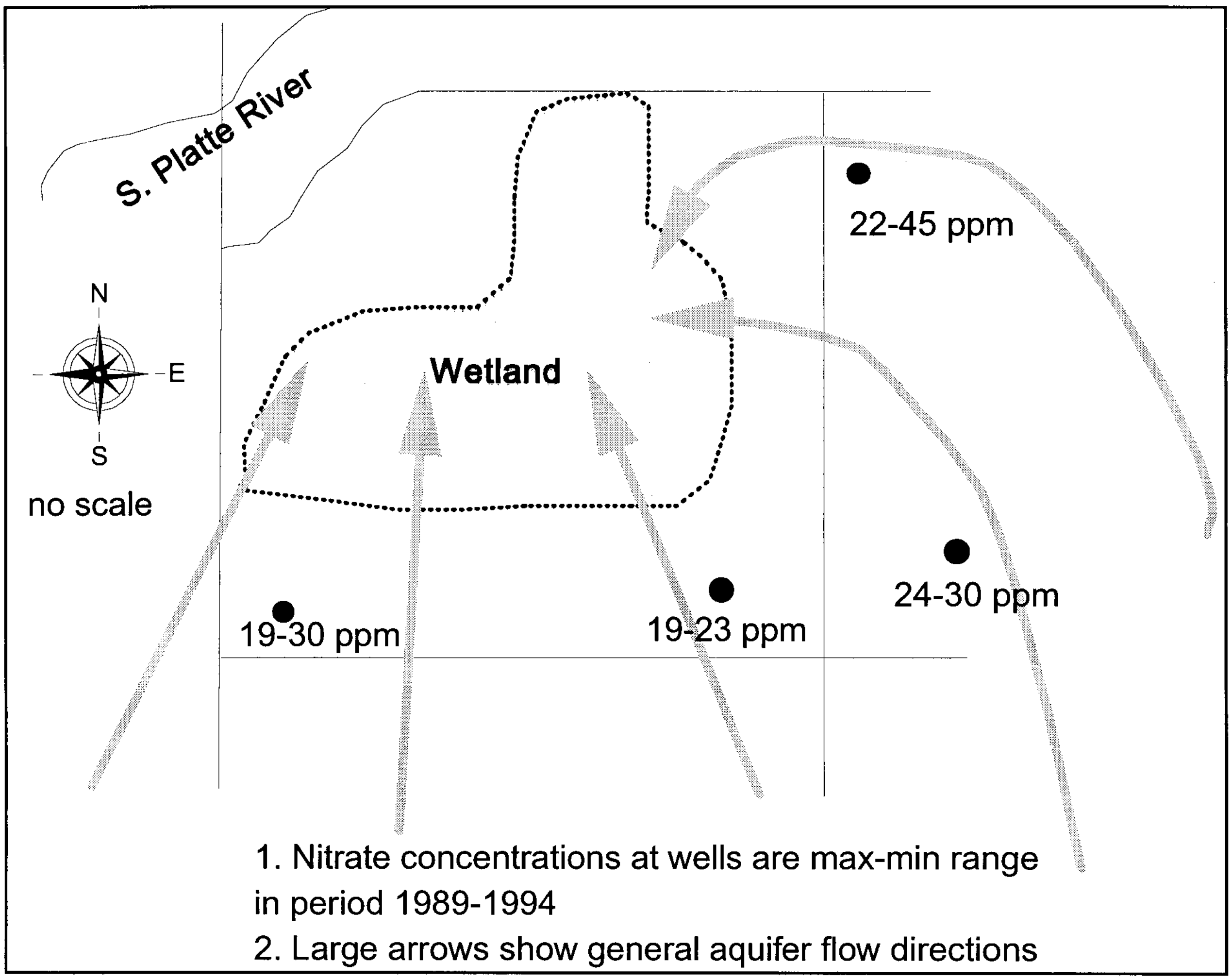


Figure No 13. Ground water nitrate concentrations surrounding wetland.

Table 1. Study area land use/land cover category areas.

Class	Description	hectares	acres
1	water	54	133
2	roads and streets	251	620
3	feedlots	177	439
4	shrub range	323	797
5	medium range	196	485
6	short range	330	816
8	non-agricultural grasses	1,356	3,353
10	riparian wetlands	279	692
11	perennial wetlands/grasses	518	1,281
12	urban turf/grass	128	318
14	agricultural bare soil/fallow	264	652
15	alfalfa	2,036	5,031
16	corn	4,939	12,203
17	sugar beets	340	841
18	pinto beans	399	985
19	onions	57	142
20	irrigated grass/hay/pasture	754	1,863
22	winter wheat	59	146
23	field borders	990	2,446
27	trees	308	760
Total		13761	34005

Table 2. Irrigated Crops in the study area.

Description	hectares	acres
corn	4,939	12,203
alfalfa	2,036	5,031
sugar beets	340	841
dry beans	399	985
onions	57	142
irrigated grass/hay/pasture	754	1,862
urban turf	129	318
Total	8,654	21,384

Table 3. Vegetation/land use in wetland capture zone.

