2.1 Basin Hydrology

This detailed assessment of phosphorus required an analysis of basin hydrology to properly evaluate the importance of the varying rainfall/runoff relationships for low, average and high flow conditions throughout the state. This section will discuss how these three flow conditions were defined and how rainfall and runoff volumes were determined for this analysis. The determination of flow conditions are especially important since they facilitate computation of nonpoint phosphorus sources and allow for the comparison of point and nonpoint phosphorus sources for the varied climatic conditions that occur across Minnesota. Following statistical analysis of the historical rainfall and runoff volumes, recent (1979-2002) water year (October 1 to September 30) data was identified to represent low, average and high flow conditions within each basin. A more detailed discussion about the approach and methodology for assessment of the basin hydrology is included in Appendix A.

2.1.1 Minnesota Basins

Figure 2-1 shows the ten major Minnesota basins considered in this analysis, along with locations of the USGS flow gaging sites used to estimate runoff during the various flow conditions. The ten major drainage basins within Minnesota vary greatly in their characteristics. Table 2-1 provides a summary of some of the characteristics of each basin. As shown in the table, there is significant variability of runoff and precipitation across the state. There is also a significant difference in land cover between basins, particularly between the southwest and northeast parts of the state.

2.1.2 Calculation of Basin Runoff Volumes

The phosphorus load estimates in this study were determined for low, average and high flow conditions, for each of the ten basins (further discussed in Appendix K). The phosphorus load estimates for each flow condition are based on the annual runoff volumes that have been determined from recent water year flow data. A characteristic of most of the basins is that water is received from upstream basins (such as the Lower Mississippi which receives flow from the Minnesota, St. Croix and Upper Mississippi basins) or water flows into the basin from neighboring states or provinces (Minnesota and Rainy River basins). The Upper Mississippi River is the only basin in the state that is a headwater basin (wholly within Minnesota). Therefore, flow and phosphorus data measured at the "outlet" or mouth of the basin will include both water and phosphorus originating from outside of Minnesota or from other upstream Minnesota basins. For example, 53 percent of the watershed area of the Red River of the North (which is the border between North Dakota and Minnesota), at the

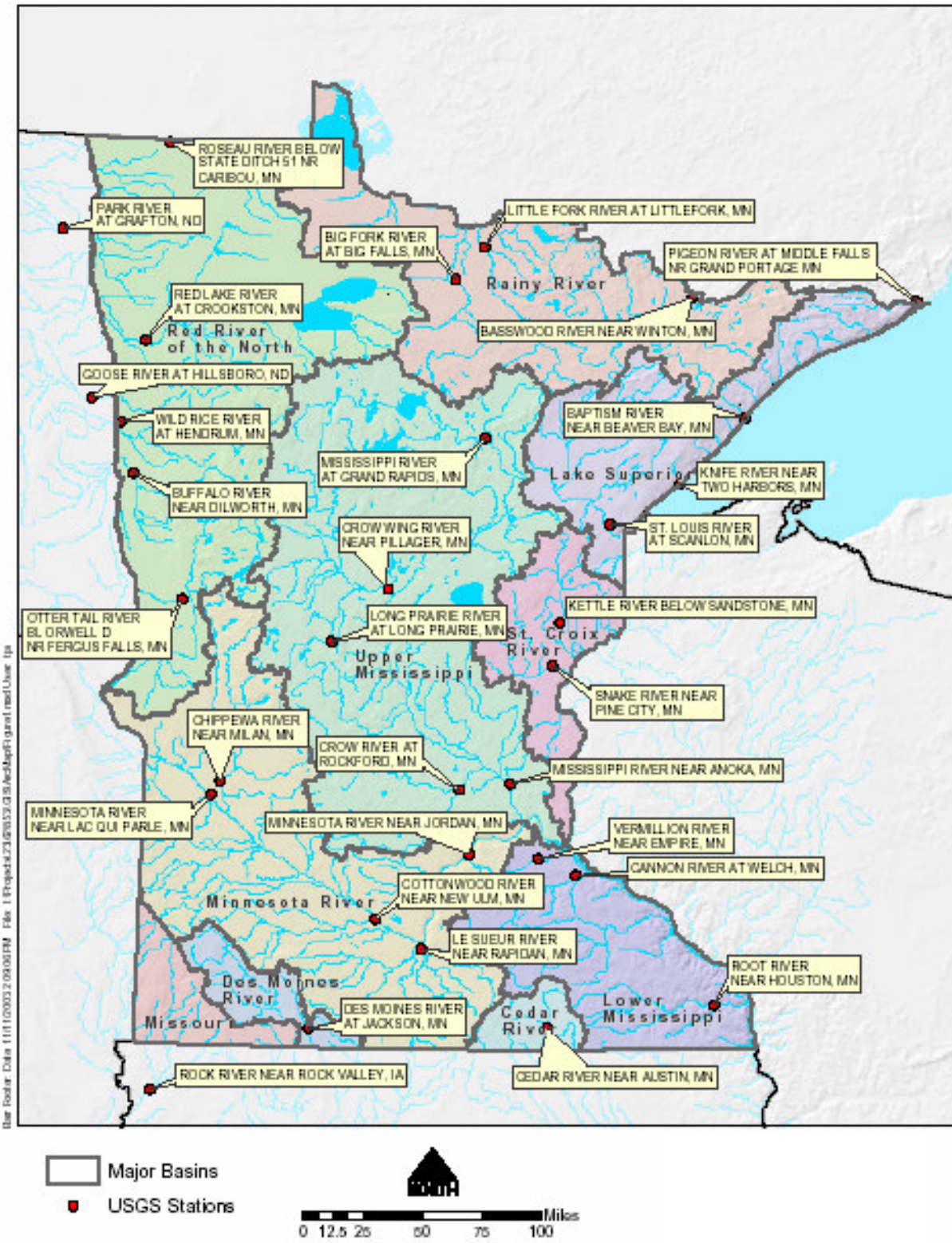**Figure 2-1   Major Basins with USGS Flow Gaging Stations**

Table 2-1   Basin Characteristics

Basin	Area (Sq Miles)*	Average Precipitation (1979-2002)	Average Runoff (1979-2002)	Land Cover Percentages**					
				Urban	Forested	Tilled Agricultural	Pasture/ Grassland	Wetland/Open Water	Other
Cedar River	1,028	32.06	9.80	3.4%	3.3%	83.4%	6.2%	3.7%	0.0%
Des Moines River	1,535	27.98	5.68	1.8%	1.8%	79.9%	11.0%	5.5%	0.0%
Lake Superior	6,149	29.11	12.44	1.4%	57.1%	2.6%	3.5%	33.3%	2.1%
Lower Mississippi	6,317	33.29	10.28	2.4%	15.4%	52.2%	24.8%	5.1%	0.1%
Minnesota River	14,943	28.14	5.61	2.2%	4.6%	72.7%	12.6%	7.8%	0.1%
Missouri	1,782	27.16	5.25	1.5%	1.0%	78.9%	16.0%	2.6%	0.0%
Rainy River	11,236	26.20	8.01	0.4%	41.4%	2.0%	2.3%	52.5%	1.3%
Red River	17,741	23.29	3.42	0.7%	12.0%	54.6%	8.8%	23.8%	0.2%
St. Croix River	3,528	30.61	9.71	1.3%	36.8%	10.8%	20.6%	30.1%	0.2%
Upper Mississippi	20,100	28.07	6.87	3.5%	29.1%	20.2%	16.7%	29.7%	0.7%
State Wide	**79,202**	**27.39**	**6.83**	**1.9%**	**22.7%**	**38.1%**	**12.0%**	**24.7%**	**0.6%**

*Drainage area within Minnesota
**Based on USGS National Land Cover Database (1992)

Manitoba border, is in the State of North Dakota. Since this study is only concerned with phosphorus contributions from Minnesota, a methodology was developed to estimate only Minnesota's contribution of water. Runoff from the Minnesota portions of the ten basins were calculated using state-wide flow maps for the three flow conditions. Each map, developed using ESRI ArcView software, consists of a state-wide 1 km x km grid of values representing runoff in inches. Using these grids, runoff averages over the basins were determined. The methods used to develop these maps are described below.

2.1.2.1   River Discharge Data

Monthly mean stream flow data were collected from the United States Geologic Survey for 27 gaging stations in Minnesota, two in North Dakota and one in Iowa for a total of 30 gages. The stations were selected based on their length of record and the location of the gage within each of the ten basins. Annual runoff in inches, for each gage was determined by summing the monthly mean flows for each water year (October 1 – September 30) and dividing by the contributing watershed area to arrive at runoff in inches per year. The watershed areas were delineated using the Minnesota Department of Natural Resources Division of Waters Watershed Basin (1995) GIS Layer. This layer was developed using data from USGS 1:24,000 Quadrangle Maps.

2.1.2.2   Precipitation Data

Basin-wide precipitation data were made available from the State Climatology Office of the Minnesota Department of Natural Resources. The data consisted of monthly values calculated from a grid-based archive of historical monthly precipitation totals for the period of 1892 – 2002. These

data consisted of estimated monthly total precipitation over each watershed, in inches, for each of the ten basins. Data for the period of 1979 – 2002 water years were used in this study.

2.1.2.3   Runoff Frequency Curves

The result of the basin runoff computations was a table of annual runoff values, in inches over each of the 30 watersheds. These data were used to develop two frequency curves for each of the 30 gages and were based on these following periods of record:

- Using all water years data were available
- Using water years 1979 – 2002

For curve one, the time period of available flow data varied greatly. Some gages had data available for up to 100 years and others only a dozen or so years.   The second curve was developed to reflect current climatic and drainage conditions. For the period from 1979 to 2002, a complete record of data was available for most of the gages used. Since this period reflected current watershed drainage characteristics and climatic trends, the 1979-2002 record was used to develop the runoff mapping.

The frequency curves were developed using a statistical analysis of the annual basin flows adopted from *Guidelines for Determining Flood Flow Frequency*, Bulletin #17B, U.S. Water Resources Council, Sept. 1981. The Weibull plotting position method, described in this reference, was implemented to assign an exceedence probability (the probability of the flow being greater than or equal to a value) to every annual flow record in the time series. The probabilities were then plotted on semi-log paper to fit a trend line to the data. Different statistical equations were analyzed to determine which equation best describes the data. The frequency curves were then based on the best-fit equation, typically a Pearson Type III distribution.

From the frequency curves developed for the 1979-2002 water year period, runoff values from the 90 (dry year), 50 (average year) and 10 (wet year) percent probability were determined. The 90 percent value means that, on average, 90 percent of the years will have runoff exceeding this value. The 50 percent value shows the runoff amount that would be exceeded during one-half of the years, on average. The 10 percent value is the flow which would be exceeded during only 10 percent of the years. The 90 and 10 percent probabilities were the respective probabilities selected to represent low and high flow conditions, because they do not represent extreme events; rather they represent typical dry and wet periods for the basins (a 1 in 10 chance of occurring on any given year), respectively.

2.1.2.4 Precipitation Frequency Curves

Frequency curves were also developed for the basin-wide precipitation data. The data were summarized by water year and the same methodology used to develop the flow – frequency curves was utilized for the precipitation data.

2.1.2.5 Runoff Maps

The centroid (or center of the watershed) for each of the 30 USGS gaged watersheds was determined. The resulting X and Y coordinates of the centroid (in UTM Coordinates) were determined and were assigned the runoff values for the watershed. A table was constructed with the UTM coordinates and runoff values. This table was imported into Surfer Software and interpolated using the Kriging routine to create three state-wide 1 kilometer x 1 kilometer grids representing the dry, average and wet condition runoff values. The resulting Surfer grid files were imported into ArcView Spatial Analyst extension and were overlain with the boundaries of the major basins to provide an estimation of the wet, average and dry condition flow volumes based on the 10, 50 and 90 percentile frequencies, respectively.

It is important to note that, in general, the year in which the 10th percentile wet year flow volume occurred does not necessarily coincide with the year in which the 10th percentile wet year precipitation amount was observed. River discharge is not only a function of precipitation, but is affected by a number of hydrologic conditions such as drought and floods occurring in preceding years. For example, if the preceding year was much dryer than normal, much of the current year's rainfall (even though above average) may be used in refilling lake and wetland basins and replenishing soil moisture. The intensity of rainfall is another factor in the generation of runoff. For a given amount of precipitation, more of it will run off if the precipitation occurs during a heavy thunderstorms rather than rain falling during a gentle day-long shower. Therefore, there may be below-normal flow in years where precipitation is above-average. In this study it was assumed that the 10th percentile flow does occur in the same year that the 10th percentile rainfall occurs. The same assumption was made for the 50th and 90th percentile years. This simplifying assumption had to be made to facilitate a direct comparison between the three flow scenarios examined.

2.2 Phosphorus Sources to POTWs and Minnesota Surface Waters

As discussed in Section 1.2, the requirement to study the concept of lowering phosphorus in the wastewater stream and the effect on water quality mandated that this assessment inventory the sources and amounts of phosphorus entering three different sizes and categories of POTWs, along with the sources and amounts of phosphorus entering Minnesota surface waters for each major basin

and for the entire state of Minnesota from point- and nonpoint-sources. Section 2.2.1 presents the methodology used to inventory the sources and amounts of phosphorus entering POTWs, by size and category, as well as estimate the amount of phosphorus entering surface waters from point sources. Section 2.2.2 provides the methodology used to assess the sources and amount of phosphorus entering surface waters from nonpoint sources. Section 2.2.3 presents the methodology used to determine the bioavailability of the point and nonpoint sources that have evaluated for this analysis. Section 2.2.4 discusses the methodology used for an assessment of effluent total phosphorus reduction efforts by wastewater treatment plants.

2.2.1   Point Sources of Phosphorus

This section provides a discussion regarding determination of point sources of phosphorus to Minnesota watersheds and the sources of phosphorus discharged to Minnesota publicly owned treatment works (POTWs). A detailed discussion about the assessment of this source category is contained in Appendix B. For the purposes of this analysis, point sources of phosphorus include domestic (public and private) and industrial facilities that discharge treated wastewater to surface water through distinct discharge points and are regulated under state and federal pollution permit programs. Wastewater is generated by a number of sources and falls into two general categories: Domestic/Residential wastewater and Industrial and Commercial wastewater. Wastewater from these two sources is discharged to one of three categories of wastewater treatment facilities (WWTFs); POTWs, privately owned wastewater treatment systems for domestic sources, and industrial wastewater treatment systems. Land disposal of wastewater does not discharge to surface waters and was not considered as part of this analysis.

POTWs include wastewater treatment facilities owned and operated by public entities (cities and sanitary districts usually). These facilities treat varying proportions of domestic wastewater and commercial/industrial wastewater. For the purposes of this study, POTWs have been subdivided into the following additional categories:

1. Size (based on Average Wet Weather Design flow)
   a. Small – less than 0.2 million gallons per day (mgd)
   b. Medium – from 0.2 mgd to 1.0 mgd
   c. Large – greater than 1.0 mgd

2. Waste Treated (% by flow volume treated)
   a. POTWs that serve mainly households and residences - less than 20 % industrial or commercial contributions

b. POTWs that have some commercial or industrial contribution – between 20% and 50% industrial or commercial contributions

c. POTWs that are dominated by a variety of commercial and industrial contributions – greater than 50% industrial or commercial contributions

Privately owned wastewater treatment systems include those designated for treatment of domestic sources and that are privately owned and operated. This category of facility is generally small and serves a limited number of residences. Mobile home parks, resorts, and small communities are examples of privately owned wastewater treatment facilities.

Wastewater generated as a byproduct of an industrial or commercial process can either be discharged to a POTW for treatment or it can be treated (if needed) on site and discharged to a surface water under its own NPDES permit. In most cases, wastewater discharged from an industrial wastewater facility is from an industrial process. This category also includes noncontact cooling water.

2.2.1.1 Data Sources for Wastewater Treatment Facilities

Identification of the point sources of phosphorus and load estimates was accomplished with existing data and literature information. No direct monitoring of waste streams was undertaken for this portion of the study. The following sources of existing data were utilized:

- Minnesota Pollution Control Agency's Delta Database

- MNPRO Database

- Metropolitan Council Environmental Services

- Minnesota Department of Health (MDH)

- Individual contact with Minnesota Communities

The MPCA maintains a database of information required by NPDES permit holders and the monitoring data required by the permit, referred to as the Delta database. Data from the years 2001, 2002 and the first half of 2003 were used in this analysis. The Delta database contained data for more than 1,300 separate permits, many with multiple discharge points called stations, and all available flow and phosphorus data contained therein was used for this study. Since many permits do not include limits and/or monitoring requirements for phosphorus, there was no phosphorus data available for some permits. As a result, it was necessary to extrapolate phosphorus data from other permit information (e.g. permit application data and basin average phosphorus for similar facilities, etc.). Discussions with MPCA staff provided a list of the water sources for most of the noncontact

cooling water dischargers in the state. Information on noncontact cooling water additives was also provided by MPCA staff.

The Minnesota Department of Trade and Economic Development maintains a database (MNPRO) that contains information regarding community profiles for each city in Minnesota. The MNPRO database was used to obtain the following information:

1. A complete listing of Minnesota communities
2. Information on the type of wastewater treatment system a community utilizes for wastewater treatment
3. Population of the community
4. A list of businesses and industries in each community, the NAICS code and number of employees for each business.

All population data obtained from the MNPRO database were from 2001 estimates. The other data obtained from the MNPRO database were provided by the communities and there may be some variation regarding the dates this information was reported.

The Metropolitan Council Environmental Services (MCES) owns and operates the eight Twin Cities Metropolitan Area wastewater treatment facilities. The Industrial Waste & Pollution Prevention (IWPP) Section, located within MCES's Environmental Planning and Evaluation Department, regulates and monitors industrial discharges to the sewer system to ensure compliance with local and federal regulations. IWPP Section staff issue Industrial Discharge Permits to industrial users of the Metropolitan Disposal System. For each MCES industrial permit holder, MCES provided the following information:

1. Name and location of permit holder
2. SIC code number for each permit holder (was converted to NAICS code number)
3. Flow and phosphorus estimates (phosphorus data were not available for all permit holders)
4. Employee counts

The Minnesota Department of Health (MDH), the agency that regulates the quality of drinking water supplies in Minnesota, provided a list of communities that supplement their water supply with continuous phosphate additions (for corrosion control and iron and manganese sequestration) from 2001 to 2003. The MDH list provided the water treatment facility's annual flowrate for all 360 of the systems that add phosphorus. In addition, they provided the residual phosphorus concentrations for

the 120 systems that are required to add phosphorus for corrosion control. These data were used to calculate the total phosphorus contribution to the POTWs from the municipal water supplies.

