Lower Green Bay and Lower Fox Tributary Modeling Report

Source Allocation of Suspended Sediment and Phosphorus Loads to Green Bay from the Lower Fox River Subbasin Using the Soil and Water Assessment Tool (SWAT)

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CHAPTER 1. INTRODUCTION AND PROJECT OVERVIEW

The overall objective of this project is to assist in developing load allocations of phosphorus and suspended sediment in the 1580 km^2 Lower Fox River Subbasin (Figure 1-1). To accomplish this task, sources of phosphorus and sediment export to lower Green Bay were identified and quantified through watershed model simulations and other techniques. A modified version of the USDA-ARS Soil and Water Assessment Tool model (version 4/18/2001), which was developed by Arnold et al. (1996), was applied to the Lower Fox River Subbasin to simulate daily stream flow, and suspended sediment and total phosphorus loads from both urban and rural non-point sources within the subbasin. These loads were then routed down the Fox River along with point source and urbanizing loads to lower Green Bay. Simulated export to Green Bay from the subbasin was conducted for numerous scenarios including: (1) Baseline 1992 conditions; (2) Baseline 2000 conditions; and (3) alternative management or policy scenarios which were compared to the Baseline 2000 conditions. All scenarios utilized a 1977 to 2000 climatic period for SWAT simulations, or a portion thereof.

Modifications were made to the FORTRAN code to facilitate the modeling process, and to improve the ability of the model to simulate stream flow and phosphorus and suspended sediment export under conditions in northeast Wisconsin. The model was primarily calibrated with discharge and constituent data from Upper Bower Creek (36 km^2). The Nash-Sutcliffe coefficient of efficiency (NSCE) was used as the primary criterion to calibrate the model for suspended sediment and phosphorus loads. Model validation was conducted with daily stream flow data and available constituent data from Upper Bower Creek, Duck Creek, the East River, and the Upper East River at Midway Road.

Simulated data were compared to observed discharge and constituent loads. Direct comparisons between individual events, statistical measures and graphical relationships supported the conclusion that the model can be applied to predict sediment and phosphorus loads at the subwatershed and watershed scale with an acceptable degree of accuracy. Furthermore, there was sufficient evidence to support the conclusion that the model can be applied to predict sediment and phosphorus loads to Green Bay from watersheds in the subbasin with an acceptable degree of accuracy. In conclusion, the SWAT model, as applied in this project, can be reliably used as a tool to make improved management decisions.

Figure 1-1 shows the watersheds, subwatersheds, major streams, weather stations and monitoring stations that were utilized in this project. The Lower Fox River Subbasin empties into Green Bay, and it is the lower most subbasin in the 16,500 km^2 Fox-Wolf Basin. In this report, the Lower Fox River Subbasin shall simply be referred to as the "subbasin". Additional USGS-operated monitoring stations installed through the Lower Fox River Watershed Monitoring Program (LFRWMP; www.uwgb.edu/watershed) and by the Green Bay Metropolitan Sewerage District (GBMSD) are also displayed in Figure 1-1 because information developed from data collected at these stations will be used to calculate daily sediment and phosphorus loads that can serve to assess the accuracy of the modeled results presented in this report.

This report contains the following chapters which describe: (2) SWAT methods and inputs; (3) computation of barnyard loads; (4) point source loads; (5) modifications to the SWAT model; (6) model calibration, (7) model validation and assessment; (8) comparisons between simulated and observed loads to Green Bay and watershed outlets; (9) sensitivity analysis; (10) modeled results, and allocation of loads to subwatershed outlets, watershed outlets, and to Green Bay; (11) alternative scenarios; and (12) summary and conclusions.

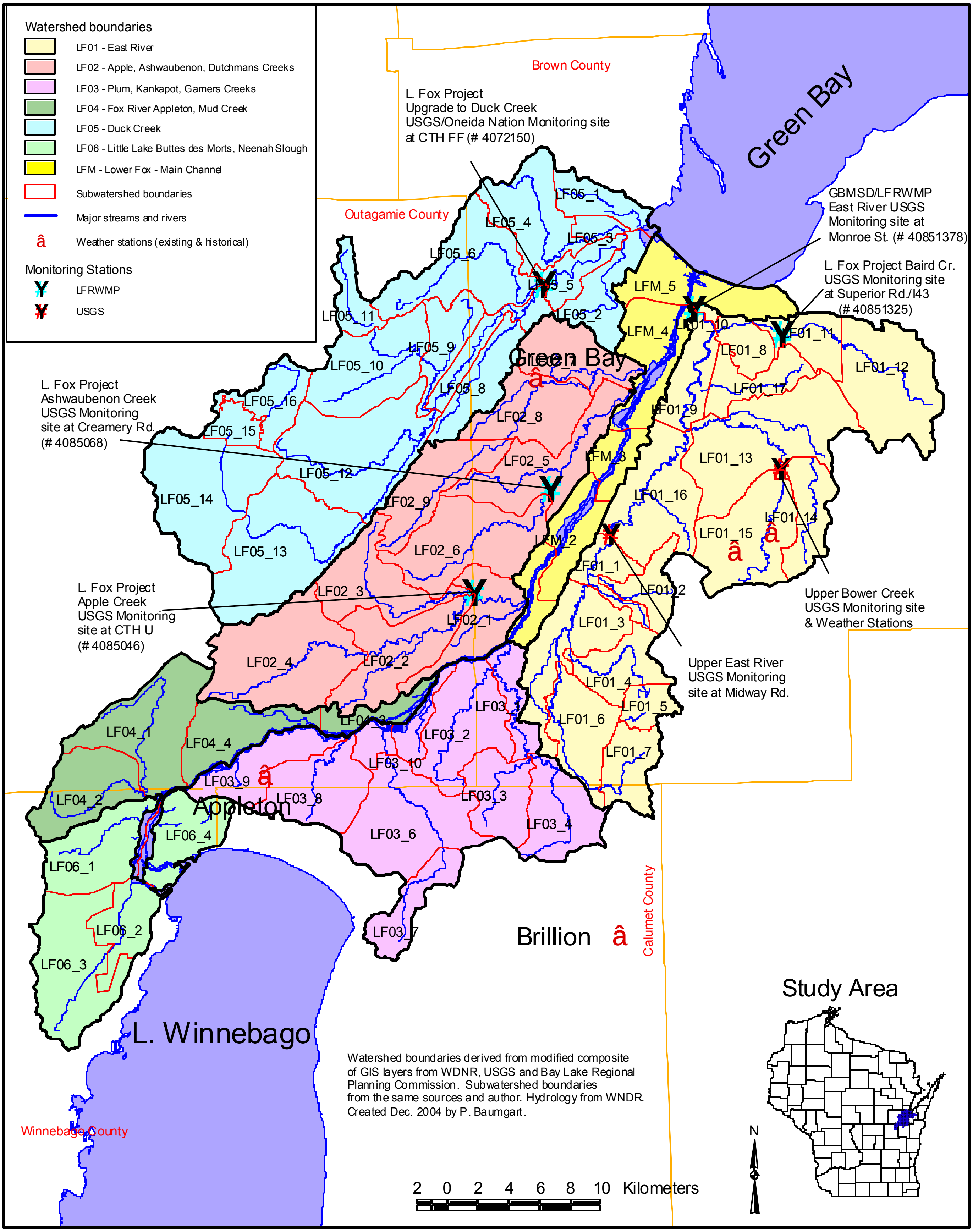

Figure 1-1. Lower Fox River Subbasin watersheds, subwatersheds and major tributariies.

CHAPTER 2. METHODS AND SWAT MODEL INPUTS

APPROACH

The Soil and Water Assessment Tool (SWAT; Arnold et al. 1996, Neitsch et al. 2001) model requires numerous inputs including watershed boundaries, surface and groundwater hydrology, climatological data, soils, land use information, crops and other vegetation, and tillage and nutrient management practices. This chapter contains a brief description of the model and describes the methods used to supply these inputs to the model. The model framework that was used to simulate agricultural, forest, grassland, wetlands, urban and urbanizing areas within each subwatershed will also be described. Simulation periods included: 1977 to 1996; 1977-2000, 1989-2000 and subsets derived from these periods.

SWAT Model: SWAT was developed by the USDA-ARS to improve the technology used in the SWRRB model (Simulator for Water Resources in Rural Basins; Williams et al., 1985; Arnold et al., 1990). SWAT is a distributed parameter, daily time step model that was developed to primarily assess non-point source pollution from watersheds and large complex river basins. SWAT simulates hydrologic and related processes to predict the impact of land use management on water, sediment, nutrient and pesticide export.