Class	Description	hectares	acres
1	water	4.5	11.2
2	roads and streets	4.4	11.0
3	feedlots	.2	.2
4	shrub range	3.6	9.0
5	medium range	3.6	9.0
6	short range	7.7	19.0
8	non-agricultural grasses	109.0	269.3
10	riparian wetlands	35.9	88.8
11	perennial wetlands/grasses	24.8	61.2
12	urban turf/grass	.6	1.6
14	agricultural bare soil/fallow	.9	2.2
15	alfalfa	128.9	318.5
16	corn	294.2	727.0
17	sugar beets	28.6	70.6
18	pinto beans	.9	2.2
20	irrigated grass/hay/pasture	12.9	32.0
22	winter wheat	18.0	44.6
23	field borders	17.3	42.8
27	trees	15.6	38.6
Total		711.7	1,758.6

Table 4. Ground water nitrate concentrations in study area.

concentration of nitrate-N	percent coverage	hectares	acres
00 - 10 ppm	22	3,048	7,533
11 - 20 ppm	42	5,763	14,240
21 - 30 ppm	29	4,021	9,935
> 30 ppm	7	920	2,274
Total	100	13,752	33,982

Evapotranspiration Losses from Wetlands. Wagner (1994) estimated annual evapotranspiration rates of 71 cm (28 inches) for perennial wetlands and wetland grasses and 107 cm (42 inches) for riparian wetlands in the nearby Cache la Poudre River basin using the Kimberly-Penman method. Using these estimates for the reported acreages for the study area (518 and 280 ha, respectively for perennial and riparian wetlands -- table 2), the total loss of water from the study area wetlands via evapotranspiration is 6.6 million cubic meters (5,400 acre-ft).

Nitrogen Loading to the Aquifer. Assuming an average annual nitrate leaching rate of 100 kg/ha (89.2 lb/acre) for all irrigated crops in the Gilcrest study area (cataloged in Table 2), the total nitrogen loading rate for the study area is 865,000 kg (as nitrogen) (953 tons) from agricultural sources. A water leaching rate of 50 cm/yr (25 cm evapotranspiration, 33% efficiency of water applied) for the same irrigated acreage represents 43 million cubic meters (35,000 acre-ft) of water. This leaching quantity gives an estimated average nitrate concentration of 20 mg/l in the leachate. Similarly calculated loading rates for the wetland service area are 71,200 kg/yr of nitrogen in 2.2 million cubic meters (1,800 ac-ft) of water.

Water Quality Sampling Studies. In this study, a single grab sample was taken in late 1994 from surface waters at the center of the wetland and at the wetland outlet where it passes under County Road 46. Another grab sample was taken in May, 1995. The nitrate concentration in the surface waters at the center of the wetland was 8.57 ppm, at the outlet in 1994 - 13.4 ppm and at the outlet in May 1995 - 13.5 ppm.

DISCUSSION

Remotely sensed satellite imagery at a ground resolution of 30 meters was sufficient to develop GIS data layers for land use and land cover maps within the study area. Discrimination of the vegetation and open water coverages within the wetland required higher resolution data sets. The rasterization of NAPP aerial photography yielded 7.12 meter resolution. Classification of the high resolution wetland digital data sets provided an estimate of the open water areas and the amount of riparian cattails and perennial grasses.

The amount of water sampling for nitrogen analysis was insufficient to statistically correlate the irrigation water-leached nitrates from the capture zone and the removal of nitrates by the various natural processes within the wetland. However, there is a definite trend indicating a decrease in nitrate concentration between the ground water and the wetland outflow into the South Platte River. This trend suggests some level of nitrate treatment occurring in the wetland.

The use of the particle tracking program provided an estimate of the wetland capture zone for tracking the leached nitrates from irrigated crops within the capture zone.

The interpolated map of ground water concentrations (Figure 2) shows that, while estimated 20 mg/l concentration of the leachate is a realistic average concentration for the study area, significant portions of the area are both above and below that value (64% below, 36% above). This demonstrates the drawback of one general water and nitrate loading rate for a large area study. As an extension of the research of which this project is a part, attempts are being made to estimate spatially variable leaching rates for the study area based on variations in management practices and controlling physical parameters. It is important to note here that other factors such as denitrification and phreatophytic uptake may, under appropriate conditions, affect the aquifer nitrate balance.

The use of the generalized ground water flow and particle tracking model allowed estimation of the approximate region from which the wetland receives its water -- and nitrates. While the simplifying assumptions made for this study certainly introduced considerable error, reasonable agreement was obtained between simulated and observed heads over most of the study area. Incorporation of well pumping and temporally variable recharge in the ground water simulations is a reasonable extension of the process presented here and should allow estimation of nitrogen and water delivery rates to the wetlands through time, allowing consideration of more detailed conditions such as annual cycles in wetland treatment capacity.

For comparison to the results from this study, another researcher (Tibbetts, 1994) reported nitrogen concentrations in a prairie wetland north of Fort Collins in the Meadow Springs Ranch at 0.2 mg/L with alkalinity levels of 190 mg/L. Hardness (Ca) was 193 mg/L and Mg hardness was 37 mg/L. Iron levels were reported at less than 0.1 mg/L.

SUMMARY AND CONCLUSIONS

The procedures described above provide a workable methodology to determine the potential removal of nitrates from groundwater flowing into a wetland. Wetlands are potential biological treatment facilities for denitrifying and uptaking nitrates from aquifer water and from agricultural return flows due to irrigation.

A significant conclusion resulting from this research is that a benchmark has been established for the planning of additional sampling and modeling of the wetland study site. A trend in the nitrates exiting the wetland compared to the groundwater aquifers surrounding and feeding the wetland is pronounced. A need for additional modeling and water quality sampling work is indicated using the GIS data layers and additional information for the 1995 and 1996 growing season.

The exercise described in this report demonstrates that the information and technology exist to evaluate the role of wetlands in the water quantity and water quality processes at a large spatial scale. A framework for such an evaluation and a simple case example are presented to demonstrate the variety of data sources and software support available for conducting a wetland nitrate reduction evaluation. The study attempted to expose some of the potential pitfalls and potential for additional research.

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