A number of Minnesota communities were contacted to obtain data or to verify information regarding their wastewater treatment facilities. The types of information provided by these communities included:

- **Industrial Phosphorus Data.** Fourteen out-state (non-metro) communities with industrial phosphorus monitoring programs were contacted and provided data on influent loadings from industrial and commercial dischargers to their wastewater treatment facilities.

- **Population Data.** Many communities were contacted to determine the population served by the wastewater treatment facility.

- **Industrial Discharge Information.** Many communities and industries were contacted to verify the type and volume of wastewater discharge from an industrial source.

The following literature sources were reviewed to obtain information on the sources and amounts of phosphorus discharged to wastewater treatment facilities:

- **Chemical Economics Handbook – Industrial Phosphates** - The handbook provides detailed information on the mass of phosphorus consumed annually in the United States for major commercial, nonagricultural phosphate chemical products. The report provided historical data for the years 1984 through 2000 and forecasted data for the year 2005 for the following major commercial products:
    - Detergent builders
    - Water supply chemicals
    - Food and beverages
    - Dentifrices (oral hygiene products)

- Metcalf and Eddy, Inc. (1991) discusses the components that make up wastewater as well as typical wastewater flowrates and characteristics.

- A number of studies were conducted in the late-1970s and early-1980s that analyzed residential wastewater. These studies segregated wastewater from toilets (human wastes), garbage disposals, dishwashing water, food soils, baths and showers, laundry discharges, and automatic dishwasher detergent, and provided typical flowrates and pollutant characteristics (including phosphorus) for each of these sources. These studies found the following to contribute phosphorus to residential wastewater:
    - Human wastes
    - Garbage disposals
    - Dishwashing water

- Food soils
- Laundry discharges (completed prior to the ban on phosphorus in laundry detergent)
- And automatic dishwasher detergent

The data were provided in terms of daily per capita use rates. It was assumed that no major changes had occurred in the estimates for human waste, garbage disposal waste, and food soils and these data were used to estimate source amounts discharged to wastewater treatment facilities.

- Ligman, Hutzler and Boyle (1974) characterized the types of wastewater generated in a domestic household. They surveyed a total of 50 rural and urban households to determine the various sources and amounts of wastewater generated from the bathroom, the kitchen and the laundry and determined that there was no statistical difference in wastewater pollutant loads for each household.

- Siegrist, Witt, and Boyle (1976) characterized waste flows from individual rural households. They found that on average human waste contains approximately 1.6 grams of phosphorus per person per day.

- Boyle, Siegrist and Saw (1982) focused on treatment of graywater, but also provided a summary of the characterization of wastewater from households.

- Strauss (2000) provided information on the nutrient concentration in human waste and determined that humans excrete in the order of 2 grams of phosphorus per day.

2.2.1.2 Approach for Determining Phosphorus Discharged to POTWs

In addition to determining the point source loading of phosphorus to surface waters in each basin from each of the three types of treatment facilities (POTWs, privately owned treatment facilities, and industrial wastewater treatment systems), the other objective was to identify the sources and estimate the amount of phosphorus discharged to POTWs. Although not required by the legislation (see Section 1.2), the sources of phosphorus and an estimate of the amount discharged into privately owned treatment works was also completed. Finally, the major types of industrial discharge categories were also identified for the industrial wastewater treatment systems. Phosphorus loading to each type of treatment facility was categorized into the primary sources that were considered important (described below).

The following individual and/or categorical sources of phosphorus were evaluated for each POTW:

- Commercial/industrial process wastewater sources (including noncontact cooling water)

- Finished water supply and water treatment chemicals (such as polyphosphate compounds or orthophosphate compounds used for corrosion control purposes)

- Industrial and institutional automatic dishwasher detergent (ADWD)

- Residential automatic dishwasher detergent

- Dentifrices

- Groundwater intrusion into sanitary sewers

- Food soils and garbage disposal wastes (food soils include waste food and beverages poured down the sink, and food washed down the drain as a result of dish rinsing and washing)

- Ingested Human wastes

The following individual and/or categorical sources of phosphorus were evaluated for each privately owned treatment facility:

- Residential automatic dishwasher detergent

- Food soils and garbage disposal wastes

- Human wastes

It was assumed that these systems were small and that no industries would be discharging to a privately owned treatment facility and that the communities served by these systems would not be on a public water supply. Therefore, the commercial/industrial process wastewater sources, finished water supply and water treatment chemicals sources, industrial and institutional automatic dishwasher detergent and groundwater intrusion into the sanitary sewers sources were assumed not to contribute to these facilities.

Because much of the information gathered during the literature search for the various components of the influent wastewater was based on per capita values, it was necessary to accurately determine the population served for each of the POTWs and privately owned wastewater treatment facilities. The population served for each facility was not readily available for all of the permitted facilities. Therefore, the following stepwise approach was taken:

1. When available, the population served by a treatment facility as listed in the Delta database was used, unless comments from individual wastewater treatment plant operators required a modification to the estimates.

2. If population data were not available from the Delta database, the population of the community corresponding to the permit was assumed to equal the population served by the WWTF, which was obtained from the MNPRO database.

3. Communities and the populations served by individual sewage treatment systems (ISTS, [see Appendix H]) were compared to the communities having an NPDES permit as listed in the

Delta database. If a community had both a NPDES permit to discharge to a surface water and was also listed has being served by an ISTS, the difference of the City's population and the ISTS population was used as the population served by the treatment facility. If no information was available, the permit holder was contacted to verify the population served by each system.

4. The complete listing of communities within the state of Minnesota as contained in the MNPRO database were compared to both the NPDES list and the unsewered communities list to verify that all communities within the state were counted. Any unaccounted community with a population greater than 1,000 was contacted to determine their disposition wastewater treatment.

5. Communities with a population of less than 1,000 persons that did not have an NPDES permit and were not listed in the ISTS or unsewered community databases were assumed to be served by ISTS.

A wide variety of commercial and industrial operations discharge wastewater into POTWs under terms of wastewater discharge permits. Industrial process discharge monitoring data from MCES were collected for the eight MCES facilities. In addition to the MCES data, commercial and industrial process monitoring data were collected from the cities of Luverne, Melrose, Moorhead, St. Cloud, Winona, Faribault, Glencoe, New Ulm, Owatonna, Plainview-Elgin, Rochester, Zumbrota, Mankato and Marshall. In addition to the industrial monitoring data, the NAICS code number and number of employees were also obtained. Using this information, the estimated phosphorus load per employee was calculated for the various NAICS code numbers. This information was used to estimate the industrial/commercial process wastewater component of the POTW phosphorus loads. The quantities of phosphorus discharged to the sewer system by commercial and industrial operations for which data were obtained was estimated by extrapolating discharge data to an annual total. The data obtained for the various NAICS code industries were used to estimate the Industrial and Commercial wastewater components of the POTW phosphorus loads where no data were available. An average phosphorus load per employee was then calculated for each NAICS code number. The MCES industrial information received had employee count available for most of the facilities permitted. In addition, MNPRO listed the employee count for all the industries in their database. Employee count was used as the method of adjusting the phosphorus load for the variation of industry sizes within a NAICS code number (four to six digit matches). If there was no match found at the four-digit level, then no estimate of the phosphorus contribution was made.

Phosphorus-based chemicals are sometimes used for corrosion control and metal sequestration purposes in water supply systems. The Minnesota Department of Health provided a list of community water supplies and the average residual phosphorus concentration in the water supply for

the systems that are required to monitor their phosphate additions for the years 2001 through mid-year 2003. The average residual phosphorus concentration from this data was used for each of the communities that were known to add phosphorus, but for which there was no concentration data available. Literature values (Metcalf and Eddy, 1991) indicate that, on average, 70 percent of the water supplied from a water treatment facility is discharged back into a wastewater treatment facility. The phosphorus contribution from municipal water supplies to a POTW was calculated by estimating the annual phosphorus mass used in treatment of the water supply from the MDH data and assuming 70 percent of it is discharged to the POTW.

To estimate the residential ADWD detergent component of the WWTF phosphorus loads, the 2000 data on annual phosphate utilization for ADWD detergent formulation in the United States from the SRI publication Chemical Economics Handbook - Industrial Phosphates (SRI, 2002) was used, along with the estimated U.S. population for 2000, to estimate a per capita ADWD detergent usage of 0.085 kilograms per capita per year (kg/p·yr). This use rate was applied to the population served by each of the POTWs and privately owned treatment facilities to estimate the ADWD detergent components of the phosphorus loads.

Commercial and institutional ADWD detergents are used in restaurants, cafeterias, hotels, hospitals and other institutions, etc. These facilities are not considered as part of the commercial and industrial process wastewater phosphorus contribution as discussed previously. To estimate the commercial and institutional ADWD detergent component of the influent POTW phosphorus loads, 2000 data on annual phosphate utilization for commercial and institutional ADWD detergent formulation from SRI (2002) was used, along with the estimated U.S. population for 2000, to estimate a per capita commercial and institutional ADWD detergent usage of 0.04 kg/p·yr. This per capita use rate was applied to the population served by each of the POTWs to estimate the commercial and industrial ADWD detergent components of the phosphorus loads.

Other consumer products such as scouring cleaners (Comet[®] and Ajax[®]) and home cleaners (Spic & Span[®] and Lime Away[®]) no longer contain phosphorus. Therefore, it was assumed that there was no phosphorus contribution from these products. Commercial and institutional cleaners may use phosphate-based cleaners, but it was assumed that discharge of this source would be accounted for in the industrial and commercial process wastewater component and was not categorized separately.

Several sources were reviewed to determine the phosphorus loading to WWTFs from garbage disposals and from food soils (Siegrist, 1976 and Boyle et al, 1982). For the purposes of this report, food soils are defined as waste beverages and food washed down the sink and food washed down the

sink through dish rinsing and dish washing. The most recent per capita discharge rate of 0.1895 kgP/p·yr was applied to the populations served by each of the WWTFs to determine the phosphorus loading from this source.

Dentifrices are substances such as toothpaste and denture cleaners. Using 2000 data on annual phosphate consumed from dentifrices (from SRI, 2002) and the estimated U.S. population for 2000, the estimated per capita phosphorus contribution from dentifrices was 0.0115 kg/p·yr.

An attempt was made to determine the phosphorus loading from car and truck washes, but there was not enough data available to determine either the amount of flow or the number of car washes discharging to Minnesota POTWs. In addition, since it has become common for car washes to recycle or reuse wash water, no phosphorus load estimate for this source was made in this report.

Measurable effects from inflow and infiltration (I & I) at WWTFs will depend on the age of the sewer system piping, the total length of the sewer system piping and the joint construction of the sewer pipes. An average infiltration rate was obtained from data provided by MCES, based on average annual I & I flow estimates for their eight wastewater treatment facilities. These facilities vary in size and age and were considered to be representative of the systems throughout the state. The average I & I rate was approximately 10 percent of the total influent annually for the eight Twin Cities Metropolitan Area wastewater treatment facilities operated by MCES. The phosphorus concentration in I & I was estimated from phosphorus concentration data provided by the MPCA for each of the aquifers throughout the state. An average phosphorus concentration of 0.035 mg/L was assumed to be representative of the shallow groundwater throughout the state.

Human waste-derived phosphorus was separated from the total phosphorus load to each of the POTWs and privately owned treatment systems by difference, subtracting all other estimated phosphorus contributions from the total phosphorus inflows. This value was converted to a per capita value and then used to validate the computations for each WWTF by comparing it to literature values for blackwater (ingested human waste). Literature values ranged from 1.2 grams of phosphorus per capita per day (g/p·d) (Siegrist, 1978) to 2 g/p·d (Strauss, 2000).

2.2.1.3 Approach for Determining Phosphorus Loading to Surface Waters

Data on all municipal, private and industrial and commercial dischargers were obtained from the MPCA Delta database. As a first step, the stations for each permit were reviewed to verify that a valid discharge to a surface water was occurring for each station in each permit. As a result of this review, the following stations were deleted for this study:

1. Stations that represented land application of wastewater,

2. Stations that strictly represented a stormwater runoff discharge,

3. Permits that had no influent and effluent flow data. It was assumed that if there was no data for either the influent or the effluent stations, that there had been no discharge from that facility.

The NPDES discharges were separated into the following categories as part of the review process:

1. Domestic vs. industrial flow was verified. In a few cases, the Delta database designation was modified. For example, prisons and schools were changed from an industrial source to a domestic source

2. Noncontact cooling water sources were noted, and

3. Mine pit dewatering sources were noted

Next, the influent and effluent flowrates for the NPDES surface water permits and stations were reviewed. If only influent flow data were available from the Delta database, the effluent flow was assumed to be equal to the influent flow. Similarly, if only effluent flowrates were available from Delta, the influent flowrates were assumed to be equal to the effluent flowrates. Pond systems presented a challenge in that they discharge intermittently and, when they do, the flowrate is relatively high. For many pond systems there was no discharge information available because they had not discharged during the period of record. In other instances the average annual effluent flow from a pond system greatly exceeded the annual average influent flow, so the average annual effluent flowrate was assumed to be equal to the measured influent flowrates for pond systems. For industrial wastewater treatment systems, only effluent flow data were required for this analysis. Following flowrate database development all flowrate data were then validated. The average flowrate and standard deviation was calculated for each permit station. Permits with high standard deviations were manually reviewed to spot the general trend in discharge rates and correct obvious errors.

The approach used to determine the phosphorus loading from each of the three types of facilities to the basin is very similar and is described below. Phosphorus loads were determined by multiplying the influent and effluent flowrates by the influent and effluent phosphorus concentrations, respectively. Phosphorus concentration data was obtained from the Delta database. Since many permits do not include limits and/or monitoring requirements for phosphorus, there were no effluent phosphorus data available for these permits. In addition, many facilities that have an effluent phosphorus limit monitor only the effluent phosphorus and do not monitor the influent phosphorus concentrations. In these cases, it was necessary to estimate phosphorus concentrations from other

sources. The annual influent and effluent phosphorus loads for each wastewater treatment facility and the effluent phosphorus loads for the industrial sources for which data were available were estimated as the products of the average phosphorus concentrations and flowrates extrapolated over the monitoring period. Missing POTW and privately owned treatment facility phosphorus concentrations were estimated by assuming the calculated basin average phosphorus concentration (as described in the previous paragraph) for similar facility types. In a limited number of cases calls were made to the permittee to verify phosphorus effluent concentrations.

The various types of industries discharging phosphorus from industrial wastewater treatment systems were identified. For each industrial wastewater discharger, their North American Industry Classification System (NAICS) code number was identified. This NAICS code allowed the data to be sorted by industry type. Effluent phosphorus concentrations for industrial wastewater treatment systems that did not have monitoring data were estimated from phosphorus data for industries with like NAICS codes. Noncontact cooling water dischargers were identified through review of the NPDES permit data. When available, the amount of phosphorus in these discharges was calculated from data contained in the Delta database. For each noncontact cooling water discharge, the source of the water was identified as were additions of phosphorus-based corrosion control chemicals. In calculating the phosphorus loads associated with noncontact cooling water, reported data on discharge volumes and phosphorus concentration were used whenever they were available. However, when the phosphorus concentration of noncontact cooling water was not specified in the permit data, the source of the cooling water was determined and any information on phosphorus additives was investigated with the MPCA. If the source of the cooling water was the municipal water supply and no phosphorus was added, it was assumed that the phosphorus concentration discharged was equivalent to the municipal water supply value. If the source of the cooling water was an on-site well, the phosphorus concentration was assumed to be equal to the groundwater phosphorus concentration. Finally, if the source of the cooling water was the same body of water that received the effluent and no phosphorus was added for water treatment, it was assumed that there was no additional phosphorus load to the surface water.

2.2.2   Nonpoint Sources of Phosphorus

This section provides a discussion regarding determination of nonpoint sources of phosphorus to Minnesota watersheds. For the purposes of this analysis, nonpoint sources of phosphorus include diffuse runoff associated with rainfall and snowmelt events as well as atmospheric fallout and discharge from distinct discharge points that are not individually regulated under state and federal

pollution permit programs. Detailed discussions about the assessment of these source categories are contained in Appendices C through J.

2.2.2.1   Agricultural Runoff

Runoff from agricultural lands contributes phosphorus to surface waters primarily through rainfall and snowmelt runoff from pasture and cropland, as well as direct runoff from open feedlots. The complex nature of the source and transport factors that determine how much phosphorus might be associated with runoff from agricultural lands required that separate approaches be used to estimate phosphorus loadings to surface waters from cropland and pasture runoff, which is described in Section 2.2.2.1.1, and direct runoff from open feedlots, discussed in Section 2.2.2.1.2. Each section provides a general discussion about how the phosphorus contribution to surface waters from each source of agricultural runoff was quantified. More detailed discussions about the methodology used for each analysis is included in Appendices C and D.

2.2.2.1.1   Cropland and Pasture

A combination of transport and source factors directly influence phosphorus (P) movement from cropland and pasture to surface waters (Sharpley et al., 1993). The USDA developed a P Index that integrates both transport and source factors to identify areas vulnerable to P export (Lemunyon and Gilbert, 1993). Transport factors include the mechanisms by which P is delivered to surface waters, such as erosion and runoff. Source factors represent the amount of P available for transport, including soil test P and P applied (rate and method) in fertilizer and organic forms. The objectives of this analysis were to assess phosphorus loadings to Minnesota's ten major drainage basins from agricultural runoff and erosion, under various flow conditions, and evaluate the uncertainty of this assessment. This section discusses how the phosphorus contribution to surface waters from cropland and pasture runoff was quantified. A more detailed discussion about the methodology used for this analysis is included in Appendix C.