With SWAT, a large heterogenous river basin can be divided into hundreds of subwatersheds; thereby, permitting more realistic representations of the specific soil, topography, hydrology, climate and management features of a particular area. In addition, point source loads and outputs from other models can be input to the model. Major crop and management components used in the EPIC model (Sharpley and Williams, 1990) have been added to SWAT; consequently, it can better represent the actual cropping, tillage and nutrient management practices typically used in Northeastern Wisconsin. Modeled output data from SWAT can be easily input to a spreadsheet or database program, thereby making it easier to model large complex watersheds with various management scenarios efficiently.

Major processes simulated within the SWAT model include: surface and groundwater hydrology, weather, soil water percolation, crop growth, evapotranspiration, agricultural management, urban and rural management, sedimentation, nutrient cycling and fate, pesticide fate, and water and constituent routing. SWAT also utilizes the QUAL2e submodel to simulate nutrient transport. SWAT allows the use of a separate input file for each subwatershed, hydrologic response unit, routing reach, soil, groundwater, pond/wetland, management practice, stream water quality reach, and chemical type. A number of other files are also utilized by SWAT including: basin, weather, tillage, crop, pesticide, fertilizer, irrigation, reservoir, lake water quality, and routing configuration files. Control of these files is managed through a single "control-input-output" file which allows for much flexibility. A more detailed description of this model can be found at the following Internet address: http://www.brc.tamus.edu/swat/.

Although the definitions of total suspended solids (TSS) and suspended sediment are different, these terms shall be used interchangebly in this report. The reasons for doing this include: (1) the uncertainty inherent to modeling sediment loads is likely to be larger than the differences between these two parameters; (2) TSS and suspended sediment concentrations and loads from various sources were used to calibrate and validate the model; and (3) the two parameters are not that different in the subbasin given the shallow overland slopes and clay till soils present in the subbasin.

Subbasin Configuration: As illustrated in Figure 1-1, the Lower Fox River subbasin was divided into seven major hydrologic units (watersheds): (1) LF01 - East River; (2) LF02 - Dutchman, Ashwaubenon, and Apple Creeks; (3) LF03 - Plum, Kankapot and Garners Creeks; (4) LF04 - Appleton Watershed, which includes Mud Creek; (5) LF05 - Duck Creek; (6) LF06 - Little Lake Buttes des Morts Watershed, which includes the Neenah Slough Creek; and (7) LFM - Lower Fox River Main Channel. These watersheds were further delineated into a total of 65 subwatersheds according to surface hydrology, land use and the placement of monitoring stations. In most cases, the subwatersheds shown in Figure 1-1 were delineated such that their size was similar to that of the primary calibration site: Upper Bower Creek subwatershed (36 km^2). The methods used for delineating the Lower Fox River Subbasin and providing inputs to the model are described below. As previously mentioned, the Lower Fox River Subbasin shall simply be referred to as the "subbasin" in this report.

Application of Geographical Information System: PC ARC/INFO (vector-based GIS), ARCVIEW, and ARCVIEW Spatial Analyst (grid-based GIS) were used to construct, process and analyze a variety of GIS coverages to supply inputs to the SWAT model. All of these software programs were developed by Environmental Systems Research Institute, Inc. (ESRI). Unless otherwise indicated, all raster-based layers were processed with a 30 square meter cell resolution.

The following GIS data layers were used to provide inputs to the SWAT model and to prepare GIS-based maps and analyzes:

1. 1:24k WDNR watershed boundaries
2. East River subwatershed boundary coverage from the Bay Lakes Regional Planning Commission
3. Upper East River subwatershed boundary coverage from the USGS
4. Digital soil surveys from Brown, Calumet, Outagamie and Winnebago counties
5. WDNR 30 meter digital elevation model (DEM), used to derive overland slope
6. 1:24k surface water hydrology from WDNR
7. USGS 1:24k Quadrangle Digital Raster Graphic Images - topographic maps
8. WISCLAND 1992 Land Cover, based on satellite imagery, from WDNR
9. Land use images and maps from the East Central Wisconsin Regional Planning Commission
10. Land use GIS shapefile from the Brown County Planning Department
11. Sewer Service Boundaries from the East Central Wisconsin Regional Planning Commission
12. Sewer Service Boundaries from the Green Bay Metropolitan Sewerage District
13. Miscellaneous: roads, county boundaries, etc. from the WDNR
14. Brown County buffer strip coverages and associated stream hydrology layer
15. 1992 Digital orthophotos for Brown, Calumet and Winnebago counties, provided by WDNR

GIS coverages were projected into WTM-NAD83/91 coordinates. Further details concerning the use of these GIS layers is described below. Fox-Wolf Basin 2000 evaluated the ARCVIEW BASINS/SWAT interface for WDNR and EPA in 2000 (SWAT99) and 2001 (SWAT2000). Although this evaluation showed this interface to be a highly useful tool, the ARCVIEW BASINS interface was not utilized in this project because the SWAT model and GIS, in combination with the automated export/import capabilities of spreadsheet and database software programs offered greater flexibility, such as the ability to simulate a variety of crop rotations while also simulating different levels of crop tillage.

WATERSHED DELINEATION AND HYDROLOGY

Watershed Delineation: Several information sources were used to create a preliminary watershed boundary GIS coverage for the subbasin. The 1:24,000 statewide watershed boundary GIS layer (wsdnt024), provided by the Wisconsin Department of Natural Resources (WDNR), was supplemented with other GIS layers including: (1) "Small Project Layer" from the WDNR, which contained delineations of some subwatersheds within the Duck, Apple and Ashwaubenon Priority Watershed Project; (2) subwatershed boundaries for the lower portion of the East River, produced by the U.S. Geological Survey, Madison, Wisconsin (applied in McIntosh 1994); and (3) delineation of the upper portion of the East River, extracted from an East River Watershed coverage produced by the Bay Lakes Regional Planning Commission, Green Bay, Wisconsin. These layers were combined to produce a preliminary watershed boundary coverage for the subbasin.

The preliminary watershed boundary layer was then converted into an ARCVIEW shape file for further editing within ARCVIEW. Digital Raster Graphic versions of 1:24,000 U.S. Geological Survey (USGS) topographic maps (DRG's) obtained from the WDNR were used as background coverages within ARCVIEW to assist in adding additional subwatershed boundaries to the shape file. With the exception of two of the DRG's (Kaukauna and Green Bay East), the DRG's in the subbasin were based on 1992 aerial imagery. Standard versions of USGS 1:24,000 7.5 Minute Quadrangles topographic maps were also obtained from USGS, Reston, Virginia; all of these maps were based on 1992 imagery. In addition to the manual delineation process, the ARCVIEW interface for SWAT was applied to the 30-meter digital elevation model (DEM) to automate the delineation process and to clear up any questionable boundaries.

Watershed boundary delineation was primarily based on surface hydrology and topography, but land cover and land use were also utilized for delineating some urban areas. The final subbasin watershed delineation (Figure 1-1) was then used to create subwatershed-specific model inputs by overlaying this boundary layer with other GIS layers.

Hydrology: A 1:24,000 hydrology coverage which encompassed the entire lower Fox River Subbasin was provided by the WDNR. This coverage was provisional, so it contained no annotation or hydrological attributes. The hydrology layer was "intersected" with the subbasin watershed boundary coverage to provide stream segments that coincided with watershed boundaries. This layer was then used to assign main channels and routing channels to the individual subwatersheds along with the corresponding stream lengths and channel slopes, which served as model inputs. Figure 1-1 shows the main and routing channels. Elevation changes, as determined with the DRG topographic maps, were combined with main and routing reach channel lengths to compute the corresponding channel slopes required by SWAT.

CLIMATOLOGICAL INPUTS

Climatological Data: The locations of the weather stations used in this study to provide measured daily precipitation and temperature data are shown in Figure 1-1. Simulated temperature and precipitation data were not used. The Green Bay airport was the only NOAA National Weather Service (NWS) Station utilized in this study. Three weather stations located in the Upper Bower Creek watershed were maintained and operated by the USGS. A station near Greenleaf was operated by the University of Wisconsin (UW). The remaining stations were official NWS cooperative observers. With the exception of the USGS site at Bower Creek and the UW site, all of the other data were supplied in ASCII format by the UW-Extension Geological and Natural History Survey State Climatology Office in Madison, Wisconsin. Table 2-1 summarizes the stations type, data availability, and the years that were used in this study.

Table 2-1. Sources of climatological data used in SWAT simulations, and assignment of climatological data to subwatersheds.