This analysis was accomplished by using and extending a regional phosphorus index approach published by Birr and Mulla (2001). Phosphorus index values were estimated for Minnesota watersheds and agroecoregions based on phosphorus transport and source factors such as erosion during dry, average and wet years, streamflow during dry, average and wet years, contributing distance from surface waterbodies during dry, average and wet years, soil test phosphorus, and rate and method of land applied phosphorus from fertilizer and manure. Phosphorus index values were compared with field data on phosphorus loss from four sites over five years to estimate phosphorus export conditions. Phosphorus export coefficients were multiplied by the cropland contributing area within 100 m of surface water bodies to obtain phosphorus loadings from the edge of this

contributing area. It should be noted that throughout most of Minnesota, we believe that the risks of phosphorus transport to surface waters are greatest in the contributing corridor within about 100 m from surface waterbodies. Due to topographic variations along surface waterbodies, in some areas phosphorus contributions from overland runoff and erosion may occur from as far away as several hundreds of meters. In contrast, where berms are present along waterbodies it may be unlikely for a significant amount of surface runoff or erosion to enter surface water. Thus, the 100 m contributing corridor should be viewed as a regional average for contributions of P to surface waters from runoff and erosion on adjacent cropland.

Several alternative agricultural management scenarios were investigated and compared to a baseline scenario involving an average climatic year and existing rates of adoption of conservation tillage and existing rates of phosphorus fertilizer applications. The first alternative management was a scenario in which moldboard plowing is used on all row cropland. This is a worst case scenario for erosion, and exemplifies phosphorus losses typical of an era that existed twenty or more years ago. This scenario allows us to evaluate the extent of progress in controlling phosphorus losses over the last twenty years due to improvements in tillage management. The last scenario involves decreasing or increasing the area of cropland within 100 m of surface waterbodies. Decreases in area of cropland could correspond to land retirement programs such as those promoted in the Conservation Reserve and Conservation Reserve Enhancement Programs. Increases in cropland area would correspond to putting grass or forest riparian areas into production, alternatively this could be viewed as increasing the distance for cropland areas (now assumed to be 100 m) that contribute phosphorus to surface waters.

The following sections provide an overview of the modified phosphorus index, developed at the regional scale by Birr and Mulla (2001), and an approach for revising and utilizing the modified phosphorus index to estimate phosphorus loadings from agricultural sources to each of the ten major drainage basins in Minnesota during low, high and average flow conditions.

Birr and Mulla (2001) developed a modified version of the P Index, originally developed jointly by the USDA (ARS, CSREES, and NRCS), to prioritize phosphorus loss vulnerability at the regional scale from 60 watersheds located within Minnesota. This modified (regional) version of the P Index uses readily available data associated with the transport and sources of P. Transport factors include the mechanisms by which P is delivered to surface waters, such as erosion and runoff. Source factors represent the amount of P available for transport, including soil test P and P applied (rate and

method) in fertilizer and organic forms. The following discussion describes how each of the transport and source factors were initially determined by Birr and Mulla (2001):

- Soil erosion potential was calculated using the Universal Soil Loss Equation (USLE) as outlined by Wischmeier and Smith (1978). The Minnesota state soil geographic database (STATSGO) was used to supply many of the variables needed to calculate erosion potentials for each of the watersheds (USDA, 1991). Erosion potential was calculated for each soil type within a STATSGO map unit. Rainfall runoff factors (R) for each county were based on values provided by Wischmeier and Smith (1978). The STATSGO database provided a soil erodibility factor (K) for each soil type within a STATSGO map unit. The slope-steepness factor (S) represents an average of the high and low slope values given for each soil type within a STATSGO map unit. The slope-length factor (L) was assumed to be 46 m. A 1:250,000 scale landuse/landcover coverage developed by the USGS in the late 1970s and early 1980s was used to determine erosion potentials spatially coincident with cropland and pastureland (USEPA, 1994). An erosion potential value for all cropland and pastureland within a watershed was determined using the percent of each STATSGO map unit covering a watershed. The landuse coverage did not differentiate spatially between cropland and pastureland; however, Census of Agriculture data indicate that pastureland represents about 11% of this classification category in Minnesota (National Agricultural Statistics Service, 1999). Differences in potential erosion for the two land uses were accounted for in the determination of the C factor based on the proportion of hay reported for a particular county. Cropping management factors (C) were adapted from values provided by the USDA (1975) and Wischmeier and Smith (1978) for corn, wheat, soybean, hay, sugar beet, potato, oat, and barley. The C factors were calculated for each county based on the area of each harvested crop covering the county. Watershed values for the C factors were weighted based on the proportion of the watershed that was covered by the county. The C factor calculations include crop rotation effects but not the variation in tillage effects. The conservation practice factor (P) was assumed to be 1, because it could not be accurately quantified at the regional scale. The overall erosion potential value for each watershed represents the product of the area-weighted C factor and the variables R, K, and LS for each watershed (A = RKLSCP).

- Average annual runoff values for each watershed were derived from the average annual discharge monitored from 1951 to 1985 for 327 stations distributed throughout Minnesota (Lorenz et al., 1997).

- The area of cropland and pastureland within 91.4 m of drainage ditches and perennial streams (the primary contributing corridor) was determined using hydrography coverages developed by the Minnesota Department of Transportation (1999) and the USGS (1999). The USGS landuse/landcover coverage (USEPA, 1994) was used to determine the percentage of cropland and pastureland within the 91.4 m proximity to watercourses for each watershed.

- Mean soil test P levels for each county represented a 5-yr database consisting of 22,421 Bray-1 extractable P (Brown, 1998) samples analyzed by the University of Minnesota's soil testing laboratory. Soil test P levels for each watershed were based on the area of the watershed covered by each county.

- Data for P-fertilizer sales by county were obtained from the Minnesota Department of Agriculture (1997). Fertilizer P values for watersheds were based on a summation of area-weighted county-based values intersecting the watersheds. The total area of fertilized land within each watershed was determined using the same procedure based on reported county values (National Agricultural Statistics Service, 1999). The aggregated fertilizer P value was divided by the aggregated reported fertilized land for each watershed to determine fertilizer P application rates.

- The P content of livestock manure was calculated based on the total number of cattle, swine, broilers, and turkeys reported within each county (Midwest Planning Service, 1985; Schmitt, 1999; National Agricultural Statistics Service, 1999). The total amount of manure P was derived for each watershed based on the summation of area-weighted county values intersecting the watersheds. The reported total cropland area was also determined using the same procedure (National Agricultural Statistics Service, 1999). The aggregated total P content of manure was normalized by the aggregated total cropland area for each watershed to determine organic P application rates.

- For the modified P Index, each site characteristic is assigned a weighting factor based upon the premise that site characteristics have a varying impact on P loss to runoff. Each site characteristic has an associated P loss rating value (very low, low, medium, high, and very high) using a base of 2 to reflect the higher potential for P loss associated with higher rating values. The P Index rating is the summation of the product of the rating value and corresponding weighting value for each site characteristic. Because P application method could not be accurately depicted at the regional scale, the highest organic and fertilizer P application method rating values were used to represent a worst-case scenario. Categories

corresponding to the rating values were derived by segregating the distribution of statewide values for each site characteristic into five classes using the quantile classification method available in ArcView software (ESRI, 2000).

This section provides an approach for revising and utilizing the modified (regional) phosphorus index (from Birr and Mulla, 2001) to estimate phosphorus loadings from agricultural sources to each of the ten major drainage basins in Minnesota during low, high and average flow conditions. The following adjustments to the modified phosphorus index computations and supplementary tasks were used to improve and update the analysis of phosphorus loading:

- The MPCA has developed and updated a feedlot inventory and manure management database (with an associated GIS coverage), based on registered feedlot data obtained from each of the counties. The total amount of manure P was derived for each agroecoregion and watershed based on the summation of area-weighted township values intersecting the agroecoregions or watersheds. The aggregated total P content of manure can then be normalized by the aggregated total cropland area for each agroecoregion or watershed to determine and revise the organic P application rates.

- Phosphorus fertilizer sales data by county for the most current crop year (2002) were obtained from the Minnesota Department of Agriculture and used to update this part of the modified phosphorus index computations based on a summation of area-weighted county-based values intersecting the agroecoregions or watersheds.

- GIS coverages for runoff volumes in each agroecoregion or watershed under average, high and low flow conditions were developed to evaluate how phosphorus export from agricultural lands would be expected to change with varying climate conditions. Runoff volumes were estimated as described in Sections 2.1 and presented in Section 3.1. In addition, rainfall runoff erosivity (R values) was estimated for the USLE for dry, average and wet years corresponding to the low, average and high flow conditions. These estimates were based on an algorithm developed for monthly precipitation data by Renard and Freimund (1994). The modified phosphorus index values and total phosphorus export were then computed for each of the agroecoregions or watersheds under high and low flow conditions, using the corresponding values for runoff volume and rainfall runoff erosivity.

- Based on farm survey data collected by the Minnesota Department of Agriculture, phosphorus application methods are generally much better than those assumed by Birr and

Mulla (2001). A majority of farmers apply their phosphorus fertilizer with the planter or using incorporation before crop planting. In view of this, a statewide medium loss potential was applied for method of fertilizer P application method, corresponding to fertilizer applied before the crop and incorporated immediately. An initial scenario involving a medium loss potential for the method of manure application was developed for the entire state. Subsequently, a second scenario was developed assuming variability in the loss potential associated with method of manure application. Manure P application methods vary primarily in response to the type of animal species. Manure from beef, dairy, and poultry is high in solids, while manure from hogs is high in liquid. Beef operations tend to be small in scope, have a tendency towards inadequate manure storage facilities, and manure from these operations tends to be hauled on a daily basis. Beef operations also tend to involve cattle wading in streams. Dairy operations tend to have adequate manure storage facilities, and manure is applied followed by a tillage operation to incorporate manure. Poultry operations tend to have adequate manure storage facilities, and the manure is incorporated using tillage following land application. Hog operations tend to have adequate storage facilities, and the manure is land applied using injection. In terms of the phosphorus index, this means that beef operations tend to have a very high phosphorus loss potential, dairy and poultry operations tend to have a medium loss potential, while hog operations tend to have a low loss potential. The geographic variability in phosphorus loss potential associated with these variations in method of manure application was evaluated using the number of animal units of different species from the MPCA feedlot inventory database. The effect of this variability and/or uncertainty in method of manure application was estimated using the modified phosphorus index.

- Birr and Mulla (2001) states that spatial trends in soil erosion potential observed throughout Minnesota are potentially influenced by both the underlying assumptions used in the methodology and the exclusion of factors that control soil erosion. A lack of detailed information pertaining to the spatial variation in C and P factors may have caused the spatial distribution of erosion potential values to vary more gradually across the region than is realistic. The spatial variation in the C factor of the USLE was estimated by accounting for the effects of crop rotations, the effects of conservation tillage on crop residue levels, and the effects of existing acreage of land in Conservation Reserve Program (CRP). Typically the C factor for land in CRP is 0.001 or so, while row cropland has a C factor varying from 0.05 to 0.4 depending on the rotation and the amount of crop residue present. Three scenarios were evaluated to account for the influence of tillage methods on crop residue levels remaining

after planting. These were a scenario involving conventional tillage with no residue left (worst C scenario), and a scenario involving conservation tillage leaving more than 50% of the soil covered by crop residue (best C scenario). This is not typical of existing crop rotations or tillage management systems in Minnesota, nor is it a goal of existing watershed restoration or conservation programs to achieve this high level of crop residue cover. Also estimated was a scenario for average crop residue cover (average C scenario) based on county tillage transect data for the percent of fields with conservation tillage (30% residue cover). In the average C scenario, we developed a weighted C factor based on the relative area of cropland in conservation tillage versus moldboard plowing. Data for the C factors of various crop rotations with varying levels of crop residue were estimated using tables provided by the USDA-NRCS. Thus, using information on crop rotations, crop residue levels, and acreage of land in CRP, we developed scenarios for both soil erosion by water and the modified phosphorus index involving the C factor of the USLE. Variability in the P factor of the USLE was estimated using the Local Government Annual Reporting System (LARS) database of conservation practices provided by the Board on Soil and Water Resources (BWSR). This database was edited to estimate the area of supporting conservation practices affecting the P factor implemented from 1997-present in Minnesota counties. These practices include terracing, contour strip cropping, filter strips, sediment basins, and restored wetlands. Each practice was assigned a typical P factor. Since supporting conservation practices have typically been implemented for the last 50 years, we assumed that the area where these practices were implemented was 10 times greater than the area determined using the LARS database. A county average P factor was then determined using the area weighted P factors for land with supporting practices and the land without supporting practices (P=1). The variability and/or uncertainty associated with conservation practices, such as conservation tillage, contour stripcropping, terracing, and other supporting practices was then estimated for agroecoregions and watersheds using the modified phosphorus index.

Two different approaches were tested for converting phosphorus index values to edge of field phosphorus losses to surface waters. The first method attempted to estimate phosphorus losses from the edge of field based on monitoring data for phosphorus loads in 53 Minnesota streams and rivers. This method did not successfully produce meaningful results. The second method estimated phosphorus losses from the edge of cropland fields based on export coefficients which were derived from the phosphorus index values. This is the method used for final estimates of basin wide phosphorus loadings to surface waters from the edge of cropland fields. The following discussion provides details about each methodology:

- Existing data for phosphorus loads measured by watershed water quality monitoring was summarized for 53 ditches, streams and rivers throughout Minnesota. The data was separated according to flow conditions into phosphorus loads for dry, average and wet years. Estimates for phosphorus losses discharged to surface waters in the same watersheds from non-agricultural rural, streambank erosion, and point sources of phosphorus were also obtained. The monitored phosphorus loads were adjusted by subtracting the losses from non-agricultural rural and point sources of phosphorus, and by subtracting half of the phosphorus losses from streambank erosion. Only half of the streambank erosion losses were subtracted because much of the sediment from streambank erosion is transported as bedload, which is not measured in most water quality monitoring studies. The remaining phosphorus loadings were then divided by the area of cropland within 91 m of streams and ditches to provide an estimate of the potential phosphorus losses from the edge of cropland fields. The resulting adjusted phosphorus yields were not very consistent with expected results, and were not deemed meaningful. Many of the adjusted phosphorus yields were negative in dry years because the point source loadings were larger than the monitored phosphorus loadings in the watershed. This could be due to phosphorus uptake by algae or plants. In wet years the adjusted phosphorus yields exhibited a huge range, from nearly zero to several hundreds of kg P/ha. This was most likely the result of several factors. The first factor is that the phosphorus monitoring load data were collected using a variety of methods, ranging from grab samples to automated water quality sampling. The second is that the monitored loads were collected over different lengths of time, ranging from a single season to multiple years. The third factor is that the adjusted phosphorus losses were not corrected to account for contributions of phosphorus from ISTS, atmospheric deposition, or urban runoff. This led to unrealistically high adjusted phosphorus loads during average and wet years. The fourth factor is that the phosphorus delivery ratio from each non-agricultural source should be varied by source and by flow regime when adjusting the monitored loads. For example, the delivery ratio for streambank erosion (assumed to be 0.5) would vary with flow regime. As a result, this approach for estimating edge of field phosphorus losses from agricultural sources was not used.

- Birr et al. (2002) found that there is a strong linear correlation ($r^2 =0.82$) between a version of the modified phosphorus index values (from Birr and Mulla, 2001) and the pathway (or field scale) phosphorus index values. The modified phosphorus index values are typically thirteen times higher than the pathway phosphorus index values. Similarly, there is a strong linear correlation between the estimated pathway phosphorus index values and the observed

phosphorus export (expressed in kg/ha/yr) at the field scale. The pathway phosphorus index values are typically five times higher than the total phosphorus export, at the field scale (Mulla, 2003). This suggests that we can estimate phosphorus losses from the edge of cropland fields by dividing the phosphorus index results by a factor of approximately 65. This gives an estimate of the losses of total phosphorus to surface waters from cropland and pastureland in units of kg/ha/yr, which represents the phosphorus export coefficient for agricultural land. Since the version of the modified phosphorus index used in this study is slightly different from the one used by Birr et al. (2002), we decided to develop a relationship between the phosphorus index and the phosphorus export coefficient using phosphorus loss data compiled from University of Minnesota research at four sites in or near Minnesota. The sites are located near Morris, Minnesota (Ginting et al., 1998), Lancaster, Wisconsin (Munyankusi, 1999), and two sites in Scott County, Minnesota (Hansen et al., 2001). These sites involved measurements of total phosphorus losses from the edge of agricultural fields (typically a corn and soybean rotation) ranging in area from 0.5 to 1.6 ha. Data from these sites were collected between 1996 and 2000. Two of these years experienced average climatic conditions, two were a little wetter than average, and one was a little drier than average. Fields were treated using a range of tillage and manure management methods. The tillage treatments included moldboard plowing, chisel plowing, ridge tillage, and no-tillage. Manure treatments included no manure, heavy rates of manure, and variations in timing of manure application. Total phosphorus losses from the fourteen individual treatments at these four sites ranged from 0.1 to 2.3 kg/ha/yr, with an average of 0.68 kg/ha/yr in total phosphorus loss from the edge of field. The counties where these four research sites are located have a range of tillage practices, with the percent of farmland having at least 30% crop residue cover ranging from about 47% in Scott and Stevens counties to about 64% of cropland with at least 30% residue cover in Houston county, the nearest county in Minnesota to Lancaster, Wisconsin. The phosphorus index values for an average climatic year and the existing residue cover adoption rates indicated above are 24, 32, and 43 in the Chippewa, Root and Lower Minnesota watersheds, respectively. If we take the P Index values for each watershed and divide them by the average phosphorus losses for the study sites in that watershed, the resulting conversion factor (or divisor) is 78. If on the other hand, we take the average phosphorus index value for these three regions of 33 and divide this by the average phosphorus loss from the edge of field in these experiments at four sites (0.68 kg/ha), we obtain 48.5 as the conversion factor between the phosphorus index and the phosphorus losses from the edge of field. This conversion factor is somewhat lower than both the conversion

factor of 65 initially obtained using the relationship between the matrix and pathway versions of the phosphorus index, and the conversion factor of 78 obtained by averaging the divisors obtained for each watershed. Taking the divisor of 48.5 as the most realistic estimate for the conversion factor, and rounding this conversion factor up to 50 for significant digits, we then divided all the phosphorus index values for each watershed and agroecoregion in Minnesota by 50 to obtain phosphorus export coefficients. The resulting phosphorus export coefficients for an average year are 0.43 kg/ha/yr for major watersheds and 0.44 kg/ha/yr for agroecoregions. For wet years the export coefficients are 0.65 kg/ha/yr for watersheds and 0.68 kg/ha/yr for agroecoregions. For dry years the export coefficients are 0.21 kg/ha/yr for watersheds and 0.22 kg/ha/yr for agroecoregions. According to Heiskary and Wilson (1994), recommended phosphorus export coefficients for Minnesota agricultural lands are 0.2, 0.4, or 0.6 kg/ha/yr for low, mid, and high export risk conditions. Hence, our statewide average export coefficients for low, mid, and high export risk conditions (0.21, 0.43, and 0.65 kg/ha/yr) compare favorably with those recommended by Heiskary and Wilson (1994).

The procedure for estimating basin wide loads of phosphorus exported from the edge of agricultural fields is to multiply the export coefficients described above by the area of cropland within a distance of 100 m of surface water bodies (perennial and intermittent streams, ditches, wetlands, and lakes). On average, about 32% of all cropland lies within this distance of surface water bodies statewide, with a range of from 21 to 52% in major river basins. This procedure accounts for the variability in risk of phosphorus loss from the edge of field due to climatic effects as well as the variability in soil, management and hydrologic factors. Variability in the phosphorus index values across the state are translated into variability in phosphorus losses from the edge of field using the export coefficient. On top of this, we added another 10% to the phosphorus loadings to account for contributions from cropland farther than 100 m from surface waterbodies. This is consistent with results from research conducted by Sharpley et al. (1994), Daniel et al. (1994) and Gburek et al. (2000), who concluded (in SERA-17, 2004) that only 10% of the phosphorus loadings to surface waters from overland transport on agricultural lands arise from outside the primary contributing corridor (100 m or farther from surface water bodies). The added 10% does not include additional phosphorus contributions that arise from surface tile inlets or subsurface tile drains. As previously discussed, we believe that the risks of phosphorus transport to surface waters are greatest in the contributing corridor within about 100 m from surface waterbodies. Due to topographic variations along surface waterbodies, in some areas phosphorus contributions from overland runoff and erosion may occur from as far away as several hundreds of meters. In contrast, where berms are present along waterbodies it may be unlikely for a significant amount of surface runoff or erosion to enter surface water. Thus, the 100 m

contributing corridor should be viewed as a regional average for contributions of P to surface waters from runoff and erosion on adjacent cropland.

As mentioned above, the current methods of estimation do not consider the influence that surface tile intakes farther than 100 m may have on phosphorus loadings. To include the effects of surface tile intakes we would need to know the number of tile intakes per unit area, the area of cropland contributing to tile intake flow, and the phosphorus export coefficients for surface tile intakes. These data are not available for Minnesota in enough detail to be confident about their representativeness. Since depressional areas around tile inlets generally trap 60-80% of the sediment and phosphorus flowing to the inlets, the phosphorus export coefficient for surface tile intakes is smaller than that for direct overland flow to surface waters (Ginting et al., 2000). Ginting et al. (2000) studied phosphorus loads carried by surface tile intakes in two small catchments located in the Watonwan watershed of the Minnesota River basin. They found that, over a three year period with slightly below precipitation amounts, phosphorus loads carried by surface tile intakes averaged 0.099 kg/ha annually, with measured concentrations of phosphorus in surface tile intakes as high as 4 mg/L. This loading (0.099 kg/ha) is significantly smaller than the amounts of phosphorus transported by surface runoff and erosion in the same region (0.68 kg/ha). There were three surface tile intakes studied by Ginting et al. (2000), and the average phosphorus load transported by each tile intake annually was 2.82 kg/yr. Surveys of surface tile intake density in 32 small watersheds within the Minnesota River basin (MPCA, 1994) show that there is one surface tile intake for every 23 to 1210 acres in the watershed. The average is one surface tile intake for every 100 or so watershed acres (the acreage that actually contributes to surface tile intake P loads is smaller than this, but few data exist to know what the contributing acreage actually is). If we assume that there is one surface tile intake for every 100 acres within the poorly drained soils of the Minnesota River basin, we estimate that there are roughly 33,333 surface tile intakes in the basin. Assuming a phosphorus load of 2.8 kg/yr for each tile intake, the total phosphorus loading from surface tile intakes to surface water bodies in the Minnesota River basin would result in 94,000 kg per year. This is approximately 18% of the phosphorus loading from cropland within 100 m of surface waters in the Minnesota River basin during an average year (517,862 kg/yr).

Similarly, the current methods do not consider the influence of subsurface tile drainage on phosphorus export to surface waters. Randall et al. (2000) studied losses of phosphorus in subsurface drainage in a four year manure and fertilizer study on a Webster clay loam typical of the poorly drained soils in the Minnesota River basin. According to Randall et al. (2000), on average over half of the drainage flows carry non-detectable amounts of phosphorus. The remainder of drainage flows

have a concentration of total phosphorus averaging about 0.03 mg/L (with maximum observed concentrations of about 0.12 mg/L), for an average annual loss of 0.027 kg P/ha. If this rate is applied to the area of cropland in the Minnesota River basin having tile drainage, it gives a phosphorus loading of about 30,000 kg/yr, which is quite small (6% of total) compared to the phosphorus loading from cropland within 100 m of surface waters during an average year (517,862 kg/yr). Subsurface drainage phosphorus loads from other major basins would be much smaller, because tile drainage is of limited extent in basins other than the Minnesota River basin. The plains of the Cedar, Lower Mississippi and the southern watersheds in the Upper Mississippi River basins have similar geomorphology, precipitation and land uses that would also control drainage practices, but no attempt was made to quantify the phosphorus loads from subsurface drainage in these basins as part of this analysis. The phosphorus loadings from subsurface tile drains collected by Randall et al. (2000) are the only data published in peer reviewed journals from Minnesota studies. Other studies of phosphorus losses in Minnesota subsurface tile drainage include those conducted by Alexander and Magdalene (1998) from 1995 to 1997 at the Rollings East Tile (RET) site, and by the Minnesota Department of Agriculture from 1998 to 2001 at the Red Top farm, both of which are located in the Minnesota River basin. The study by Alexander and Magdalene (1998) does not estimate phosphorus loadings from subsurface tile drainage, instead, it reports only the concentrations of phosphorus measured. The concentrations of phosphorus measured in subsurface tile drainage by Alexander and Magdalene (1998) are very comparable in seven out of ten storms they monitored to the concentrations measured by Randall et al. (2000) over a four year period. In two other storms monitored by Alexander and Magdalene (1998), the phosphorus concentrations ranged between 0.42 and 1.5 mg/L, much higher than those measured by Randall et al. (2000). At the Red Top farm study, based on 9 field years of water quality monitoring data for average climatic years, the annual average phosphorus loading from subsurface tile drains was 0.11 kg/ha. These larger field drainage systems were constructed of concrete tiles which differ from the smaller plot based plastic drain tiles studied by Randall et al. (2000). Based on this comparison, we conclude that more research is needed to accurately define the mean and range in phosphorus loading from subsurface drainage tiles in the Minnesota River basin. Not enough research data are available to reliably estimate the phosphorus loadings from surface tile intakes or subsurface tile drains to surface waters in the Minnesota River basin during dry or wet climatic years. As a first approximation, we can scale the phosphorus loadings from tile drains so that they have the same relative ratio as the phosphorus index based loadings for the Minnesota River basin in dry, average and wet years (262,851; 517,862; and 759,749 kg/yr, respectively). This gives phosphorus loadings from subsurface tile drains of 15,227 kg/yr during dry years and 44,013 kg/yr during wet years. Using the same approach, phosphorus loadings from

surface tile inlets during dry and wet years would be 47,711 and 137,906 kg/yr, respectively. As previously discussed, this approach substantially overestimates the phosphorus loadings in dry years.

2.2.2.1.2 Feedlot Runoff

The primary way that feedlots contribute phosphorus to surface waters, apart from land application of manure, is through open lot runoff during precipitation and snowmelt events. Overall, a small fraction of the total manure phosphorus generated at feedlots enters waters during precipitation and snowmelt events. Many feedlots do not have an open lot because they keep animals inside the barn most or all of the time, while many of those with outdoor open lots collect runoff in impoundments or treat the runoff as it passes through downslope vegetation. Yet many feedlots still maintain open lots. This section discusses how the phosphorus contribution to surface waters from feedlot runoff was quantified. A more detailed discussion about the methodology used for this analysis is included in Appendix D.

Most of this manure phosphorus (P) generated will be applied to cropland. However, a fraction of the manure P can be lost in feedlot runoff during precipitation or snowmelt events. Most feedlots with open lot runoff are from smaller beef, dairy and swine feedlots, with much fewer instances of non-compliance observed for moderate and large sized feedlots (Mulla et al., 2001). Phosphorus runoff loading from open lot feedlots can be estimated with a feedlot evaluation model developed in Minnesota (Young et al., 1982). The (FLEval) model was developed to estimate pollutant loadings at the feedlot edge and to account for any contaminant retention/treatment that occurs in downslope vegetation and cropland. The Board of Water and Soil Resources developed an equation to estimate annual loadings and annual runoff from the FLEval model predictions. The model predicted that between 0.1 and 1.1 percent of phosphorus generated at feedlots with inadequate runoff controls will enter surface waters.

The following discussion summarizes the steps taken to develop estimates of P loading to surface waters from open lot runoff:

- **Step 1.** Determine the number of beef, dairy and swine animal units found at all feedlots with open lots (excluding feedlots with 1000 or more animal units).

  Step 2. Multiply the results in step 1 by the annual manure P generated by each type of livestock. This provides P generated by livestock in all open lots.

  Step 3. Multiply the results in step 2 by the estimated percentage of open lot feedlots that contribute phosphorus during certain storm events. This provides P generated by livestock at feedlots that contribute P to waters.

Step 4. Multiply the results in step 3 by the typical fraction of P that is lost to surface waters during low, average and high flow years. This provides the estimated P loading to surface waters from open lots.

A more detailed discussion of the results of each of the above steps is included in Appendix D. The results of each of the calculations for the 4 steps is shown and discussed in Section 3.3.2.2.

2.2.2.2 Atmospheric Deposition

Phosphorus in the atmosphere can be derived from a number of sources, including natural sources such as pollen, soil (from wind erosion) and forest fires, as well as anthropogenic sources such as fertilizer application and oil and coal combustion. Agricultural activities (pre-planting field preparations, harvesting) can increase the amount of soil-derived phosphorus in the atmosphere. Phosphorus can also be released into the atmosphere in vapor form from various materials (sewage sludge, landfills) by microbial reduction processes. The atmosphere contributes phosphorus and phosphorus-containing material to terrestrial and aquatic ecosystems by wet (precipitation in various forms such as rain, sleet or snow) and dry (very small particles) deposition. This section provides a general discussion about the methodology used to quantify the amount of phosphorus entering surface waters from this source category. A more detailed discussion of the methodology used for this analysis is included in Appendix E. The results of the phosphorus loading computations for this source are discussed in Section 3.3.3.

A literature review indicated that limited data are available from Minnesota sources to estimate phosphorus deposition to the surface waters. The previous best source of information for precipitation input (wet deposition) of phosphorus to Minnesota watersheds is Verry and Timmons (1977). No data on dry deposition of phosphorus in Minnesota was identified. The following sources of data were considered to be the best available for providing estimates of atmospheric phosphorus inputs to Minnesota's surface waters.

MPCA:
1. Nutrient (including phosphorus) and metal concentrations in precipitation from a special study conducted from August 1999 to September 2001 at four monitoring sites in Minnesota
2. PM10 air concentrations determined from particulate filters and elemental speciation of the PM10 mass by X-ray Fluorescence (XRF) analysis for the 30 sites included in the Statewide Air Toxics Monitoring Study (1996-2001).

National Atmospheric Deposition Program (NADP):

1. Annual volume weighted calcium concentrations in precipitation for the period of record from NADP sites located in, and adjacent to, Minnesota.

2. Monthly volume weighted calcium concentrations for four sites (Fernberg, Marcell, Camp Ripley, Lamberton) for use in establishing the relationship between phosphorus and calcium in precipitation for NADP sites.

Minnesota Department of Natural Resources, State Climatology Office: Annual normal precipitation amount for each river basin basis was obtained from the State Climatology Office.

The phosphorus concentrations from the special study, along with NADP calcium data, were used to derive the relationship between phosphorus and calcium in precipitation for the four NADP monitoring sites. The relationship between phosphorus and calcium in precipitation at these four NADP sites was then applied to the entire state.

Data files for PM10 air concentrations and elemental speciation of the PM10 mass by XRF analysis were obtained from the MPCA for the 30 sites included in the Statewide Air Toxics Monitoring Study (1996-2001). The two key parameters to be obtained from the particulate filters were calcium and phosphorus concentrations. Calcium concentrations were typically available for each sampling period. However, upon review of the individual site data files, phosphorus concentrations were not available, so an alternative method for deriving phosphorus concentrations for the particle filters was employed for this analysis. This alternative method assumes that the relationship between phosphorus and calcium in precipitation is transferable to the particulate filter data (i.e., the same material being washed out in the precipitation is the same material being dry deposited and collected on the particulate filters). The critical assumptions and the details of calculating phosphorus air concentrations from the particulate filter data is further described in Appendix E.

2.2.2.2.1 Dry Deposition
The following steps were taken to estimate the areal phosphorus deposition rate from dry fallout:

1. Establishing the relationship between phosphorus and calcium on particle filters.

    a. The relationship of phosphorus and calcium on the particle filters is assumed to be the same as the relationship of phosphorus and calcium in precipitation; the soil dust being washed out in precipitation is the same dust being dry deposited and collected on the PM10 filters.

    b. The best source of phosphorus and calcium in precipitation data is the special study conducted by the St. Croix Watershed Research Station. The total phosphorus and

calcium concentrations (hereafter denoted as total [P]) and total [Ca] in precipitation data) were determined from August 1991 – September 2001 at 4 sites: Fernberg (Ely), Marcell, Camp Ripley, Lamberton; referred to as "reference sites".

c. The relationship on a sample-by-sample basis (milligrams per square meter; mg/m^2) between total [P] and total [Ca] in precipitation at the 4 reference sites was established through regression analysis:

$$y = 0.0289x \quad \text{(through zero)} \quad (R^2 = 0.42)$$

Where: y = Total phosphorus in micrograms per square meter ($\mu g/m^2$)

x = Total calcium in $\mu g/m^2$.

2. Extrapolating the relationship of [P] and [Ca] from precipitation to the particulate filters.

a. Since the regression equation for [P] and [Ca] in precipitation goes through zero, this regression equation can be applied to data from other media under the assumption that the ratio is the same (i.e., particulate filter data) without having to convert units. Essentially forcing the regression equation through zero creates a ratio of [P] to [Ca] that can be applied to other data.

b. In this regard, the regression equation from above can be modified as follows for application to the particle filter data.

$$y = 0.0289x \quad \text{(through zero)} \quad (R^2 = 0.42)$$

Where: y = Total phosphorus in micrograms per square meter cubic meter ($\mu g/m^3$)

x = Total calcium in $\mu g/m^3$.

3. Estimating [P] in air at the MPCA's air monitoring locations.

a. The regression equation from 2.b. was then used to estimate [P] in ambient air at the MPCA air monitoring sites. Annual [Ca] concentrations in micrograms per cubic meter were calculated for each monitoring site based on the individual sample [Ca] concentrations. The annual average [Ca] in air is then used in the regression equation to derive an estimate of annual average [P] in air.

4. Calculating dry phosphorus deposition

a. Monitoring sites locations were mapped with respect to basin boundaries:

Cedar River: Albert Lea

Des Moines River: Pipestone

Lake Superior: Virginia (2 sites), Duluth (2), Silver Bay, Hibbing

Minnesota River: North Mankato, Brandon Township, Granite Falls, Willmar, Swift County

Mississippi (Upper)   St. Paul (3), Minneapolis (3), Bemidji, Elk River, Fort Ripley, Alexandria, Hutchinson, St. Cloud, St. Michael, Grand Rapids, Little Falls

Mississippi (Lower): Rochester, Goodhue County, Apple Valley, Winona

Missouri River: Pipestone

Rainy River: Warroad, International Falls

Red River: Fergus Falls, Moorhead, Perham

St. Croix River: West Lakeland, Pine County (Sandstone)

b. Calculation components for phosphorus deposition in a basin:

- Estimated phosphorus air concentration; if more than one site assigned to a basin then the average phosphorus in air concentration used in the deposition calculation.

- The estimated phosphorus air concentration (or the average phosphorus air concentration if more than one site is in a basin) is to be split into two size fractions based on MPCA collocated PM10 and PM2.5 samplers (average from 5 sites):

    42% fine fraction (< 2.5 microns)

    58% coarse fraction

- A deposition velocity for each particle size fraction was estimated based on the information from Meyers (2003):

    Fine fraction deposition velocity = 0.5 centimeters per second (cm/s);

    Coarse fraction deposition velocity = 3 cm/s.

- The coarse and fine particle deposition is summed together to provide a "total" particle deposition estimate.

- Conversion factors: convert seconds to years, cm to meters, and $\mu g/m^3$ to kg/ha.

The reader should note that for the dry deposition estimate, no adjustments were made in the estimation of dry deposition in a dry or a wet year; data are not available at this time to derive estimates of dry deposition during different precipitation regimes. The dry deposition rates were applied to area estimates of surface waters (open water + wetland as designated in USGS NLCD GIS coverage) in each basin.