Weather Station	Station Type	Period used in Simulations	Subwatersheds Assignments (see Fig. 1-1 for subwatershed locations)
Green Bay Austin Straubel International Airport	NOAA NWS site	Precip. and Min./Max. Temperature 1976-2000	LFM1 - precip & temp: All LF01 - temp: All LF01 - precip: subs 8,9,10 LF02 - precip & temp: subs 5,6,7,8,9 (Dutch. & Ash.) LF05 - precip: all except 13 LF05 - temp: All
WHBY Radio in Appleton	coop. observer	Precip. and Min./Max. Temperature 1976-2000	LF02 - precip & temp: sub 1,2,3, and 5 (Apple Cr.) LF03 - precip: subs 1,2,5,6,7,8,9,10 LF03 - temp: All LF04 - precip. & temp: All LF05 - precip: sub 13 only LF06 - precip & temp: All
Brillion	coop. observer	Precip. 1976-2000	LF01 - precip: subs 5,6,7 LF03 - precip: subs 3,4
Greenleaf	UW-AWON	Precip. 1993-1996	LF01 - precip: subs 1,2,3,4
USGS Bower Creek Rain Gages (average of up to 3 gages)	USGS	Rainfall 1990-1997 with some missing periods	LF01 - precip: subs 1,2,3,4,11, 12,13,14,15 16,17 Except Greenleaf used for subs 1,2,3,4 from 1993-1996, and Green Bay used during rest of record
Seymour Cooperative	coop. obs.	Precip. 1983-1996	none (not used, see narrative)

Days with trace amounts of precipitation were set to zero. Data from the closest available site were substituted whenever daily values were missing. Missing daily values were replaced with values from nearby stations. The Green Bay NWS site contained the most complete data set. Data from the Seymour site were not utilized because there was a limited data set (1983-present), and there were obvious phase differences between this station and the others (Baumgart 1998). Data from all weather stations were processed into an ASCII format that was compatible with SWAT. A separate precipitation or temperature file was utilized for each weather station and model simulation period. Measured precipitation and temperature data from Appleton, Brillion, Bower Creek and Green Bay stations were used to produce daily weather data sets for a number of simulation periods, including the 1976-96 period that was applied to the baseline (1992), and the 1976-2000 period that was applied to year 2000, and future conditions scenarios (all periods were inclusive: Jan. 1 to Dec. 31). The first year of each model simulation period was used only to initialize the model. That is, the model was run at least one extra year at the beginning of each period, without producing any output for that year, so that the model variables could be given sufficient time to stabilize and better reflect actual conditions.

The weather database furnished with the SWAT model was used to supply the SWAT weather generator with statistical weather information for the Green Bay NWS site. This information generates miscellaneous climatological data, such as rainfall intensity. Monthly average dew point temperatures from New London were added to this data set. New London is located approximately 55 km west, southwest of the Green Bay weather station (Figure 1-1). Dew point temperatures for the New London site were furnished with the EPIC model (Sharpley and Williams 1990). Monthly average wind speed data from Knox (1996) were added to the data set for the Green Bay NWS site. SWAT does not utilize measured snowfall inputs; consequently, snowfall was simulated by SWAT according to the measured precipitation amount and the average measured air temperature.

Subwatershed Climatological Assignment: Weather station locations were provided by the Wisconsin State Climatological Office. An ARC/INFO coverage containing these coordinates was created and daily precipitation and temperature data from the closest weather station was assigned to each subwatershed. An alternative method which relies on subwatershed-specific daily precipitation estimates from several nearby weather stations based on a distance formula, as previously employed by Marcus (1993), was not used here because of problems caused by time lags between actual events at different locations, and between the date that the precipitation was reported to have occurred at different locations (Baumgart 1998). The manner in which the temperature and precipitation data sets for each weather station were assigned to the subwatersheds in the subbasin is summarized in Table 2-1.

SOILS

Area-weighted values for soil parameters required by SWAT were created by processing digital soil surveys and tables as described in the sections that follow. This procedure was judged to be better than assuming that the soil parameters associated with the dominant soil series in a subwatershed are representative of an entire subwatershed.

Creation of Subbasin Soil Coverage: Digital GIS soil coverages and accompanying tables for four counties were obtained from the following sources: (1) Calumet County - National Resource Conservation Service Office in Madison, Wisconsin (NRCS); (2) Brown County - Brown County Planning Department; (3) Outagamie County - Outagamie County Planning Department; and (4) and Winnebago County Land Conservation Department. The locations of these counties are displayed in Figure 1-1. The digital soil surveys are based, in part, on previous NRCS work performed by Link et al. (1974), Barndt et al. (1978), Otter et al. (1980), and (Mitchell et al. 1980).

All of the GIS soil coverages used in this study were provisional. That is, they were not yet officially certified by the NRCS when they were originally processed in 1997 (Baumgart 1998), so they may contain minor errors which should not have any noticeable effect at the subwatershed scale. Subsequent comparisons between provisional coverages and final NRCS-approved soil coverages show no substantial differences. Soil coverages were supplied in an ARC/INFO export format. The export files were: (1) imported and built into PC ARC/INFO polygon coverages; (2) appended into a single county-wide coverage; (3) inspected for obvious errors and cleaned accordingly; (4) appended into a single four-county soils coverage; (5) "clipped" to produce a soil coverage that coincided with the subbasin outline; and (6) "cleaned" and "built" into a subbasin-wide polygon soils coverage with WTM, NAD83/91 coordinates (Wisconsin Transverse Mercator, North American Datum of 1983, HPGN). A "county" field was added to the ARC/INFO polygon table so that individual county NRCS soil tables could be linked to the combined ARC/INFO soils coverage.

Soil Table Processing: County-specific NRCS soil database tables contained information relating each soil series with the soil parameters required by SWAT (e.g., bulk density, available water capacity, etc.). Each of the county NRCS soil tables had to be processed separately because a soil series in one county does not necessarily have the same parameter values as the identically labeled soil series in another county. SWAT requires information for each soil layer that is associated with a soil series. The soil series in the subbasin had anywhere from 2 to 5 layers listed in the tables, and each layer has specific values assigned to the required soil parameters. Because area-weighted values were desired for each subwatershed, the different soil series depths were "normalized" by assuming that each soil series contained four layers. Parameter values for the deepest individual layer within each soil series were then extended to the remaining layers below it. Thus, if a soil series had only 3 layers, the parameter values associated with the last layer were assumed to extend to the required fourth layer below it. Soil layers were assumed to extend to the following depths: 1st = 203 mm, 2nd = 686 mm, 3rd = 1067 mm, and 4th = 1524 mm.[1] While this depth-normalizing procedure loses some information, it seemed appropriate given that there were over 100 soil series in the Outagamie County portion of the subbasin alone.

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[1] Adding an extra layer to a Kewaunee soil series file increased the simulated sediment yield of a test watershed only slightly (42.3 vs 43.0 t/ha), thereby indicating that the process of adding an additional soil layer was not likely to affect sediment loads.

Subwatershed-specific Soils Determination: Area-weighted average values for soil parameters including: USLE K-factor, NRCS hydrologic group, available water capacity, saturated conductivity, clay percentage, organic carbon and bulk density were generated for each subwatershed according to the procedure described by Baumgart (1998). LOTUS 1-2-3 was used to create a script (macro) that automatically exported subwatershed-specific soil files (e.g., LF0205.sol) into a SWAT compatible text file format for all 65 subwatersheds in a single operation. This operation was also done for routing reach, hydrologic response unit and subwatershed input files.

Soil Phosphorus Levels: Soil phosphorus (P) test data for the four counties in the subbasin including Brown, Calumet, Outagamie and Winnebago were provided by the University of Wisconsin Soil Testing Lab. The cumulative frequency distribution of soil Bray-P test levels (ppm) for the four counties in the Lower Fox Subbasin are summarized in Table 2-2 for the 1995-99 period. A mean of 41 ppm, median of 31 ppm and a maximum of 679 ppm were calculated for the 33,452 data points in this data set. Approximately 74% of all soil samples were below the 50 ppm soil-test phosphorus criteria for surface water protection under the Chapter 590 Nutrient Management Standard. This leaves roughly 26% of the soil samples above the criteria. However, since the whole field will be averaged, it is safe to say that less than 26% of the tested fields would not meet the criteria. About 90% of the soil samples tested below 80 ppm; whereas, roughly 6% tested above 100 ppm and 2% tested above 150 ppm.

Table 2-2. Soil Test Phosphorus Cumulative Frequency Distribution.

Soil Bray-P1 (ppm)	% <	% >
0	0.0%	100.0%
10	7.6%	92.4%
20	29.0%	71.0%
30	48.6%	51.4%
40	63.6%	36.4%
50	74.1%	25.9%
60	81.3%	18.7%
70	86.4%	13.6%
80	89.9%	10.1%
90	92.3%	7.7%
100	94.2%	5.8%
150	98.0%	2.0%
200	99.4%	0.6%
300	99.9%	0.1%
700	100.0%	0.0%

In the 1992 and 2000 Baseline Scenarios, model input soil levels were based on the average 1995 to 1999 soil-test phosphorus concentration of 40 ppm (Bray-P1) in Brown County (Combs and Peters 2000).

Soil Phosphorus Levels - Effect on Stream Phosphorus Levels: An attempt was made to see if there was a relationship between relatively high measured concentrations of phosphorus found in streams within the subbasin, and the levels of phosphorus found in the soil (as Bray-P1). To investigate this question, the linear relationship between soil Bray-P1 levels and dissolved phosphorus concentrations in runoff from 0.83 sq. meter plots that was observed by Andraski and Bundy (2003) for a Manawa silty clay loam soil in Fond du Lac County was applied to the aforementioned data set (dissolved P in runoff = -0.08 + 0.012 * soil Bray-P1;

$r^2 = 0.66$; $p<0.01$). Applying this regression equation to the soil test data from all four counties produced an expected average dissolved phosphorus concentration in runoff of 0.42 mg/L from a similar plot and experimental design. Perhaps not coincidently, this value is fairly close to the observed average values of dissolved phosphorus (or ortho-P) that have been measured in streams from watersheds whose soils are somewhat similar to that found in Fond du Lac County. For example, the average dissolved phosphorus (or ortho-P) concentration in rural streams draining watersheds with less permeable soils within the Lower Fox River Subbasin is about 0.41 mg/L.[2] Substituting the regression equation that Andraski and Bundy (2003) determined for plots at Lancaster and Arlington gives a much lower dissolved phosphorus concentration of 0.11 mg/L in runoff from plots with a similar experimental design.

Of course, extrapolating results from the relationships that were found in small plots to what might be expected at the field edge or in a stream is questionable. However, the fraction of dissolved phosphorus found in streams within the Lower Fox River Subbasin are higher than anticipated based upon the literature. Andraski and Bundy (2003) also found that the fraction of dissolved phosphorus in runoff was greater at the Fond du Lac site (dP = 17%), where the clay content was higher than at the other two sites (dP = 5%). Preferential deposition of phosphorus attached to larger sediment particles is expected to occur along the transport path from the source to the stream and within the stream itself, thereby increasing the proportion of dissolved phosphorus from that which occurred at the source.

Soil Phosphorus Levels - Alternative Scenarios: Two alternative scenarios were performed which evaluated the effect of soil phosphorus levels that are different than current levels. In the first scenario, it was assumed that the modeled input soil level was 25 ppm, which was the average soil-test P level of soil samples tested in 1974-79 within three counties (Brown 27 ppm; Calumet 23 ppm; Outagamie 25 ppm), but excluding the 32 ppm average in Winnebago County (Combs et al. 1996). This soil-test P level seemed reasonable as it was achieved in the past. In addition, Bundy and Sturgul (2001) reported that the statewide average soil-test P levels rose from 29 ppm (1964-67) to 36 ppm (1974-77), and then to 40 ppm (1978-81), so the average level of soil-test P assumed for this alternative scenario could have been even lower if the Lower Fox Subbasin followed the statewide increasing trend between these periods. According to Kelling et al. (1998), optimum soil-test P for soils in the Lower Fox River Subbasin, which are classified as subsoil fertility group C, are between 16 and 20 ppm for growing high demand crops like corn, 6 to 12 for soybean, and 18 to 25 ppm for alfalfa. When soil-test P levels are optimum, it is recommended that phosphate be added at harvest removal rates (Kelling et al. 1998). Hence, cash crop operations with mainly a corn-soybean rotation may to be able to get by at an even lower level of 20 ppm.

The second scenario was performed with an assumed average concentration of 50 ppm, which is the highest level permitted under NR 151 before nutrient management actions must be taken to reduce or maintain the soil-test P value for a field (unless the P-Index for the field is acceptable). Under these two scenarios, phosphorus inputs were adjusted so that phosphorus in the form of manure or commercial fertilizer was applied at rates sufficient to meet crop removal needs. Otherwise, phosphorus levels in the soil would increase over time if applied phosphorus exceeded harvest removal rates and other losses.

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[2] Monitored sites included in the average include: South Ashwaubenon at Noah Road, North Ashwaubenon at CTH U, South Apple, North Apple, Dutchman at CTH U, East River at CTH Z and Midway, Dutchman at CTH U, Baird at Northview, Apple at CTH D, Plum at CTH D, Kankapot at CTH CE (based on USGS, WDNR and Fox-Wolf Basin data summarized by the author, unpublished data 2003).

OVERLAND SLOPE AND SLOPE LENGTH

The statewide 30-meter digital elevation model (DEM) was clipped to the watershed boundary and intersected/combined with both the Wisconsin Initiative for Statewide Cooperation on Landscape Analysis and Data (WISCLAND) land cover image (used to delineate hydrologic response units or HRU's), and the subwatershed delineated GIS layer, to produce area-weighted overland slope averages that were specific to each of the seven major groups of HRU's within each of the 65 subwatersheds. In this way, each major HRU group within a subwatershed (agriculture and barnyards, urban, grassland, forest, wetland, golf course, barren), was assigned an area-weighted average slope value that was specific to both the HRU group and the subwatershed. Overland slopes in the subbasin are shown in Figure 2-1 as a percentage. Most of the subbasin has fairly shallow slopes. Notable exceptions include the Niagara escarpment in the East River watershed and areas adjacent to major stream channels. Slopes derived at different scales such as from field measurements, higher resolution DEM's, or published in soil surveys may be substantially different than the values used in this project. However, the relative differences in slope between the subwatersheds or HRU's ought to be fairly similar, regardless of the methodology used to derive slope.

Importantly, the current ARCVIEW interface for SWAT, and the BASINS interface, both assign the same slope to all HRU's within a subwatershed. For example, the interface does not compute different slopes for landuses such as wetland, crop land or forest that happen to be in the same subwatershed. For this project, HRU-specific information was deemed important because slopes often vary between different landuses. For example, the average slope of agricultural land in the subbasin is 1.96%; whereas, the average slope of forested land is much more steep (3.92%). This level of analysis is even more critical where there is a large proportion of wetlands, for which the average slope would substantially reduce the slope of the other HRU's in the subwatershed (unless other procedures are taken to exclude slope values from wetland areas).

The 30 meter resolution of the DEM did not permit the direct calculation of slope length. Instead, Equation 1 was used to calculate slope length on the basis of an empirical relationship between slope and slope-length.

$$\text{slope length (in meters)} = 91.4 \text{ meters} / (\% \text{ slope} + 1)^{0.4} \qquad \text{(Eq. 1)}$$

This equation was set to conform closely with soil map default values when measured slope lengths are not available. The SWAT ARCVIEW interface employed in BASINS increases the slope length according to several slope intervals, so the approach used here seems reasonable. Within SWAT, slope length and slope are used in the Modified Universal Soil Loss Equation (MUSLE) to generate sediment loads (see Equation 9 in Chapter 6). Therefore, errors in slope length can have a major impact on simulated sediment loads. However, relative changes in slope length generally have 50% or less of an effect on sediment loads compared to the same relative changes in slope (Williams and Berndt 1975).