2.2.2.2.2 Wet Deposition

The following steps were taken to estimate the areal phosphorus deposition rate from wet deposition:

1. Establishing the relationship between phosphorus and calcium in precipitation.

    a. NADP routinely analyzes rain samples for pH, alkalinity, major cations (including calcium and potassium) and major anions (including sulfate, nitrate). Since calcium concentrations are available for all samples that were analyzed, and calcium is a signature for soil contributions, the relationship between phosphorus and calcium would need to be established. The use of NADP data also provides some consistency in the data used for estimating wet phosphorus deposition.

    b. The best source of phosphorus in precipitation data is the special study conducted by the St. Croix Watershed Research Station. The total phosphorus concentrations (hereafter denoted as total [P]) in precipitation data) determined from August 1991 – September 2001 at 4 sites: Fernberg (Ely), Marcell, Camp Ripley, Lamberton; referred to as "reference sites". The special study also provided measurements on total [Ca] in precipitation.

    c. An initial analysis identified that the total [Ca] from the special study was approximately two times greater than the [Ca] reported by NADP for the same time period. The NADP does not acidify samples; therefore the NADP reports dissolved [Ca]. To compensate for NADP reporting dissolved [Ca], and to provide the best estimate of [P] in precipitation from the auxiliary (NADP) sites, it was determined that the relationship between [P] and [Ca] in precipitation should be determined by using the total [P] concentrations from the special study conducted by the St. Croix Watershed Research Station and the dissolved [Ca] reported by NADP for these same "reference" sites.

    d. The volume-weighted relationship on a sample-by-sample basis between total [P] in precipitation and dissolved [Ca] in precipitation from NADP at these same reference sites (collocated sampling occurred) was established by MPCA staff (Dr. Ed Swain, 2003) through regression analysis:

$$y = 0.0671x - 0.4586 \ (R^2 = 0.47)$$

Where: y = Total phosphorus in micrograms per liter (µg/L)

x = NADP calcium (dissolved) in µg/L.

2. Extrapolating the relationship of [P] and [Ca] in precipitation to other locations.

    a. The regression analysis based on total [P] and dissolved [Ca] concentrations for the reference sites was then used to estimate [P] in precipitation at other NADP monitoring sites (referred to as "auxiliary sites"). Annual volume-weighted [Ca] in precipitation data (annual volume weighted average) were obtained for the auxiliary sites from NADP and the regression equation from above was then used to estimate total [P] in precipitation for each auxiliary site.

b. The auxiliary monitoring sites will supplement the information from the reference sites in calculating wet phosphorus deposition to specific basins.

3. Calculating wet phosphorus deposition

a. Monitoring sites locations were mapped with respect to basin boundaries and assignments to watershed made based on site locations:

> Cedar River: Lamberton
> Des Moines River: Lamberton
> Lake Superior: Hovland, Wolf Ridge, Fond du Lac
> Minnesota River: Lamberton
> Mississippi (Upper): Marcell, Camp Ripley, Cedar Creek
> Mississippi (Lower): Wildcat Mountain
> Missouri River: Lamberton
> Rainy River: Voyageurs Nat. Park, Marcell, Fernberg
> Red River: Icelandic State Park
> St. Croix River: Grindstone Lake, Cedar Creek

b. Calculation components for phosphorus deposition in a basin:

o   Annual average precipitation for the basin (obtained from State Climatology Office)

o   [P] in precipitation (annual, volume weighted average; measured at one of the reference sites or estimated for one of the auxiliary sites; if more than one site assigned to a basin then the average [P] in precipitation used in the deposition calculation)

o   Area estimate (hectares or acres) of open surface water (surface water + wetland as designated in GIS) in a basin.

2.2.2.3   Deicing Agents

The use of deicing chemicals has increased in the U.S. since the 1940s and 1950s to provide "bare pavement" for safe and efficient winter transportation. As more and more transportation agencies adopted the "bare pavement" policy, the use of salt, salt and sand mixtures, liquid brines and alternative deicers increased with the need to maintain this standard for pavement conditions during inclement weather. Other road agencies in Minnesota such as cities, townships and counties use deicing agents to maintain a similar standard for pavement conditions during inclement weather. The search for alternatives to salt for road deicing has been prompted primarily due to the infrastructure corrosion concerns and the impacts of chloride on water quality and vegetation.   Recently, some limited research has documented water quality concerns related to phosphorus and other chemicals present in deicing agents, as well as the alternative compounds. This section provides a general

discussion about the methodology used to quantify the amount of phosphorus entering surface waters from this source category. A more detailed discussion of the methodology used for this analysis is included in Appendix F. The results of the phosphorus loading computations for this source are discussed in Section 3.3.4.

Review of the existing scientific literature with regard to deicing agents as a phosphorus source was concerned with three major areas; 1) usage patterns of deicing agents in Minnesota and other states with regard to road types and road management agency, 2) the phosphorus content of deicing agents – salt, sand, and deicing alternatives, and 3) the impact of weather patterns on usage levels.

Phosphorus loading computations were primarily based upon the MnDOT data sources as this was the most detailed data set and extended over the longest time period. Loading calculations for TCMA counties were obtained from published data and other road types were extrapolated using the MnDOT data trends, applications rates and deicing mixtures. The MnDOT database was the most comprehensive and most useful in determining application rates across the range of conditions for wet, dry and average years. The applications rates for each MnDOT District, and thus for each basin, is based upon the use of statewide averages based upon their relationship to snowfall amounts over a winter. Application rates for salt and sand were then adjusted to account for the wet, dry and average conditions based upon the ratios derived from the 1971 – 2003 time period and the relationship between the years of detailed information provided in the Salt Solutions Report and MnDOT's Work Management System Reports (SRF Consulting Group, 1998; MnDOT, 2003). The use of brine for deicing has increased in recent years, but the period of record for its application is limited and thus 2002 rates were used in the calculations as insufficient data was available to attempt to adjust for year-to-year variability in its application rate.

MnDOT's road classes (service levels) were used to further define the application assumptions for the mix ratios of deicers used on the three road types maintained by MnDOT. Based upon an examination of the 2003 – 02 deicer usage report the total salt plus sand application, in tons per lane mile, was modified for the three types of roads maintained by MnDOT (MnDOT, 2003).

01 - Interstate Trunk Highway – uses a 100% salt assumption (assuming "super commuter" service level)

02 - U.S. Trunk Highway – uses a 70% salt assumption (assuming "urban commuter" service level)

03 - Minnesota Trunk Highway – uses a 50% salt assumption (assuming "rural commuter"
service level)

County and local road agency specific data was less readily available for use in this analysis, except for the TCMA counties. An analysis was undertaken using the 1994 – 1997 data available for the TCMA to develop usage rates for the County State Aid Highway (CSAH) system. The TCMA deicer usage rates were summarized based upon average conditions (1994 – 95) for both salt and sand usage on a lane mile basis. The 1995 – 1997 period was used for calculation of the wet year conditions. The dry year conditions were used based upon the 90th percentile summary statistics. These usage numbers were applied to all CSAH miles across the state as they were viewed as the more heavily traveled and thus more highly maintained roads in both the TCMA and out-state areas. Deicer usage rates for other county highways and local roads were developed based upon an even smaller database of actual usage rates. As such, the usage rates for the "rural" counties in the TCMA – Scott, Carver and Chisago counties – were used to develop usage rates for other roads included in this analysis. An analysis was undertaken using the 1994 – 1997 data available for these TCMA in manner consistent with the CSAH analysis described above.

As the concern over and documentation of the environmental impacts of deicing agents has increased, a number of authors and agencies have attempted to document the concentrations of other elements or compounds of concern that are introduced into the environment through road deicing. This analysis summarized and utilized the phosphorus concentrations from these analyses of the various deicers.

As a review of existing literature was undertaken it became obvious that the application rates and mixtures of deicers used are strongly predicated by weather conditions. An examination of the MnDOT records indicated that the number of "events" per season appeared to be the driving factor in the quantities of material applied. The high variability in the number of events between regions of the state in any given year, as well the year-to-year variability in the number of events precluded the use of events in this analysis. The MnDOT application guidelines provided some insight into how the variations in weather patterns impacted usage levels by counties and local units of government. Based upon an assessment of the snow data and usage levels provided by MnDOT for the period of 1971 to 2003 the amount of winter snow was used as a surrogate for the number of events. The winter snow fall amount at MSP Airport was used to define average, dry (low snowfall – 90th percentile) and wet (10th percentile) conditions.

2.2.2.4   Streambank Erosion

The stability of stream channels is a complex issue that is highly influenced by the dynamics of natural and anthropogenic disturbances. The banks of unstable streams typically undergo erosion, both in the form of particle detachment from hydrodynamic drag and mass failure following erosion of the bank toe. The phosphorus attached to eroded streambank material is immediately delivered to the receiving water where it may ultimately become available for biologic uptake, re-deposited downstream, or transported with the flow out of the system. This section provides a general discussion about the methodology used to quantify the amount of phosphorus entering surface waters from streambank erosion. A more detailed discussion of the methodology used for this analysis is included in Appendix G. The results of the phosphorus loading computations for this source are discussed in Section 3.3.5.

Simon and Hupp (1986) developed a six-stage, semi-quantitative model of channel evolution in disturbed channels, for bed-level trends, that qualitatively recognizes bank slope development. The third and fourth stages represent stream degradation, characterized by the lowering of the channel bed and basal erosion, with a subsequent increase in bank heights and slopes, leading to mass-wasting from slab, pop-out and deep-seated rotational failure.

Several researchers have determined that the stream sediment load is proportional to stream discharge (Lane, 1955; Glysson, 1987; Tornes, 1986; Kuhnle and Simon, 2000; Syvitski et al., 2000). Instantaneous flow and sediment transport data are used to develop sediment-transport rating curves, which are typically based on logarithmic regression relationships. A steep regressed slope to the rating relationship indicates both high sediment availability and high transport capacity. The slope of the suspended-sediment rating relationship varies (Simon, 1989a; Simon et al., 2003), depending upon the stage of channel evolution. Simon (1989a) determined that the highest slope of the suspended-sediment rating relationship corresponds to the stream stages (III and IV) that are undergoing the highest degree of degradation. Migration of knickpoints (or vertical step-changes in bed surface elevation) up tributary streams during Stage III, and bank failures by mass wasting during Stage IV, both serve to significantly increase sediment yield (Simon, 1989a). For re-stabilized streams (Stage VI), the slope of the suspended-sediment rating relation is approximately 1.5, as opposed to 1.0 for "natural" streams (Stage I).

The approach used to assess this source of phosphorus utilized the data and techniques from the available literature to estimate total phosphorus loadings to the surface waters within each of the ten major basins in Minnesota. The literature search and review of available monitoring data involved a

compilation of streambank erosion studies completed within Minnesota, along with an evaluation of the literature pertaining to sediment yield from Minnesota watersheds, to define the contribution of streambank erosion to the total phosphorus budget. Wherever possible, streambank erosion studies completed for Minnesota streams were used to determine the phosphorus load under low, average and high flow conditions for the respective basins. Sediment yield literature specific to the various regions of the state was consulted to develop an approach and assist with the assessment of the remaining unstudied watersheds.

Five published studies were found that specifically addressed streambank erosion for streams that originate in Minnesota. Wherever possible, average annual streambank sediment erosion, average annual erosion per stream mile, slope of suspended sediment rating relation, sediment erosion as a percentage of observed downstream suspended solids loading, and EPA Level III Ecoregion were expressed for each stream studied. Most of the estimates of streambank sediment erosion were the result of stream channel surveys (including aerial photos) to evaluate streambank retreat (or migration) and eroding bank area to determine the average annual volume of material eroded. One study (Sekely et al., 2002) also produced a probability plot of annual streambank erosion rates.

In addition to the streambank sediment erosion studies, two regional studies have been completed involving sediment yield data for Minnesota watersheds (Tornes, 1986; Simon et al., 2003). Tornes (1986) analyzed the average annual sediment yield data for 33 USGS gaging stations, in or adjacent to Minnesota, while Simon et al. (2003) determined sediment yield, on the basis of the 1.5-year recurrence interval flow rate, for each of the EPA Level III Ecoregions. Tornes (1986) determined the average annual sediment yield for each of the gaging stations by developing sediment-transport curves for each of the stations and applying the relationships to flow-duration curves to calculate and sum the sediment loadings at each interval. Simon et al. (2003) determined sediment yield quartiles, minimum, and maximum yields, on the basis of the 1.5-year recurrence interval (or effective discharge) flow rate, for each of the EPA Level III Ecoregions.

The approach for determining phosphorus loading from streambank erosion generally involved the following steps:

- Convert published streambank erosion estimates into average annual sediment yield

- Using the published sediment-transport curves from Tornes (1986), determine the relationship between average annual sediment yield and the slope of the sediment-transport curve segment containing the 1.5-year recurrence interval flow rate, as a surrogate for the effective discharge

- Apply average annual sediment yields from published streambank erosion estimates and Tornes (1986) to respective watershed units in GIS and determine average annual area-weighted monitored sediment yield for each of the EPA Level III Ecoregions in Minnesota

- Compare average annual monitored sediment yield for each of the EPA Level III Ecoregions in Minnesota to the effective discharge rate sediment yields published by Simon et al. (2003) for the same ecoregions and make adjustments, if necessary

- Apply average annual sediment yield for each of the EPA Level III Ecoregions to the unmonitored portions of the state and estimate streambank sediment erosion component based on difference between average annual sediment yield for ecoregion and estimated annual sediment yield for stable (Stage VI) stream, with slope of suspended sediment rating relation equal to 1.5 (per Simon, 1989a)

- Estimate annual streambank sediment erosion for all watersheds under low and high flow conditions, based on the probability plot relationship (taken from Sekely et al., 2002) of annual streambank erosion rates

- Combine the streambank erosion sediment loadings associated with each watershed with the average soil test phosphorus concentration (based on 16 surface samples collected from Blue Earth River escarpments, as described in Sekely et al., 2002) to calculate the total phosphorus load associated with sediment loading estimated from streambank erosion in each basin for each flow condition

2.2.2.5   Individual Sewage Treatment Systems/Unsewered Communities

"Undersewered" areas are communities or residential areas which have a crude sewage collection system with little or no treatment component and/or have individual systems which are non-conforming. Individual sewage treatment system (ISTS) refers to a sewage treatment and disposal system located on a property, using subsurface soil treatment and disposal for an individual home or establishment. MPCA (2002a) states that most undersewered communities and many failing septic systems outside of undersewered areas have relatively direct connections to surface waters through tiles lines and road ditches, resulting in a very high delivery potential. "Failing" ISTS are specifically defined as systems that are failing to protect groundwater from contamination, while those systems which discharge partially treated sewage to the ground surface, road ditches, tile lines, and directly into streams, rivers and lakes are considered an imminent threat to public health and safety (ITPHS). This section provides a general discussion about the methodology used to quantify the amount of phosphorus

entering surface waters from the ISTS/unsewered communities source category. A more detailed discussion of the methodology used for this analysis is included in Appendix H. The results of the phosphorus loading computations for this source are discussed in Section 3.3.6.

The conventional ISTS consists primarily of a septic tank and a soil absorption field. Septic tanks remove most settleable and floatable material and function as an anaerobic bioreactor that promotes partial digestion of retained organic matter (EPA, 2002). Septic tank effluent, which contains significant concentrations of pathogens and nutrients, has traditionally been discharged to soil, sand, or other media absorption fields for further treatment through biological processes, adsorption, filtration, and infiltration into underlying soils which are suitable for treatment and disposal. Phosphorus is present in significant concentrations in most wastewaters treated by ISTS. Monitoring below ISTS systems has shown that the amount of phosphorus leached to groundwater below an operating ISTS depends on several factors: the characteristics of the soil, the thickness of the unsaturated zone through which the wastewater percolates, the applied loading rate, and the age of the system (EPA, 2002). The amount of phosphorus in ground water varies from background concentrations to concentrations comparable to that of septic tank effluent. Phosphorus export to surface waters from ISTS and unsewered communities is dependent on the following factors:

- Phosphorus content of waste load

- Population served by ISTS or unsewered communities

- Compliance of treatment systems with performance standards

- Characteristics of soil absorption field, groundwater conditions and proximity to surface waters

Data pertaining to the phosphorus content of the untreated waste load from unsewered communities was addressed in the Point Sources Technical Memorandum (Appendix B), prepared for this project. For the purposes of this analysis, the phosphorus contained in untreated sewage discharge from non-conforming ISTS or unsewered communities consists of the following sources, with the corresponding per capita loadings of phosphorus (see Appendices B and H):

Source	Phosphorus Load (kg/cap/yr)
Automatic dishwasher detergent	0.1250
Dentifrices	0.0115
Food soils and garbage disposal wastes	0.1895
Ingested Human wastes	0.5585
Total	0.8845

Dentifrices include toothpaste and other dental care products. Food soils include waste food and beverages poured down the sink, and food washed down the drain as a result of dish rinsing and washing. The total per capita phosphorus load of 0.8845 kg/yr (1.946 lbs/cap/yr), was assumed to apply to the population served by ISTS or unsewered communities throughout the state.

The number of people served by ISTS was estimated from a variety of data sources. Two of the data sources were spreadsheets provided by the Minnesota Pollution Control Agency, another was the 1990 Census (United States Census Bureau, 1990), and the last was estimated based on the POTW population served from the Point Sources Technical Memorandum (Appendix B). This last method using the difference between the 2000 Census (United States Census Bureau, 2000) population and the POTW population served were used in the study to estimate phosphorus loadings from ISTS. This data showed good consistency with the other data available for ISTS in Minnesota. By using the third method, a total accounting of domestic waste disposal is provided in this study.

The MPCA developed a spreadsheet, updated in September, 2003, providing a list of unsewered communities within Minnesota (MPCA, 2003). The major basin for each of these communities was estimated by assigning an approximate geographic location based on a city, township, lake/county, or township-range-section location (whichever provided the most detailed location).

The Minnesota River basin had a significant number of households served by sewage treatment systems that involved direct discharge to a tile drain line (Tetra Tech, 2002). The majority of these systems, referred to as direct-to-tile ISTS, include a septic tank with no other treatment. Assuming that most of the direct-to-tile ISTS are located in rural areas with tile lines, Tetra Tech (2002) extracted data from the Minnesota River Assessment Project, or MRAP (MPCA, 1994), to develop a

relationship between the number of direct-to-tile ISTS and cropland. The ISTS densities and cropland were then mapped by minor watersheds across the Minnesota River basin. The geographic trend in density was assumed to be consistent with the MRAP designations for three nutrient source regions, and the average density of direct-to-tile ISTS per 10,000 acres of cropland was determined for each source region. For this analysis, the assumptions about direct-to-tile ISTS density per 10,000 acres of cropland for each source region were retained for the Minnesota River basin. Since no assessments of direct-to-tile ISTS had been published for any other basins in Minnesota, several of the minor watersheds in surrounding basins were assumed to have direct-to-tile ISTS densities comparable to the three Source Regions, based on knowledge of the presence of drain tiles, cropland and their proximity to the MRAP study areas. The amount of cropland and area of each Source Region was determined and multiplied to determine the total number of direct-to-tile systems for each basin. The population served by direct-to-tile ISTS was estimated by multiplying the number of systems by the average household size for each basin.

The MPCA maintained a spreadsheet with the number of ISTS by local units of governments (LUG) with ISTS ordinances in 2002 (MPCA, 2002). Included in the spreadsheet was the LUG name and type (e.g. city, township or county). An estimate of the number of full time and seasonal residences served by ISTS was included in the spreadsheet. There was also an estimate of the number of systems failing to protect groundwater and an estimate for the number of systems which are considered an ITPHS. The population served was estimated by multiplying the number of full time residences by the population per household values (for the 2000 census) for the LUG's respective county.

Based on the availability of data and the potential for variation in phosphorus export from undersewered communities and the various types of conforming and nonconforming ISTS, phosphorus loadings were estimated for each of the following source categories:

- Unsewered communities

- Direct-to-tile ISTS

- Conforming and nonconforming seasonal ISTS

- Remaining conforming and nonconforming ISTS

The populations associated with unsewered communities and direct-to-tile ISTS in each basin were assumed to receive treatment from septic tanks before discharging to surface waters. The number of

seasonal residences had also been estimated in the MPCA ISTS LUG spreadsheet (MPCA, 2002). Since no data was available for the population served by seasonal ISTS, a household size of 2.1 was assumed and applied to the number of seasonal residences in each basin. No literature was found, so it was assumed that seasonal residences are occupied for four months each year. It was further assumed that, since seasonal residences are typically located in close proximity to surface waters, nonconforming ISTS (both failing and ITPHS) would contribute all of the 43 percent of phosphorus passing through a septic tank to surface waters. Conforming seasonal ISTS were assumed to remove 80 percent of the total phosphorus loading, due to treatment from the septic tank and soil absorption field, before discharging to surface waters in each basin.

Since most of the permanent residences are not typically located as close in proximity to surface waters as seasonal residences, it was assumed that both fully conforming and failing ISTS would provide higher phosphorus attenuation for permanent residences than what was assumed for seasonal residences. Conforming ISTS were assumed to remove 90 percent of the overall total phosphorus loading, while failing ISTS were assumed to remove 70 percent of the overall total phosphorus loading, before discharging to surface waters in each basin. The nonconforming ISTS, considered an ITPHS, were assumed to be contributing all of the 43 percent of phosphorus passing through a septic tank to surface waters.

2.2.2.6   Non-Agricultural Rural Runoff

Section 2.2.2.1 discusses the methods used to estimate the phosphorus loadings associated runoff from agricultural lands, while Section 2.2.2.7 describes the methodology used to quantify the amount of phosphorus in runoff from urban land cover types. This section provides a general discussion about the methodology used to quantify the amount of phosphorus entering surface waters in runoff from unincorporated areas that are not considered agricultural land cover types (referred to as non-agricultural rural). The major natural land cover types included in the non-agricultural rural land use group are forests (coniferous, deciduous and mixed), grasslands and shrublands. Rural residential areas, transportation infrastructure, and other typically urban land uses such as residential and commercial developed areas outside the boundaries of incorporated urban areas are also included in this assessment. A more detailed discussion of the methodology used for this analysis is included in Appendix I. The results of the phosphorus loading computations for this source are discussed in Section 3.3.7.

Within some of the major basins of Minnesota, forests and grasslands still cover up to 60% of the watershed area. The hydrologic cycling of annual precipitation in natural vegetation moves most of

the water to infiltration and thus promotes stable stream base flows and reduces surface runoff. In natural plant communities, much of the phosphorus pool is retained within the plant community and the soil profile, with plant biomass creation, senescence and subsequent decomposition processes cycling nutrients back into the soil profile. The high soil infiltration rates in these plant communities lead to low surface runoff rates and little soil loss via erosion, and thus low rates of nutrient export to surface waters. In most cases the surface runoff rates are less than 10% of the annual precipitation for these plant communities and phosphorus export rates are below 0.169 kilograms of phosphorus per hectare per year (0.151 pounds per acre per year).

The scientific literature was reviewed to determine the hydrologic regimes, nutrient cycling mechanisms and phosphorus loading factors for each of the land cover types included in the Non-Agricultural Rural Runoff category. The hydrologic and nutrient export relationships examined for the rural land cover types are generally discussed in this section, while the hydrologic and nutrient export relationships for rural residential and commercial/industrial/transportation land cover types are discussed in Section 2.2.2.7.

Interception of rainfall occurs at multiple levels within the forest – tree canopy, tree and shrub layer stems, shrub canopy, herbaceous layer and ground litter – to reduce overland flows (Brooks, et al, 2003; Verry 1976). Other authors have reported little or no overland flow from intact deciduous or coniferous forests due to interception (Binkley, 2001; Knighton and Steigler, 1980; Metcalfe and Butle, 1999; Verry, 1969).

While a fair amount of literature exists on forest hydrology and nutrients, comparable literature for shrublands and grasslands is much less extensive. Many authors suggest that runoff rates and nutrient exports form these communities are low, however the supporting evidence is limited. Brye, et al. (2000) and Brye, et al. (2002) evaluated the water and phosphorus budgets of a restored prairie near Madison WI. The authors reported that rainfall interception by plant residue was a significant component of the annual water budget (nearly 70%). Higher soil storage and ET rates led to lower soil drainage and runoff volumes. Runoff volumes were 11% to 18% of the water budget, with a mean of 14.5% for the test plots. Snowmelt was responsible for nearly all of the runoff volumes. Timmons and Holt (1977) reported that phosphorus losses from grasslands to be in a range of 0.100 kg P/ha/yr to 0.250 kg P/ha/yr, with a phosphorus concentration in runoff of 0.200 mg P/L. Using the water budget data from Brye, et al (2000) and Brye, et al (2002) and phosphorus concentration data from Timmons and Holt (1977), an export loading rate of 0.169 kg P/ha/yr for ecoregion VIII was calculated. Using the water budget information from Winter and Carr (1980), Winter, et al,

(2001), Winter, Rosenberry (1995 and 1998) and Shjeflo (1968) and concentration data from USACE (2001), a phosphorus export of 0.060 kg P/ha/yr was calculated for ecoregion VI. Data from Olness, et al (1988) and Menzel, et al (1978) provided an export rate 0.175 kg P/ha/yr for grassland pasture.

A search of the literature provided no reported shrubland phosphorus export rates (Holechek, et al, 1977; Dodds, et al, 1996: Burke, et al, 1990). Most shrublands are composed of a herbaceous layer of grasses and forbs with a sparse over story of trees and/or low shrubs. MN DNR (1993) and Leach and Givnish (1999) suggest that many of the hydrologic and ecologic attributes of forest and prairie communities are present in shrublands. Low runoff rates, high annual evapotranspiration and limited nutrient losses of the two shrubland community components provide a basis to conclude that shrublands are intermediate with regard to phosphorus export. Based upon these assumptions, the nutrient export rate for shrubland was determined from the average of the grassland and deciduous forest communities. The calculated value used for this assessment is 0.129 kg P/ha/yr.

This investigation of phosphorus loadings from non-agricultural rural land uses draws upon ecoregion-based loading and export rates for phosphorus in Minnesota. The use of ecoregions allows the similarities in underlying ecological conditions to be aggregated across basin boundaries and state boundaries to develop accurate estimates of loadings. Ecoregions are defined as regions of relative homogeneity in ecological systems, such that geographic characteristics such as soils, vegetation, climate, geology, and land cover are relatively similar within the bounds of each ecoregion (Omernik, 2000). The US EPA has developed generalized "nutrient Ecoregions" that are aggregations of the Level III Ecoregions (EPA 2000d, EPA 2000e). Within Minnesota there are three EPA Level III Aggregate Ecoregions (shown in Figure 2, Appendix I). As the number of phosphorus export studies completed in Minnesota is relatively small, the use of export rates from the larger Level III aggregate regions provides a wider data set that can be extrapolated across the basins (MPCA, 2003).

The Corn Belt and Northern Great Plains – Aggregate Ecoregion VI – is comprised of rolling plains and flat lake beds, dominated by extensive, highly productive cropland (EPA, 2000a). Nutrient-rich soils significantly influence surface and subsurface water quality and high concentrations of nitrate and phosphorus cause water quality problems in many basins. The Mostly Glaciated Dairy Region – Aggregate Ecoregion VII – is dominated by forests, dairy operations, and livestock farming (EPA, 2000b). This ecoregion was mostly glaciated and includes flat lake plains, rolling till plains, hummocky stagnation moraines, hills, and low mountains. The Nutrient Poor Largely Glaciated Upper Midwest and Northeast – Aggregate Ecoregion VIII – is characterized by extensive forests,

nutrient-poor soils, a short growing season, limited cropland, and many marshes, swamps, lakes, and streams.

An assessment was completed on the literature values for phosphorus export rates to examine any differences between the three aggregate level ecoregions. The literature data was statistically summarized, where available, and the ecoregion mean value was determined for each plant community. These values were used for the phosphorus load calculations.

For the purposes of defining and quantifying the phosphorus loads to Minnesota basins, the non-agricultural rural land uses within these three Aggregate Ecoregions were classified and enumerated using the USGS National Land Cover Data (NLCD). The National Land Cover Data Set for the Conterminous United States is derived from the Landsat thematic mapper data system (Vogelmann, 2001). The NLCD cover classes included in the non-agricultural rural category include the following:

- ◆ Unincorporated Urban Areas
  - ○ Low intensity residential (outside incorporated urban areas)
  - ○ High intensity residential (outside incorporated urban areas)
  - ○ Commercial/Industrial/Transportation (outside incorporated urban areas)
- ◆ Deciduous Forest
- ◆ Evergreen Forest
- ◆ Mixed Forest
- ◆ Shrubland
- ◆ Grasslands/Herbaceous
- ◆ Urban / Recreational Grasses
- ◆ Other
  - ○ Quarries/Strip Mines/Gravel Pits
  - ○ Transitional

The development of nutrient loading estimates in the absence of direct monitoring has generally been completed by applying areal based nutrient export rates to the watershed area to calculate the annual nutrient mass (Beaulac and Reckhow, 1982: Reckhow, et al, 1980; Panuska and Lillie, 1995; Clesceri, et al, 1986a; Clesceri, et al, 1986b; McFarland and Hauck, 2001). Phosphorus export coefficients assume 100% of the land transports phosphorus that will reach surface waters. The phosphorus export coefficient is part of the total phosphorus loading equation:

$$L = \sum_{i=1}^{m} c_i \bullet A_i$$

L is total phosphorus loading from land (in kilograms per year), m is number of land use types, c_i is the phosphorus export coefficient for land use i (in kilograms per hectare per year), and A_i is area of land use i (in hectares).

Over large watershed areas, the phosphorus export is not proportional to watershed area and some attenuation of phosphorus occurs, especially in natural vegetation that have low runoff rates. Recently, authors who have examined the nutrient export issue on landscape level scales (large watersheds and higher order streams) have raised concerns over the applicability of export coefficients across large watershed areas (Birr and Mulla, 2001; Cammermeyer, et al, 1999; Johnson and gage, 1997; Jones, et al, 2001; Mattson and Isaac, 1999; McFarland and Hauck, 1998; Richards, et al, 2001; Sharpley, et al, 1993; Soranno, et al, 1996; Worrall and Burt, 1999). The underlying issue related to this concern is that not all areas in a large watershed contribute nutrients and sediment equally. Novotny and Chester (1989) showed that the sediment delivery rate decreases with increasing watershed size. They report that in humid regions only a portion of a watershed contributes to surface runoff; they called these contributory areas of a watershed the "hydrologically active areas". Soranno, et al. (1996) and Cammermeyer, et al, (1999) suggest two adjustments to account for the attenuation by including a transmission coefficient (T) that represents the proportion of phosphorus transported down slope along the path of overland flow and a phosphorus flux coefficient (f_i), that represents the phosphorus production and transport that reaches a surface water body. While this equation applies more strictly to watershed modeling with GIS software, the underlying premises apply directly to the loading assessment methodology used here. The authors suggest that the phosphorus loading equation can be modified:

$$L = \sum_{i=1}^{m} \sum_{p=1}^{n} f_i \bullet A_{p,i} \bullet T_i^{p}$$

T is the transmission coefficient ($O<T<1$) representing the proportion of phosphorus transported, f_i is the phosphorus flux coefficient, n is the number of pixels, and p is the pixel distance of overland flow.

Soranno, et al (1996) reported that the greatest contribution of loadings was derived from land uses within the riparian corridor, a corridor that varies in width depending upon topography and runoff conditions. Based upon modeling of monitored watersheds they found that the total annual rainfall

affected the phosphorus loading by creating variability as to the effective contributory area. In most cases, the transmission coefficient is determined through GIS modeling of the watershed area. The GIS-based development of transmission coefficients for use in this assessment was beyond the scope of the project. In the absence of a calculated T, an estimate of the contributory area of a watershed based upon land use and the application of a basin runoff factors were chosen for the load calculations. The basin runoff factor accounts for the differences in effective flow length and thus runoff volumes between the three precipitation scenarios (Soranno et al, 1996; Cammermeyer, et al, 1999; Barr Engineering, 2003b). The phosphorus loading estimation methodology used in this assessment assumes that c_i will be equal to f_i through the use of calculated loadings from the 100 meter contributory areas only.

The phenomenon of contributory area and variability in nutrient mass over a range of flow scenarios is a central question to the estimation of large basin loads. The literature was reviewed for a consensus on the size of this contributory area and the impact of hydrologic conditions upon the size and export estimation. Novotny and Chester (1989) calibrated and verified hydrologic models for a number of Milwaukee area basins and found that sediment delivery ratios ranged from 0.01 for pervious areas and 1.0 for completely storm-sewered urban areas. Johnson, et al (1997) found that landscape factors within the 100 meter ecotone adjacent to streams were sufficient predictors of stream water chemistry. Tufford, *et al*, (1998) reported that the land within 150 meters of streams was a better predictor of nutrient concentrations. Many authors have suggested that riparian land cover within 100 meters can mediate upslope impacts on water quality (Schmitt, et al, 1999; Cole et al, 1997; Castelle, et al, 1994; Roth, et al, 1996; Osborne and Kovacic, 1993).

Based upon the literature review conclusion that the 100 meter riparian zone has the greatest influence on water chemistry, we have chosen to estimate phosphorus loads from the 100 meter zone of land use immediately adjacent to perennial streams, lakes and wetlands in all of the basins. It should be noted that throughout most of Minnesota, it is believed that the risks of phosphorus transport to surface waters are greatest in the contributing corridor within about 100 m from surface waterbodies. Due to topographic variations along surface waterbodies, in some areas phosphorus contributions from overland runoff and erosion may occur from as far away as several hundreds of meters. In contrast, where berms are present along waterbodies it may be unlikely for a significant amount of surface runoff or erosion to enter surface water. Thus, the 100 m contributing corridor should be viewed as a regional average for contributions of phosphorus to surface waters from runoff and erosion on adjacent lands.

The NLCD land use coverage for the non-agricultural rural was determined using ArcView to create land cover quantities for all lands within 100 meters of all surface waters (as defined in Section 1.4.1). This 100 meter wide area was used for the calculation of the effective contributory area for each land cover types for each basin.

The phosphorus load for each land use was calculated by multiplying the phosphorus export coefficient by the 100 m contributory area and basin runoff factor for each land use category. The basin runoff factor is based upon the percent differences between runoff in the wet and dry precipitation scenarios compared to the average conditions for each basin. This information was generated from the calculation of runoff volumes as part of the basin hydrology (discussed in Sections 2.1 and 3.1). Use of the basin runoff factor and contributory watershed area for loading calculations, allowed for the following adjustment of the loadings based upon the annual runoff:

*Basin natural area load (kg) = Export rate (kg/ha/yr) * Contributory area (ha) * Basin runoff factor*

2.2.2.7 Urban Runoff

The conversion of land areas to urban land uses leads to changes in watershed hydrology and pollutant load rates. The areal increase in impervious surfaces in urban areas over undeveloped rural and natural land uses leads to greater surface water runoff volumes. The increased runoff coupled with human activities increases the types of pollutants and delivery rate of these pollutants to surface waters. Impermeable surfaces shed water as surface runoff, lowering the infiltration and evapotranspiration components of the hydrologic cycle. Surface runoff is generally directed to storm sewers and other conveyance systems to rapidly move the large volumes to receiving waters and prevent flooding. This section provides a general discussion about the methodology used to quantify the amount of phosphorus entering surface waters from urban runoff. A more detailed discussion of the methodology used for this analysis is included in Appendix J. The results of the phosphorus loading computations for this source are discussed in Section 3.3.8.

The methodology used for this analysis involved review of the literature to document urban runoff quality in Minnesota, determining the extent of each urban land cover type present within each basin, and calculating the variation of the estimated phosphorus loadings under each flow condition. It was apparent from the literature review that the quality and quantity of the data available was insufficient for the use of quantifying basin-specific data for this assessment. The need to quantify phosphorus loadings across basins with regard to three different hydrologic conditions (low, average and high

flow conditions) required that a method be developed to model phosphorus loadings with regard to land use and hydrologic conditions. The scientific literature was thus reviewed to determine the hydrologic regimes, nutrient cycling mechanisms and phosphorus loading factors for each of the urban land cover categories.