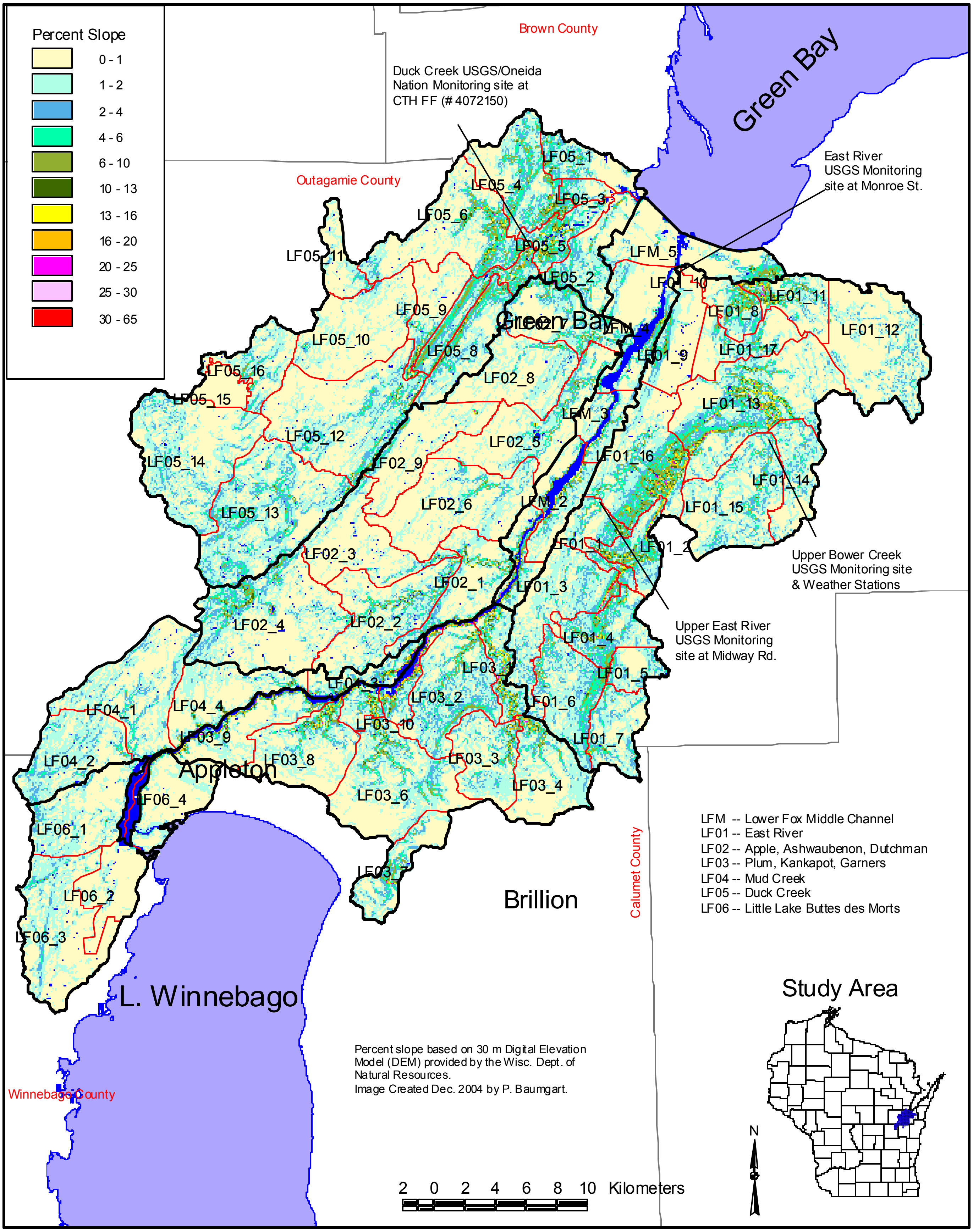

Figure 2-1. Percent overland slope in Lower Fox River subbasin. Based on 30 m Digital Elevation Model (DEM) provided by the WDNR.

GENERAL LAND USE AND MODELING METHODS

Seven major land use categories were modeled with methods that are described in greater detail later in this section. The WISCLAND classified land cover image was primarily used to assign 7 major land covers/uses which were modeled within the watershed: agriculture, urban, golf course, forest, grassland, wetland and barren. These land covers were further divided into 13 major groups of hydrologic response units (HRU's) which were directly modeled in the following fashion:

Agriculture - Dairy
1 Conventional tillage practice (CT)
2 Mulch-till (MT30)
3 No-till (NT)
Agriculture - Cash crop
4 Conventional tillage practice (CT)
5 Mulch-till (MT30)
6 No-till (NT)

7 Grassland
8 Forest
9 Wetland
10 Barren
11 Urban
12 Barnyard
13 Golf course

HRU's represent areas within a subwatershed that are similar in a hydrologic or management sense, but are not necessarily contiguous. For this project, HRU's are the total area in the subwatershed with a particular land use and/or management. No single specific farming practice could be used to model the entire watershed; therefore, various proportions of six possible agricultural practices (6 major HRU's) were used to simulate what occurred in each subwatershed. For simplicity, every subwatershed was modeled as though it contained the 13 major HRU's in the order shown above. However, in order to simulate the crop rotations that were modeled (dairy: corn, alfalfa and soybean; cash crop: corn and soybean), additional HRU's were required so that all phases of a crop rotation could be simulated in a single model run. Alternatively, separate model runs would be required to simulate each phase of a crop rotation. Since there were 6 years in a dairy rotation, 3 years in a cash crop rotation, and 65 subwatersheds, the total number of modeled HRU's was 2210 [65 subwatersheds * (6 years * 3 tillage practices + 3 years * 3 tillage practices + 7 other landuses)]. A GIS overlay operation was used to derive the proportional area of the major HRU's within each of the 65 modeled subwatersheds. The next section describes how the agricultural areas were further divided into 6 agricultural HRU's. Where a subwatershed did not contain all of the landuses, the area of the non-existent land use was assigned a negligible value (0.0000001).

Areal land cover and landuse for each of the subwatersheds in the subbasin are summarized in Table 2-3 for the 1992 Baseline Scenario and Table 2-4 for the 2000 Current Scenario (without water), which is also illustrated in Figure 2-2.

Table 2-3. Percent Land Cover and Landuse in the Lower Fox River Subbasin: 1992.

subwatershed	urban	ag	grassland	forest	wetland	barren	golf
LF01-01	5.2%	74.7%	2.9%	13.7%	2.8%	0.6%	0.0%
LF01-02	3.6%	71.0%	2.7%	22.1%	0.5%	0.1%	0.0%
LF01-03	2.9%	83.8%	2.1%	9.9%	0.9%	0.4%	0.0%
LF01-04	4.4%	73.6%	2.1%	15.7%	2.7%	1.6%	0.0%
LF01-05	1.0%	57.7%	2.2%	11.2%	27.3%	0.4%	0.0%
LF01-06	2.3%	74.5%	2.5%	19.8%	0.5%	0.4%	0.0%
LF01-07	0.9%	69.4%	3.1%	11.0%	14.8%	0.7%	0.0%
LF01-08	79.4%	11.6%	0.2%	8.2%	0.0%	0.6%	0.0%
LF01-09	63.2%	20.4%	0.4%	9.7%	4.0%	2.4%	0.0%
LF01-10	98.0%	0.6%	0.0%	1.4%	0.0%	0.0%	0.0%
LF01-11	8.7%	54.0%	1.1%	32.0%	0.9%	3.2%	0.0%
LF01-12	3.4%	74.1%	2.9%	8.8%	10.2%	0.5%	0.0%
LF01-13	14.7%	61.9%	1.2%	16.8%	1.3%	4.0%	0.0%
LF01-14	1.6%	88.6%	1.9%	5.7%	1.4%	0.8%	0.0%
LF01-15	2.3%	86.7%	1.2%	7.3%	1.6%	1.0%	0.0%
LF01-16	16.1%	58.5%	2.2%	18.6%	1.7%	3.0%	0.0%
LF01-17	43.4%	41.3%	0.6%	9.2%	0.3%	5.2%	0.0%
LF02-01	9.3%	75.9%	2.3%	7.3%	0.8%	1.3%	3.1%
LF02-02	19.7%	68.8%	0.7%	7.9%	1.8%	1.2%	0.0%
LF02-03	5.7%	84.7%	1.7%	3.8%	1.0%	0.9%	2.2%
LF02-04	21.3%	68.7%	1.1%	5.9%	1.3%	1.7%	0.0%
LF02-05	26.8%	54.5%	4.7%	10.1%	1.1%	2.8%	0.0%
LF02-06	5.7%	83.2%	1.7%	8.1%	0.5%	0.9%	0.0%
LF02-07	61.9%	18.4%	3.2%	11.4%	2.8%	2.4%	0.0%
LF02-08	16.3%	68.8%	4.5%	8.6%	1.0%	0.8%	0.0%
LF02-09	1.6%	87.2%	1.5%	7.7%	1.3%	0.6%	0.0%
LF03-01	5.6%	58.2%	1.1%	34.7%	0.0%	0.4%	0.0%
LF03-02	2.4%	88.0%	1.2%	6.9%	0.3%	1.2%	0.0%
LF03-03	2.6%	87.2%	1.1%	5.9%	0.6%	2.5%	0.0%
LF03-04	0.5%	85.8%	2.4%	8.6%	1.0%	1.7%	0.0%
LF03-05	73.8%	14.3%	0.0%	9.3%	0.2%	2.4%	0.0%
LF03-06	2.3%	88.0%	1.6%	4.4%	1.4%	2.2%	0.0%
LF03-07	0.0%	75.8%	2.5%	3.3%	17.2%	1.1%	0.0%
LF03-08	31.2%	59.2%	1.0%	4.0%	1.6%	3.0%	0.0%
LF03-09	85.6%	10.7%	0.0%	1.6%	0.1%	2.0%	0.0%
LF03-10	10.4%	57.4%	0.4%	25.5%	2.8%	3.4%	0.0%
LF04-01	61.0%	27.5%	1.2%	5.2%	0.5%	2.6%	2.1%
LF04-02	29.0%	53.4%	3.2%	8.4%	2.0%	3.9%	0.0%
LF04-03	72.0%	16.1%	1.5%	8.7%	0.0%	1.7%	0.0%
LF04-04	90.4%	5.6%	0.3%	2.1%	0.0%	1.6%	0.0%
LF05-01	24.8%	37.3%	0.1%	12.5%	19.8%	5.4%	0.0%
LF05-02	76.7%	5.7%	0.4%	9.3%	4.7%	3.2%	0.0%
LF05-03	50.0%	15.5%	0.3%	9.6%	18.3%	6.3%	0.0%
LF05-04	21.3%	52.5%	0.3%	20.9%	3.5%	1.4%	0.0%
LF05-05	35.3%	26.5%	2.6%	21.2%	2.3%	2.9%	9.2%
LF05-06	11.7%	56.2%	1.0%	18.0%	11.5%	0.6%	1.1%
LF05-07	8.1%	44.4%	2.7%	35.6%	6.4%	1.2%	1.6%
LF05-08	8.2%	68.9%	2.1%	15.9%	3.4%	1.5%	0.0%
LF05-09	2.7%	57.5%	1.6%	24.5%	13.4%	0.3%	0.0%
LF05-10	2.0%	75.5%	2.5%	12.6%	7.1%	0.3%	0.0%
LF05-11	3.2%	84.1%	1.4%	7.9%	2.9%	0.5%	0.0%
LF05-12	3.2%	72.4%	1.6%	17.3%	5.1%	0.4%	0.0%
LF05-13	6.4%	79.0%	2.9%	6.9%	3.3%	1.6%	0.0%
LF05-14	3.5%	79.4%	2.5%	5.5%	8.9%	0.3%	0.0%
LF05-15	0.0%	1.0%	0.0%	3.6%	95.3%	0.0%	0.0%
LF05-16	2.0%	84.9%	3.5%	6.2%	3.2%	0.1%	0.0%
LF06-01	47.9%	39.2%	1.8%	5.9%	2.5%	2.6%	0.1%
LF06-02	82.6%	8.8%	0.1%	2.3%	0.4%	4.0%	1.7%
LF06-03	9.2%	70.3%	1.8%	10.1%	5.2%	3.3%	0.0%
LF06-04	82.0%	9.7%	0.0%	2.9%	2.3%	3.2%	0.0%
LFM1-01	17.3%	74.4%	0.8%	7.1%	0.0%	0.3%	0.0%
LFM1-02	15.6%	68.2%	1.9%	12.1%	0.9%	1.4%	0.0%
LFM1-03	69.7%	22.4%	2.2%	4.4%	0.0%	1.4%	0.0%
LFM1-04	89.4%	3.7%	0.8%	4.8%	0.1%	1.3%	0.0%
LFM1-05	61.8%	10.8%	0.0%	15.4%	7.4%	4.5%	0.0%
TOTAL	**21.7%**	**60.3%**	**1.7%**	**10.4%**	**3.9%**	**1.7%**	**0.3%**

Table 2-4. Percent Land Cover and Landuse in the Lower Fox River Subbasin: 2000.

subwatershed	area (km2)	urban	ag	grassland	forest	wetland	barren	golf
LF01-01	12.4	10.5%	63.8%	7.1%	13.7%	3.2%	0.4%	1.2%
LF01-02	12.3	7.6%	58.4%	6.4%	22.1%	0.5%	0.1%	4.8%
LF01-03	31.8	5.5%	81.3%	2.1%	9.9%	0.9%	0.3%	0.0%
LF01-04	17.9	7.7%	69.9%	2.1%	15.7%	2.7%	1.9%	0.0%
LF01-05	8.3	1.6%	57.1%	2.2%	11.2%	27.4%	0.4%	0.0%
LF01-06	24.6	4.1%	72.7%	2.5%	19.9%	0.5%	0.4%	0.0%
LF01-07	16.9	1.7%	68.0%	3.1%	11.0%	14.8%	1.4%	0.0%
LF01-08	10.2	90.5%	1.1%	0.2%	8.2%	0.0%	0.0%	0.0%
LF01-09	17.7	76.4%	8.5%	0.4%	9.8%	4.0%	0.1%	0.8%
LF01-10	1.3	98.5%	0.1%	0.0%	1.4%	0.0%	0.0%	0.0%
LF01-11	14.5	16.2%	45.7%	1.1%	32.0%	0.9%	0.6%	3.4%
LF01-12	43.2	7.0%	68.5%	2.9%	8.8%	12.2%	0.3%	0.3%
LF01-13	55.5	27.6%	47.9%	1.2%	16.9%	1.4%	2.5%	2.6%
LF01-14	17.7	3.0%	86.8%	2.2%	5.7%	1.6%	0.7%	0.0%
LF01-15	35.9	4.6%	83.2%	2.3%	7.3%	1.8%	0.9%	0.0%
LF01-16	37.4	30.3%	44.4%	2.2%	18.6%	2.0%	2.5%	0.0%
LF01-17	16.1	65.6%	24.0%	0.6%	9.2%	0.3%	0.3%	0.0%
LF02-01	22.7	15.8%	69.8%	2.3%	7.4%	0.8%	0.3%	3.6%
LF02-02	34.8	30.3%	59.2%	0.7%	8.0%	1.8%	0.1%	0.0%
LF02-03	28.6	10.6%	80.1%	1.7%	3.9%	1.0%	0.4%	2.3%
LF02-04	53.8	35.5%	55.3%	1.1%	5.9%	1.3%	0.9%	0.0%
LF02-05	34.7	38.8%	45.1%	4.7%	10.2%	1.2%	0.0%	0.0%
LF02-06	41.3	11.0%	78.5%	1.7%	8.2%	0.5%	0.2%	0.0%
LF02-07	23.4	74.8%	7.5%	3.2%	11.4%	3.0%	0.1%	0.0%
LF02-08	27.6	28.6%	54.8%	4.5%	8.6%	1.1%	0.3%	2.1%
LF02-09	26.6	3.1%	85.1%	1.6%	7.8%	1.3%	1.1%	0.0%
LF03-01	12.4	12.1%	51.6%	1.1%	34.9%	0.0%	0.3%	0.0%
LF03-02	34.0	4.6%	84.5%	1.2%	6.9%	0.3%	1.0%	1.5%
LF03-03	21.3	5.0%	85.4%	1.1%	5.9%	0.6%	2.0%	0.0%
LF03-04	25.3	0.9%	85.3%	2.4%	8.6%	1.0%	1.7%	0.0%
LF03-05	6.3	88.9%	1.3%	0.0%	9.6%	0.2%	0.0%	0.0%
LF03-06	44.2	4.2%	87.0%	1.6%	4.4%	1.4%	1.4%	0.0%
LF03-07	16.2	0.0%	75.8%	2.5%	3.3%	17.2%	1.1%	0.0%
LF03-08	28.6	49.0%	43.0%	1.0%	4.0%	1.6%	1.3%	0.0%
LF03-09	21.7	94.3%	0.0%	0.0%	1.6%	0.1%	0.4%	3.5%
LF03-10	7.8	22.1%	42.6%	0.4%	26.5%	2.9%	0.3%	5.1%
LF04-01	39.4	73.1%	17.5%	1.2%	5.3%	0.5%	0.3%	2.2%
LF04-02	27.6	43.4%	39.6%	3.2%	8.4%	2.1%	3.4%	0.0%
LF04-03	10.4	76.2%	12.0%	1.5%	8.9%	0.0%	1.4%	0.0%
LF04-04	24.6	94.0%	3.0%	0.3%	2.1%	0.0%	0.5%	0.0%
LF05-01	13.4	49.5%	17.2%	0.1%	12.6%	19.9%	0.7%	0.0%
LF05-02	18.8	84.7%	0.6%	0.4%	9.3%	4.9%	0.1%	0.1%
LF05-03	4.6	66.9%	2.2%	0.3%	9.6%	18.4%	2.6%	0.0%
LF05-04	28.5	38.9%	35.8%	0.3%	17.2%	4.0%	0.5%	3.3%
LF05-05	11.5	58.9%	6.6%	2.7%	15.8%	2.4%	0.0%	13.7%
LF05-06	39.1	21.8%	46.8%	1.1%	16.4%	11.5%	0.6%	1.8%
LF05-07	13.3	18.6%	31.4%	3.4%	35.7%	6.8%	0.9%	3.2%
LF05-08	19.4	16.9%	59.0%	2.8%	16.0%	4.0%	1.4%	0.0%
LF05-09	15.5	6.1%	53.9%	1.8%	24.6%	13.4%	0.1%	0.0%
LF05-10	35.5	3.7%	73.5%	2.6%	12.7%	7.1%	0.5%	0.0%
LF05-11	10.5	6.0%	80.7%	1.5%	8.0%	2.9%	0.9%	0.0%
LF05-12	53.5	6.0%	69.1%	1.7%	17.5%	5.1%	0.6%	0.0%
LF05-13	55.4	11.9%	71.8%	2.9%	6.9%	3.3%	2.2%	1.0%
LF05-14	49.3	6.5%	76.4%	2.5%	5.6%	8.9%	0.2%	0.0%
LF05-15	7.6	0.0%	1.1%	0.0%	3.6%	95.3%	0.0%	0.0%
LF05-16	16.0	3.6%	83.2%	3.6%	6.3%	3.3%	0.1%	0.0%
LF06-01	28.0	60.4%	26.4%	1.8%	6.0%	2.5%	0.8%	2.1%
LF06-02	17.5	92.8%	2.9%	0.1%	2.4%	0.4%	0.1%	1.3%
LF06-03	41.5	17.9%	62.3%	1.8%	10.2%	5.3%	2.6%	0.0%
LF06-04	21.5	93.3%	0.0%	0.0%	2.9%	2.3%	1.5%	0.0%
LFM1-01	10.6	31.0%	60.6%	0.8%	7.2%	0.0%	0.3%	0.0%
LFM1-02	17.8	30.2%	51.5%	5.0%	12.1%	0.9%	0.3%	0.0%
LFM1-03	11.2	82.0%	11.4%	2.2%	4.4%	0.0%	0.1%	0.0%
LFM1-04	26.6	93.6%	0.6%	0.8%	4.8%	0.1%	0.0%	0.0%
LFM1-05	27.5	76.5%	0.6%	0.0%	15.5%	7.4%	0.0%	0.0%
TOTAL	1,581.1	29.5%	52.6%	1.9%	10.3%	4.0%	0.9%	0.8%