In an attempt to determine the range of phosphorus concentrations in urban runoff, the summary data was reviewed and the site specific data from previous or ongoing monitoring studies was examined. The available monitoring data included a combination of flow-weighted mean or event mean concentrations, expressed as median, geometric or arithmetic means. The inconsistency in data reporting limited the use of many of the data sets found during the literature review process. Schwartz and Naiman (1999) suggest using the mean concentration as the representative concentration introduces significant bias into the annual load estimates and report that the use of flow-weighted mean concentration (FWMC) provides an unbiased estimate of annual load. Data collected in the literature review, chosen for inclusion in the database, had to meet the following criteria:

- Phosphorus data was collected for the duration of individual storm events and was reported as Event Mean Concentration (EMC)

- Numerous samples had to be collected at the same monitoring location throughout a given year

- Land use was either reported in adequate detail or land use could be determined using ArcView with delineated watersheds and USGS National Land Cover Data (NLCD)

- A large fraction of the runoff generated from a monitored watershed was not routed through storm water treatment BMPs such as detention ponds

Precipitation data was also gathered from the rain gage nearest to the chosen monitoring sites. Driver and Tasker (1990) found that, in developing linear regression equations for the estimation of storm water loads, the total storm rainfall and total contributory drainage area were the most significant factors, while impervious area, land-use and mean annual climatic characteristics were also significant. The high level of correlation between land use type and effective impervious area has also been noted by many investigators (Schueler, 1987; Driver and Tasker, 1990; Beaulac and Rechkow, 1982). Likewise nutrient loadings increase with increasing impervious surface area, most likely due to the ease of washoff and transport in curb and gutter systems and on other hard surfaces

(Brezonik, *et al*, 2002; Schueler, 1994). Higher impervious percentage watersheds yield lower phosphorus concentrations, but the larger volume of water leads to the higher phosphorus loading rates (Bannerman, *et al*, 1992; Swenson, 1998; Beaulac and Rechkow, 1982). McFarland and Hauck (2001) suggest that use of multiple regression analysis using measured flows and water quality data for heterogeneous land uses allows the estimation of loads that represent average conditions accurately. For this assessment, an evaluation was completed for the monitoring data collected at the same location for multiple years and under different hydrologic conditions. This data showed that the concentration of phosphorus in stormwater at the same site is often higher during dry years compared to an average year, and is lower during a wet year compared to an average year. From the available studies that had multiple years of monitoring data, a ratio was developed by dividing the concentration of total phosphorus in runoff for a wet year by the average year, and by dividing the concentration of total phosphorus in runoff for a dry year by the average year. Overall, the wet to average ratio was 0.8 and the dry to average ratio was 1.18. To quantify the relationship between annual precipitation, land use (the four urban NLCD land uses: low intensity residential, high intensity residential, commercial-industrial-transportation, and urban recreational grasses), impervious percentage, and the annual flow-weighted total phosphorus concentration, single variable and multivariate linear regressions were performed, based on estimated impervious percentages for each land cover type. There was a significant relationship between annual flow-weighted mean total phosphorus concentration, impervious percentage, and annual precipitation.

Export coefficients are commonly reported according to land use and are developed during a given year under a particular hydrologic condition (Beaulac and Reckhow, 1982: Reckhow, et al, 1980; Panuska and Lillie, 1995; Clesceri, et al, 1986a; Clesceri, et al, 1986b; McFarland and Hauck, 2001). In some cases the export coefficient is adjusted to reflect a normal climatic year. The most common approach to estimating loads is based upon Schueler's (1987) regression of rainfall runoff volume and percentage imperviousness of a watershed combined with a flow-weighted mean concentration. The equation is widely used for loading estimates and is used in this assessment to determine runoff coefficient based upon impervious percentage:

$$\text{Runoff coefficient } (R_v) = 0.05 + 0.009 \text{ (Impervious Percentage)}$$

The pollutant load is calculated by multiplying runoff volume with the pollutant concentration to obtain a mass load. For the purposes of defining and quantifying the phosphorus loads to Minnesota basins, the land uses within incorporated areas were classified and enumerated using the USGS National Land Cover Data (NLCD). The National Land Cover Data Set for the Conterminous United

States is derived from the Landsat thematic mapper data system (Vogelmann, 2001). The NLDC cover classes included in the land uses within incorporated areas assessed are:

- Urban Developed Areas
    - Low intensity residential
    - High intensity residential
    - Commercial/Industrial/Transportation
- Deciduous Forest
- Evergreen Forest
- Mixed Forest
- Shrubland
- Grasslands/Herbaceous
- Urban / Recreational Grasses
- Agricultural lands
    - Pasture/Hay
    - Row Crops
    - Small Grains
- Other
    - Quarries/Strip Mines/Gravel Pits
    - Transitional (new development)

The percent imperviousness applied to each of these urban land uses and then used in calculation of the runoff coefficient for this assessment are as follows:

Land cover class	Percent impervious
Low intensity residential	32%
High intensity residential	42%
Commercial/Industrial/Transportation	57%
Urban / Recreational Grasses	32%
Transitional	57%

(adapted from Zielinski, 2002 and analysis of TCMA GIS coverage)

For this assessment, all of the developed urban uses are assumed to have storm water conveyance systems in place – minimally drainage ditches and conveyance channels up to full curb and gutter with piping. The number of acres for each of the four developed urban land uses was determined for the incorporated areas in each of the ten basins. To calculate the expected concentration of total phosphorus in urban runoff for each basin, the average percent imperious area for the four developed

urban land uses (high and low intensity residential, commercial/industrial/transportation and urban/recreational grasses) in each basin and the annual precipitation for the dry, average, and wet year were used as inputs to the regression model.

Phosphorus loading from the four developed urban land uses in each basin was then calculated according to the following equation:

Basin load = Concentration * Contributory area * Runoff coefficient * Annual Rainfall Depth

where: concentration is based upon the concentration regression equations developed for urban runoff in each of the basins,

contributory area is equal to the total area for each land use class,

runoff coefficient = 0.05 + 0.009 * impervious percentage,

annual rainfall depth is the annual precipitation for the loading flow condition scenario by basin.

The phosphorus load for each of the other non-agricultural land uses within incorporated areas were calculated by multiplying the phosphorus export coefficient by the contributory area and basin runoff factor, as described in Section 2.2.2.6. Phosphorus loads from agricultural land uses within incorporated areas were calculated using the same methodology as for the remaining agricultural areas statewide, as described in Section 2.2.2.1.

2.2.3 Bioavailability of Phosphorus by Source

The purpose of this section is to provide a discussion about the bioavailable fraction of phosphorus from individual point and nonpoint sources of phosphorus. A more detailed discussion of the methodology and results of this analysis are included in Appendix K. The results of the bioavailable phosphorus determinations for each source category are also presented in Section 3.2. This discussion is based on a review of the available literature and the results of POTW-specific and basin-specific sampling and analysis. This section is intended to:

- Provide an introduction to the forms of phosphorus in the aquatic environment

- Describe the results of the literature review for each category of point and nonpoint sources

- Present the results of POTW-specific and basin-specific sampling and analysis for bioavailable phosphorus

- Compare and summarize estimates of bioavailable phosphorus fraction for each source type

2.2.3.1 Forms of Phosphorus in the Aquatic Environment

In general, bioavailable phosphorus is defined as the portion of the total phosphorus in surface waters that is available for plant growth. Excess bioavailable phosphorus in freshwater systems can result in accelerated plant growth. Phosphorus is the principal nutrient causing excessive growth of algae and other aquatic plants in Minnesota's surface waters.

Phosphorus exists in water in either a dissolved phase or a particulate phase. Dissolved phosphorus in natural waters is usually found in the form of phosphates (PO_4^{-3}). Dissolved phosphates exist in three forms: inorganic (commonly referred to as orthophosphate or soluble reactive phosphorus- SRP), inorganic polyphosphate (or metaphosphate) and organically bound phosphate. Particulate phosphorus contains phosphorus sorbed to inorganic (mineral) and organic particles, including phosphorus contained within algae. Dissolved inorganic phosphate (orthophosphate) is the form required by plants for growth. The analytical procedure for measuring total phosphorus, which includes a sulfuric acid extraction, accounts for all forms of phosphorus, both dissolved and particulate, including phosphorus contained in algae.

Orthophosphates are immediately available in the aquatic environment for algal uptake. Natural processes produce orthophosphates, but major man-influenced sources include: partially treated and untreated sewage; runoff from agricultural sites; and application of some lawn fertilizers. Orthophosphate concentrations in a water body vary widely over short periods of time as plants take it up and release it. Polyphosphates are used for treating boiler waters and in detergents. Also, polyphosphates are used in drinking water treatment in many municipalities. In water, polyphosphates are unstable and will eventually convert to orthophosphate and become available for plant uptake.

Organic phosphates (particulate and dissolved) are bound or tied up in plant or animal tissue, waste solids, or associated with other organic matter. Organic phosphates are formed primarily by biological processes. They are contributed to sewage by body waste and food residues, and also may be formed from orthophosphates in biological treatment processes or by receiving water biota. After decomposition, the organic form can be converted to orthophosphate as a result of microbially-induced mineralization of phosphorus-containing organic matter.

Not all forms of phosphorus are utilized to the same degree or at the same rate by plants and microbial communities. Association of phosphorus with particulate or organic matter reduces bioavailability; such forms of phosphorus are immediately unavailable for uptake by algae. While a significant amount of phosphorus can enter water bodies in an immediately unavailable form, there is the potential for this unavailable phosphorus to undergo physical or chemical cycling processes that may convert it (all or partially) to the readily bioavailable form of phosphorus, orthophosphate. For example, the decomposition of organic matter by microbial activities can result in mineralization of phosphorus to orthophosphate. Desorption or dissolution of particle-associated phosphate represents another mechanism of conversion from unavailable to bioavailable forms.

DePinto *et al*. (1986) characterized phosphorus into three forms: orthophosphate – immediately bioavailable for algal uptake; external ultimately-available phosphorus – not immediately available but ultimately converted to orthophosphate at a specific rate; and external refractory phosphorus – not available while in the water column. Total bioavailable phosphorus is then comprised of orthophosphate and the external ultimately-available phosphorus. It is indeed the bioavailable phosphorus that affects the algal production in the aquatic environment in combination with other nutrients (e.g. nitrogen and silicon), light, and temperature.

Different sources provide water bodies with a variety of the forms of phosphorus described above, in variable proportions. Phosphorus in lakes and streams comes from both point and nonpoint sources. Point sources are typically publicly-owned wastewater treatment plants (POTWs) and permitted industrial discharges. Phosphorus discharged from wastewater treatment plants may come into the plant from a variety of sources. Nonpoint sources are typically polluted runoff from cities and farmland, erosion and sedimentation, atmospheric deposition, direct input by animals and wildlife, and natural decomposition of rocks and minerals.

A comprehensive literature search and review was conducted to compile available information on the bioavailable phosphorus fractions of individual point and nonpoint sources of phosphorus to surface waters. The results of this literature review are presented in the following discussion.

2.2.3.2   Bioavailable Phosphorus in POTW Effluent

The bioavailable phosphorus fraction in POTW effluent is generally assumed to be high compared to that of other sources to surface waters (Lee *et al*., 1980). Young *et al*. (1982) sampled the effluent from four municipal treatment plants in the vicinity of the Great Lakes during the summer of 1979 for bioavailable phosphorus. They conducted bioassays where measurement of phosphorus taken up

by *Scenedesmus* sp. provided the measure of bioavailable phosphorus fraction. They developed a series of relationships among different forms of phosphorus.

On average, 82% of the dissolved phosphorus was bioavailable in the short term (less than 30 days from sample collection). Orthophosphate was a major component of the dissolved phosphorus (69% on average). Moreover, the regression coefficient relating bioavailable dissolved phosphorus to orthophosphate was unity, indicating that the orthophosphate fraction was totally available.

For particulate phosphorus, they found that the bioavailable particulate phosphorus correlated closely with the total particulate phosphorus fractions. On average (with the samples taken from the effluent of the four wastewater treatment plants), 55% of the total particulate phosphorus was bioavailable in the short term (again, less than 30 days).

The ultimately bioavailable dissolved phosphorus (became bioavailable after 30 days) represented approximately 99 percent of the total dissolved phosphorus. The ultimately bioavailable particulate phosphorus was approximately 63 percent of the total particulate phosphorus.

Data from the wastewater treatment plants indicated that 83% of the total wastewater phosphorus in those effluent samples was ultimately available.

In addition to the information gathered from the literature review, effluent from eight Minnesota POTWs was sampled between October 13 and October 17, 2003. The samples were analyzed for total phosphorus and orthophosphate. The ultimately bioavailable particulate phosphorus was estimated using the relationship developed by Young *et al*. (1982) described above. The results of this analysis are presented in Table 2-2. The bioavailable phosphorus fraction in these samples ranged from 75-96%, with an average of 85.5%, which is typical for POTW effluents based on the results of the literature review. Measured particulate phosphorus concentrations also are consistent with expected range based on the literature. Chemical and biological phosphorus removal is implemented at all of these POTWs with the exception of Albert Lea and Wilmar. Albert Lea and Wilmar also have industrial discharges to the POTW that contain high phosphorus levels.

Table 2-2   Estimated Bioavailable Phosphorus (BAP) Fractions of Samples Collected from the Final Effluent of Eight Minnesota POTWs

City	TSS (mg/L)	Total P (mg/L)	Orthophosphate (mg/L)	Particulate P (mg/L)	Ultimately Bioavailable Particulate P (mg/L)	Particulate BAP fraction	Total BAP fraction
Albert Lea	<5.0	5.32	4.31	1.01	0.65	0.64	0.93
Alexandria	<5.0	0.187	0.102	0.085	0.07	0.78	0.90
St. Cloud	<5.0	0.250	0.068	0.182	0.13	0.70	0.78
Fergus Falls	<5.0	0.166	0.019	0.147	0.11	0.72	0.75
Mankato	11	2.04	1.57	0.47	0.31	0.66	0.92
MCES-Metro	<5.0	0.293	0.130	0.163	0.12	0.71	0.84
Rochester	13	0.948	0.286	0.662	0.43	0.65	0.76
Wilmar	10	7.24	6.41	0.83	0.54	0.65	0.96

2.2.3.3   Bioavailable Phosphorus in Runoff

The transfer of phosphorus from terrestrial to aquatic systems in runoff can occur in dissolved and particulate forms. Phosphorus loading from nonpoint sources depends on a large number of factors, such as geology and hydrology of the region, land use, and population density. For example, sandy soils have less retention of phosphorus than clays and high slope and high runoff lead to lower retention. Caraco (1995) found that population density was related to orthophosphate export from watersheds and predicted 47% of the variation in orthophosphate export in the dataset from 32 large rivers. Other variations could be related to the geochemical factors that alter orthophosphate in rivers or could be due to variability in human behaviors that lead to variable phosphorus export. For example, human agricultural practices, soil composition, diets, detergent use, and extent of sewer services and sewage treatment can vary greatly between different areas. Phosphorus loss from land not only affects the surface runoff, but also gets transferred in subsurface flow (Gaynor and Findley, 1995; Lennox *et al*., 1997; Haygarth *et al*., 1998; and Withers *et al*., 1999).

It has been shown that the orthophosphate concentration in surface runoff is related to the soil phosphorus concentration in the topsoil (McDowell and Sharpley, 2001). For example, Pote *et al*. (1996) found that that the orthophosphate concentration in surface runoff was linearly related to

phosphorus extracted by Mehlich-3 (r^2 of 0.72), Bray-I (r^2 of 0.75), Olsen (r^2 of 0.72), distilled water (r^2 of 0.82), iron oxide paper (r^2 of 0.82), acidified ammonium oxalate (r^2 of 0.85), and phosphorus sorption saturation (r^2 of 0.77).

Surface runoff from grassland, forest land or nonerosive soils carries little sediment and is generally dominated by dissolved phosphorus, although phosphorus transport attached to colloidal material also may be important where land is overstocked (Haygarth and Jarvis, 1997; Simrad *et al.*, 2000). Sharpley *et al.* (1995) also reported that runoff from grass and forestland carries little sediments, and is therefore, generally dominated by orthophosphate.

As reported by Sharpley *et al.* (1995), the discharge of organic and inorganic phosphorus in runoff from several Atlantic Coastal Plain watersheds was related to soil phosphorus composition. The high organic phosphorus content of forest soils (331 mg/kg; 70% of total phosphorus) contributed 51% of total phosphorus loss in runoff (0.31 kg/ha/y) as particulate organic phosphorus and 10% as dissolved organic phosphorus. For agricultural soils of lower organic phosphorus content (161 mg/kg, 25% of total phosphorus), only 32% of total phosphorus loss in runoff (2.41 kg/ha/y) was particulate organic phosphorus and 1% was dissolved organic phosphorus (Vaithiyanathan and Correll, 1992). Similarly, from 16 to 38% of phosphorus in runoff from Polish meadows and cultivated fields and as much as 70% of lake water phosphorus was bound to organic compounds (Szpakowska and Zyczynska-Baloniak, 1989). These losses varied seasonally, with both inorganic and organic phosphorus concentrations in canal and lake water decreasing during summer months (Ryszkowski *et al.*, 1989).

Estimates for urban runoff particulates, tributary particulates and lake sediments in the lower Great Lakes basins by bioassay methods have reported an average of 30% bioavailable phosphorus (Cowen and Lee, 1976; Williams *et al.*, 1980).