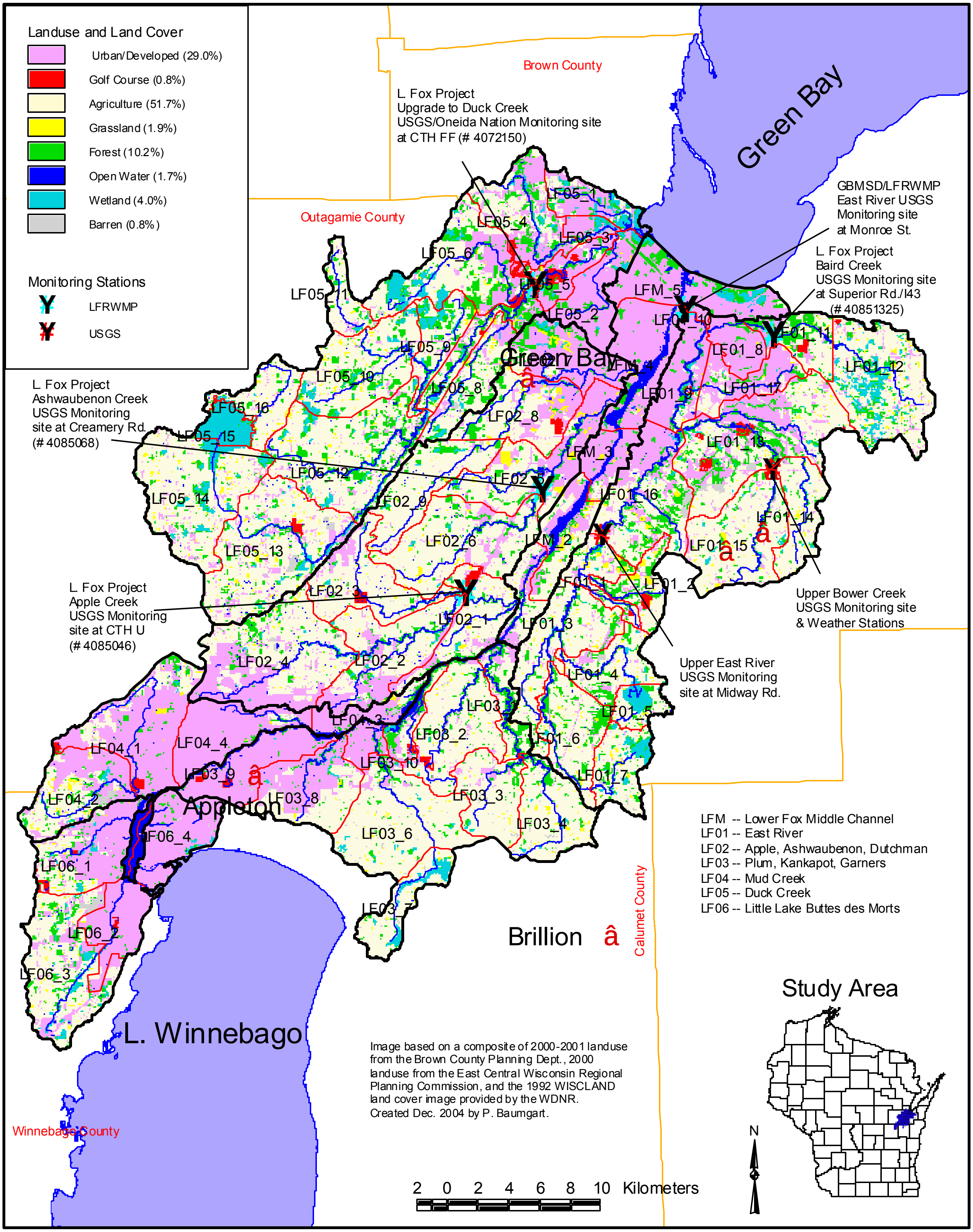

Figure 2-2. Year 2000 landuse and land cover in the Lower Fox River Subbasin. Composite based on images from Brown County Planning Dept., WDNR, and ECWRPC.

Derivation of the underlying GIS coverages is described in the following sections. With the exception of LF01-5 and LF05-15, which have a high proportion of wetlands, most of the subwatersheds are predominantly urban or agricultural. Agricultural land cover was the most prevalent land cover in the subbasin. Wetlands, grasslands and forested areas are relatively small components of the subbasin compared to urban and agricultural areas.

Land Cover Analysis with WISCLAND Classified Land Cover Image: With some major exceptions, land cover within the subbasin was primarily determined from the Level 3 classification of the 1992 WISCLAND land cover image, which was obtained from the WDNR and is based on LANDSAT Thematic Mapper images. The WISCLAND classified land cover image was reclassified to generate land covers/uses which were modeled with SWAT: agriculture (corn, forage, other row crops), urban, grassland, forest, wetland, golf course, barren and water. Major surface water areas were excluded in the model simulations. For this project, it was assumed that "other row crops" was either soybean or another fragile crop, so this land cover was simulated as soybean in the SWAT model.

2000 Landuse and Urban Areas: To create a year 2000 subbasin-wide landuse coverage, and the associated model inputs for the 2000 Scenario (Baseline and alternatives), the Brown County GIS landuse coverage, based on 2000 imagery and developed by the Brown County Planning Department, and the East Central Wisconsin Regional Planning Commission (ECWRPC) GIS landuse coverage (without Calumet County), also based on 2000 imagery, were both merged with the 1992 WISCLAND coverage. An updated landuse coverage was not yet available for the Calumet County portion of the subbasin, including subwatersheds LF01-6,7 and LF03-8,3,4,6,7 (order in increasing areal proportion within Calumet County). Therefore, a 1999 road map of the Fox Cities area was used to extend the 1992 urban areas delineated by both the 1992 WISCLAND image and 1992 Digital Raster Graphic 1:24,000 USGS topographic images to estimate urban landuse in 2000 within the Calumet County area. All urban classes including the developing classification were grouped into a single urban class. The resulting assumed landuse for the 2000 Scenario is shown in Figure 2-2.

1992 Landuse and Urban Areas: Although the WISCLAND classified land cover GIS coverage was used to classify the initial extent of urban areas for the simulated 1992 Baseline Scenario, modifications were necessary because the proportion of urban area was under-estimated. Visual inspection of the WISCLAND coverage showed large areas near or within urban centers were classified as grassland. Although some of these grassland areas may have been grass or developing areas, all urban landuses were classified as a single urban class in this project. Therefore, all such areas were instead assumed to be urban unless shown as grass in the 2000 Scenario final coverage. Thus, the proportion of grassland was decreased to the level determined for the 2000 Scenario, and urban areas were increased proportionately to balance this change. In addition, the proportion of urban area within each subwatershed was further increased to coincide more closely with the total urban area estimated by Baumgart (1998) for 1992 using 1:24k DRG images. The difference between the non-adjusted 1992 urban area and the 2000 urban area within each subwatershed, was multiplied by an adjustment factor of 1.4 to obtain the absolute amount of the increase in each subwatershed. The increase in urban area was balanced by a proportional decrease in agricultural landuse. Without these adjustments to the 1992 WISCLAND-delineated urban area, the urban fraction within the subbasin would have appeared to increase from 12% in 1992 to 29.5% in 2000. With some exceptions, the proportion of urban area indicated by the 1992 Baseline Scenario had little effect on model calibration and validation because landuse in the affected watersheds was nearly all rural.[3]

The ability of the SWAT model to simulate stream flow and phosphorus and sediment export from areas with different land uses was considered an important factor in determining which maps and/or GIS coverages would be used to distinguish major land uses. While the Source Loading and Management Model (SLAMM, Pitt and Voorhees 1995) and the Stormwater Management Model (SWMM, Huber and Dickinson 1988) were specifically designed to estimate pollutant loads and runoff from urban areas, the SWAT model is not normally intended solely for this purpose. Therefore, a detailed breakdown into many urban classes would not likely improve the accuracy of the simulation of export from urban areas with the SWAT model. This simplification seems reasonable given the highly variable nature of suspended sediment loads from urban areas (Bannerman et al. 1996, Steuer et al. 1996 and 1997, Owens et al. 1997). In addition, the scale of the subbasin and the current dominance of rural land use precluded placing a major emphasis on modeling urban loads. Instead, emphasis was placed on primarily distinguishing the location of the boundary between urban and rural land uses, and the changes that have occurred or are predicted.