2.2.3.4   Bioavailable Phosphorus in Agricultural Runoff

The sources of phosphorus from agricultural land can include soil phosphorus, manure or fertilizer applications. Those sources of phosphorus emanate from a number of source areas within the landscape and their amount, form, and timing are very variable as a result of short-term and often unpredictable changes in hydrological conditions and farming practices, including crop rotation, the application of fertilizers and manures, or the movement of animals from one field to another (Lennox *et al.*, 1997). Phosphorus may be transported to a water body from agricultural lands by leaching, runoff or erosion. The loss of phosphorus in surface runoff from agricultural lands occurs as particulate and dissolved forms (Haygarth and Sharpley, 2000). Particulate phosphorus includes phosphorus associated with soil particles and large molecular-weight or organic matter eroded during

flow events and constitute the major proportion of phosphorus transported from most cultivated lands (60-90%, Pietilainen and Rekolainen, 1991). Several studies have reported that the loss of dissolved phosphorus in surface runoff from agricultural land depends on the phosphorus content of surface soil (STP- soil test P concentration), but that the relationship varies with soil type, tillage, and crop management (Pote *et al*., 1996; Sharpley *et al*., 1996). Moreover, it will depend on the topography and soil hydrology.

James *et al*. (2002) used fractionation procedures and phosphorus adsorption-desorption assays to delineate bioavailable forms and refractory or unavailable forms of phosphorus in the runoff of the Redwood River basin, an agriculturally-dominated tributary of the Minnesota River. Over several storm periods monitored in 1999, 75% of the phosphorus load originating from the watershed was in bioavailable forms while only 25% was in refractory forms. Bioavailable particulate forms included phosphorus loosely bound to suspended sediments (19%), phosphorus bound to iron (11%), and bioavailable particulate organic phosphorus (14%). After runoff discharges to receiving waters, the former two forms of bioavailable particulate phosphorus can be transformed to dissolved forms that are available to biota for uptake via eH and pH reactions and kinetic processes, while the latter form can be mineralized via decomposition processes. Bioavailable dissolved forms included orthophosphate and dissolved organic phosphorus.

Several studies have suggested that agricultural management may influence the bioavailability of phosphorus transported in runoff (McDowell and McGregor, 1980; Wendt and Corey, 1980). Concentration and amounts of bioavailable phosphorus in runoff from corn (*Zeamays* L.) were lower from no till compared to conventionally tilled plots under simulated rainfall (Andraski *et al*., 1985; Mueller *et al*., 1984). Bioavailable phosphorus in these studies was measured by resin extraction of unfiltered runoff, and thus includes dissolved phosphorus plus phosphorus desorbed from sediment (Huettl *et al*., 1979). However, Andraski *et al*. (1985) calculated that bioavailable phosphorus averaged 20% of total phosphorus and was not affected by tillage treatment.

Sharpley *et al*. (1992) assessed the impact of agricultural practices on phosphorus bioavailability in runoff by determining dissolved phosphorus, bioavailable particulate phosphorus, and particulate phosphorus in runoff from 20 watersheds (in the Southern Plains region of Oklahoma and Texas) unfertilized and fertilized, grassed and cropped watersheds over a 5-yr period. Although bioavailable phosphorus and bioavailable particulate phosphorus losses in runoff were reduced by agricultural practices minimizing runoff and erosion, the proportion of phosphorus transported in bioavailable forms increased. Both total phosphorus (14-88% as bioavailable phosphorus) and

particulate phosphorus (9-69% as bioavailable particulate phosphorus) bioavailability varied appreciably with agricultural practices. Thus, bioavailable phosphorus is a dynamic function of physical and chemical processes controlling both dissolved phosphorus and bioavailable particulate phosphorus transport. Dissolved phosphorus transport depends on desorption-dissolution reactions controlling phosphorus release from soil, fertilizer reaction products, vegetative cover, and decaying plant residues. Bioavailable particulate phosphorus is a function of physical processes controlling soil loss and particle-size enrichment and chemical properties of the eroded soil material governing phosphorus sorption availability. The authors also found that the percent bioavailability of particulate phosphorus transported in runoff from each of these watersheds decreased with an increase in sediment concentration of runoff averaged for each watershed. They found a linear regression relationship between particulate phosphorus availability and logarithm of sediment concentration (with r^2 =0.84):

$$Particulate\ Phosphorus\ Bioavailabilty\ (\%) = 82 - 15 \log sediment\ conc.\ (g/L)$$

This relationship may be attributed to an increased transport of silt- and sand-sized (>2 μm) particles, of lower phosphorus content than finer clay-sized (<2 μm) particles, as sediment concentration of runoff increases. Further, particulate phosphorus bioavailability may decrease with an increase in size of eroded soil particles, which contain less sorbed phosphorus and more primary mineral phosphorus (i.e., apatite) of lower availability compared with finer clay-sized particles (Dorich *et al.*, 1984; Sharpley *et al.*, 1981; Syers *et al.*, 1973).

O'Connor *et al.*, (2002) compared phosphorus bioavailability of biosolids, manures and fertilizer. They found that phosphorus bioavailability was greater for phosphorus-fertilizer than manures and biosolids. However, if biological phosphorus removal is implemented in the treatment process, phosphorus in biosolids tends to be as bioavailable (74% to 132%) as fertilizer phosphorus.

A study conducted by Ekholm and Krogerus (2003), with samples from different sources, concluded that phosphorus in agricultural runoff appeared to be more bioavailable to algae (31%) than phosphorus in forest runoff (16%).

2.2.3.5 Bioavailable Phosphorus in Atmospheric Deposition

For Lake Michigan, Murphy and Doskey (1975) reported a 30-fold greater total phosphorus concentration in rainfall than in lake water. Since 25-50% of the total phosphorus in rainfall is soluble, it is directly available to organisms in the lake (Murphy and Doskey 1975; Peters 1977).

The bioavailability of dry deposition or the particulate fraction of wet deposition can be characterized by the bioavailability of phosphorus in the soils in the region.

Increases in the atmospheric deposition of phosphorus may result from annual climatic changes (Sharpley *et al*. 1995). For example, the input of phosphorus in rainfall to an Oklahoma watershed in 1981 (208 g/ha/yr) was much greater than that in either 1982 (49 g/ha/yr) or 1983 (41 g/ha/yr) (Sharpley *et al*. 1985). This increase was attributed to the low annual rainfall in 1980 (642 mm, 105 mm below average). The drier soil was more susceptible to wind erosion and the airborne material increased the phosphorus content of subsequent rainfall and dry deposition.

2.2.3.6   Comparison of Phosphorus Bioavailability from Different Sources

Many forms of particulate matter in the waters of the State of Minnesota contain a certain amount of bioavailable phosphorus, the actual rate and extent of release of the bioavailable component depends on the physical and chemical characteristics of the material. It also depends on the biological characteristics as well as the population of the microorganisms in the suspended material mineralizes the organic detritus material. Young *et al*. (1995) have compared the relative bioavailability of particulate phosphorus from various sources to the Great Lakes by comparing the bioavailable phosphorus in particulate matter from point sources (wastewater suspended solids), and nonpoint sources (suspended solids and bottom sediments from tributaries, lake bottom sediments, and eroding bluff solids from the region). A wastewater treatment plant at Ely, Minnesota was also sampled and it showed the highest rate of release of bioavailable particulate phosphorus (0.27 grams released/gram particulate phosphorus/day, or 0.27/day) among the point and nonpoint sources sampled in that study (Young and DePinto, 1982). The release rate did appear to decline in magnitude as treatment of wastewater progressed from the raw influent → biologically treated effluent → final effluent (i.e., 0.30 /day → 0.27 /day → 0.20 /day). Young and DePinto (1982) summarized the results on relative bioavailability of particulate phosphorus for the point and nonpoint sources (Table 2-3).

Ekholm and Krogerus (2003) analyzed 172 samples (during 1990-2000) representing phosphorus in point and nonpoint sources and in lacustrine matter. The bioavailability of phosphorus expressed as the proportion of potentially bioavailable phosphorus ranged from 3.3 to 89% (Table 2-4).

Table 2-3   Relative Bioavailability of Particulate Phosphorus from Various Sources to the Lower Great Lakes (Young and DePinto 1982)

Source	Bioavailable Percentage	Release Rate (1/day)
Wastewater (≤ 80%)	≤ 80%	≤ 0.4
Bottom sediments (≤ 50%)	≤ 50%	≤ 0.2
Tributary suspended sediment	≤ 40%	≤ 0.1
Eroding bluff	~0	~ 0

Table 2-4   Proportion of Bioavailable Phosphorus in Total Phosphorus by Different sources (Ekholm and Krogerus 2003)

Source	Bioavailable P (% of Tot-P)	
	Mean	Min.-Max.
Wastewater effluent from rural population	89	74-98
Biologically treated urban wastewater effluent	83	61-103
Dairy house wastewater	69	27-93
Biologically and chemically treated wastewater effluent	36	0-67
Field runoff	31	15-50
Industrial wastewater effluent	30	4-89
Fish fodder and feces	29	9-72
Large Rivers water	20	3-45
Agricultural rivers	20	12-30
Field surface soils	19	6.8-24
Forest runoff	16	0-55
Lake settling matter	7.9	1.6-21
Lake bottom sediments	3.3	0.1-11

2.2.3.7   Summary of Literature Review

The above review covers as much research and data from phosphorus bioavailability studies as could be found in the available time and resources. There is a desire to estimate the fraction of phosphorus in each potential source category identified by the MPCA as contributing phosphorus to Minnesota waters. However, the bioavailability of some of these individual source categories has not been studied; therefore, we were not able to find directly applicable estimates for bioavailable fractions in

the literature. The general categories for which data are available include: municipal wastewater treatment plants, agricultural, forest and urban runoff, and atmospheric deposition.

While the dissolved phosphorus from any of these sources can generally be assumed to be 100% bioavailable, the particulate phosphorus associated with these various source categories in general exhibit a wide range of bioavailability.

For point sources, the fraction of total phosphorus in the discharge that is bioavailable is not only governed by the sources of phosphorus to the treatment plant influent (e.g., human wastes, household cleaners, groundwater infiltration, etc.) but it will be dependent on the treatment train being employed within the plant. Data are generally available for wastewater treatment plant influent and effluent, however not for all individual phosphorus source categories. Knowing, however, that household cleaners and detergents are amended with polyphosphates, it is reasonable to assume that virtually 100% of these categories will ultimately become available by hydrolysis to orthophosphates.

For nonpoint sources, the input of total phosphorus and bioavailable phosphorus will be strongly dependent on the land use from which the phosphorus load is derived (e.g., agricultural runoff will be different from forestland runoff). Furthermore, agricultural practices can affect bioavailable phosphorus appreciably. Another determinant is the surficial geology within the watershed. We have seen, for example, that phosphorus associated with calcareous minerals like apatite is much less bioavailable than phosphorus adsorbed to iron-oxide minerals. In general, the particulate phosphorus in non-point sources derived from land runoff tends to be less bioavailable than point source particulate phosphorus.

Bioavailable phosphorus fractions for each of the specific source categories of interest were estimated by combining the results of the literature review with best professional judgment to specify a most likely value for a number of the remaining phosphorus source categories. A range was also estimated in an attempt to cover the potential range site-specific determinations might show. These estimates are presented in Table 2-5. These estimates of bioavailable fraction should be used with care, understanding the uncertainty inherent in each estimate. Nevertheless, they can be used to assess relative contributions of bioavailable phosphorus from the source categories to assist in planning additional data collection or targeting specific sources for control. As evident from the literature review, wide ranges of bioavailable fractions were noted for runoff sources, while estimation techniques for the bioavailable fraction from POTW effluent were better quantified.

Table 2-5   Estimates of Bioavailable Phosphorus Fractions for Specific Source Categories

Phosphorus Sources			Fraction of PP that is Bioavailable (Range)	Fraction of PP that is Bioavailable (Most Likely)	Fraction of DP that is Bioavailable (Most Likely)	Fraction of TP that is Particulate (Most Likely)	Estimate of TP that is Bioavailable (Most Likely)
Point Sources	Phosphorus Sources to POTWs	Automatic Dishwasher Detergent	NA	NA	1.0	0.0	1.0
		Dentifrices (toothpastes)	0 – 0.1	0.05	NA	1.0	0.05
		Other Household Cleaners or Non-ingested Sources	NA	NA	1.0	0.0	1.0
		Food Soils/Garbage Disposal Wastes	0.7 – 0.9	0.8	1.0	0.9	0.8
		Human Waste Products	0.7 - 0.9	0.8	1.0	0.3	0.94
		Raw/Finished Water Supply	0.4 - 0.6	0.5	1.0	0.1	0.95
		Groundwater Intrusion (I&I)	0.2 - 0.5	0.3	1.0	0.5	0.65
		Process Water	0.2 - 1.0	0.7	1.0	0.1	0.97
		Noncontact Cooling Water	0.4 - 0.8	0.6	1.0	0.3	0.88
		Car Washes	0.2 - 0.8	0.5	1.0	0.3	0.85
	POTW Effluent		0.6 – 0.8	0.7	1.0	0.5	0.855
	Privately Owned Wastewater Treatment Systems for Domestic Use (effluent)		0.6 - 0.9	0.8	1.0	0.3	0.94
	Commercial/Industrial Wastewater Treatment Systems (effluent)		0.2 - 0.8	0.6	1.0	0.3	0.88
Non-Point	Individual Sewage Treatment Systems		0.6 - 0.9	0.8	1.0	0.2	0.96

Phosphorus Sources			Fraction of PP that is Bioavailable (Range)	Fraction of PP that is Bioavailable (Most Likely)	Fraction of DP that is Bioavailable (Most Likely)	Fraction of TP that is Particulate (Most Likely)	Estimate of TP that is Bioavailable (Most Likely)
Sources	Agricultural Runoff	Improperly Managed Manure	0.5 - 0.7	0.6	1.0	0.5	0.80
		Crop Land Runoff	0.2 - 0.7	0.4	1.0	0.7	0.58
	Urban Runoff	Turfed Surfaces	0.2 - 0.7	0.4	1.0	0.7	0.58
		Impervious Surfaces	0.10 - 0.5	0.2	1.0	0.5	0.60
	Forested Land		0.2 - 0.5	0.3	1.0	0.8	0.44
	Roadway and Sidewalk Deicing Chemicals	salt	0.2 - 0.8	0.6	1.0	0.2	0.92
		sand	0.1 - 0.3	0.2	1.0	0.8	0.36
	Stream Bank Erosion		0.1 - 0.5	0.3	1.0	0.8	0.44
	Atmospheric Deposition	Dry	0.05 – 0.4	0.2	NA	1.0	0.2
		Wet	0.05 – 0.4	0.2	1.0	0.6	0.5

2.2.4 Assessment of Effluent Total Phosphorus Reduction Efforts by POTWs

This section provides a general discussion about the methodology used to assess the effluent total phosphorus reduction efforts of POTWs. A more detailed discussion of the methodology used for this analysis is included in Appendix L. The results of this assessment are discussed in Section 3.5.

This discussion is intended to provide the Minnesota Pollution Control Agency (MPCA) with information on current practices of cities to reduce the phosphorus concentration in their wastewater treatment plant (WWTP) effluent through such approaches as reduction in the influent phosphorus loading, chemical phosphorus precipitation, and enhanced biological phosphorus removal (EBPR). Information was collected from six Minnesota cities and two Oregon cities on their programs to reduce their effluent phosphorus loading. A small sampling of Minnesota cities was used due to the limited number of cities that had data available on phosphorus reduction and its costs. The two Oregon cities were included because of their ability to meet a very stringent effluent phosphorus limit of 0.07 mg/L. Where available, costs for the specific phosphorus reduction efforts are provided. Finally, conclusions are drawn on the effectiveness of effluent phosphorus reduction efforts based on the data provided.

As mentioned above, three approaches were used either separately or in combination by the communities surveyed to reduce their effluent phosphorus concentrations: source reduction, chemical precipitation, and EBPR. Source reduction efforts varied significantly between cities in the survey. The simplest approach was a public education campaign to promote reductions in the use of household products with high concentrations of phosphorus. The more aggressive cities implemented fees based on the phosphorus content of the sewered discharge for their significant industrial users (SIU). Pretreatment was also required in one city if a SIU exceeded a pre-defined phosphorus loading threshold.

Chemical phosphorus precipitation is the use of metal salts to promote the precipitation of metal phosphates. Iron or aluminum are the most commonly used metals. The metal salt can be added at many different points in the WWTP treatment train. The most common point of application is immediately prior to secondary clarification. The chemical used and point of application are identified for each plant surveyed. The equipment required for chemical precipitation is minimal with systems adding metal salts prior to secondary clarification needing only a bulk storage tank and a chemical dosing pump. The largest cost for chemical precipitation phosphorus treatment is operations, which includes chemical cost and the cost of additional sludge disposal. The chemical costs are provided for all WWTPs surveyed using chemical precipitation.

EBPR is achieved in the activated sludge system by promoting the growth of bacteria that can hyper-accumulate phosphorus. This is achieved by creating an initial anaerobic zone in the activated sludge system followed by the traditional aerobic zone. In addition, low molecular weight organic acids must be present in the anaerobic zone to achieve EBPR. These acids can be produced in the sewer system, in the primary clarifier, or in a separate sludge fermenter. EBPR can be implemented using a wide range of approaches. The simplest approach can be to adjust air flow within the activated sludge basins to create the anaerobic zone. The more sophisticated approaches can require separate anaerobic basins and separate sludge digestion tanks. Phosphorus is ultimately removed from the EBPR system when the bacteria, which have hyper-accumulated phosphorus, are wasted from the activated sludge system.

It should be noted that WWTPs that have not implemented phosphorus treatment (i.e., either chemical phosphorus precipitation or EBPR) will likely see a reduction in the effluent phosphorus concentration proportional to the reduction in influent phosphorus concentration. WWTPs using chemical precipitation to meet effluent phosphorus limits will not likely experience a reduction in effluent phosphorus concentration if the influent phosphorus concentration is reduced because chemical precipitation will continue to be required to meet the effluent phosphorus limit. A reduction in influent phosphorus (soluble) concentration will reduce the amount of chemical required to achieve the effluent phosphorus limit, which will ultimately result in a reduction in chemical cost for phosphorus treatment. However, if the influent phosphorus was not soluble, which is precipitated chemically, but was particulate phosphorus, which is precipitated by flocculation, there may not be a direct reduction in chemical costs. Finally, WWTPs using EBPR will not likely experience a reduction in effluent phosphorus concentration if the influent phosphorus concentration is reduced because of the limits of this technology. The cost for operating EBPR will not be affected by the reductions in the influent phosphorus concentration.