Non-agricultural Rural Land Areas: Non-agricultural rural areas were modeled in the following manner. HRU's designated as grassland, forest, wetlands and golf courses were assigned values from SWAT's default crop data sets for pasture, forest, wetland and lawn data sets, respectively. Fertilizer was only added to areas designated as golf courses. Those areas that were classified as barren in the WISCLAND image could be farm lots, quarries, bare lots or other landuses. The impact of a quarry on surface water quality could be quite different from a farm lot. Because the areal extent of this classified land cover is small, and nearly always in a rural setting, it was treated as agriculture so it was essentially lumped together with nearby agricultural land.

Forested Areas: For the 2000 Scenario, the Brown County and ECWRPC 2000 landuse images were utilized to supplement the forested land cover class indicated by the 1992 WISCLAND image. The 1992 WISCLAND image understated the proportion of forested areas, sometimes significantly so, especially along narrow riparian corridors. Therefore, for consistency, the same proportions of forested areas were also used for the 1992 Baseline Scenario, rather than relying solely on the 1992 WISCLAND image to indicate forested areas. A comparison between 1992 digital orthophotos and areas indicated as forested or natural areas by the 2000 Brown County and ECWRPC coverages shows that this process may overstate the actual amount of forested area present in 1992, and may therefore understate the amount of agricultural land in rural areas. However, the errors are substantially less than if only the WISCLAND image had been used to classify forested areas in 1992. A landuse coverage was not yet available for the year 2000 in Calumet County, so within this county, only areas classified as forest in the 1992 WISCLAND image were considered forest for all scenarios, thereby understating the actual amount of forest likely to be present.

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[3] Increasing the areal extent of urban land resulted in an increase in the manure application rate because the amount of available land to spread the manure on decreased.

Urban Growth Trend: To determine the rate of urban development, ARCVIEW was used to digitize urban boundaries within the subbasin to create coverages representing four time periods: approximately 1954-55, 1971-74, 1982-84 and 1992. The simplest and most consistent method of determining land use trends was with USGS 1:24,000 and 1:62,500 topographic maps and images ranging from published dates of 1954 to 1992. Some of the other sources of land use and land cover previously mentioned were used to supplement these maps and images when data were missing. The 2000 landuse coverage that was previously described was added to these layers to create the image displayed in Figure 2-3 which shows the estimated increase in urban area from 1954 to 2000. The estimated urban land use and percent urban area for the following years is:

1954 ----- 130 km^2 (8.2%)
1974 ----- 217 km^2 (13.7%)
1984 ----- 277 km^2 (17.7%)
1992 ----- 340 km^2 (21.6%)
2000 ----- 460 km^2 (29.5%; urban area based on different landuse analysis method)

A 7.9 km^2 average annual increase in urbanized area was estimated for the period between 1984 and 1992, whereas urbanization between 1954 and 1992 was estimated to occur at a rate of 5.5 km^2/year. The most recent estimated average annual rate of urbanization was 15 km^2 between 1992 and 2000. Urban areas within the subbasin had been growing at a fairly consistent rate of 2.6% per year between 1954 and1992. If this rate were to continue, the amount of urban area within the subbasin will double every 27 years. However, urban areas seem to have increased at a higher 3.8% annual rate between 1992 and 2000; but, some of this increase is due to a difference in methodologies. That is, the 2000 landuse coverage included many rural residential lots that were classified as urban; whereas, this was not done for the other years. Some small rural communities were also not included in the 1954, 1974, 1984 and 1992 GIS layers that were created to track urban growth trends.

Development of Alternative Scenario Where Urban Area Doubles: An alternative scenario that investigates the impact of doubling the urban area within the subbasin was developed by simply expanding the areal extent of existing urban areas with the GIS. The GIS layer that was created to simulate the approximate expanded areal coverage of urbanized areas within the subbasin under this scenario is shown in Figure 2-3, in addition to landuse trends from 1954 to 2000. This coverage was used to provide the required inputs to the SWAT model in the same manner that the 2000 landuse layer was utilized for the 2000 Current Scenario. Projected urbanized areas shown in Figure 2-3 are only rough approximations of where they might be when the urban area doubles compared to the 2000 Current Scenario. While the precise location of future urbanized areas is likely to be different than that depicted in the figure, the effect of spatial errors on the modeled output is expected to be relatively small compared to potential errors caused by faulty assumptions, such as urban loading rates. If long-term rates of urbanization continue, it is estimated that it will take until 2025-2030 to roughly double the urban area within the subbasin, at which time urban areas would comprise about 55-58% of the subbasin.

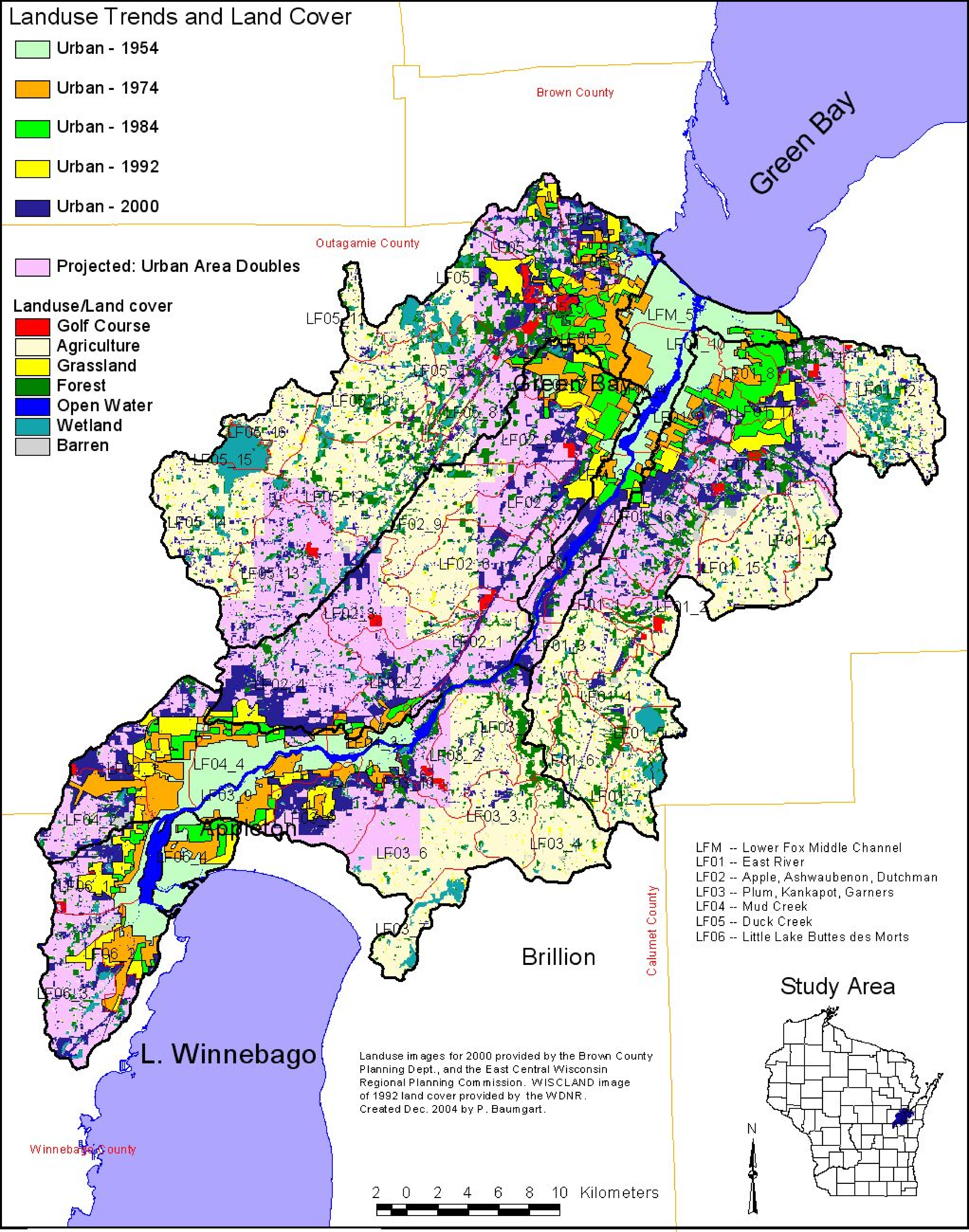

Figure 2-3. Lower Fox River Subbasin - Trends in urban landuse from 1954 to 2000, and projected extent of urban landuse when urban area doubles (approx. 2025 to 2030).

AGRICULTURAL LANDUSE AND MODELING METHODS

SWAT requires detailed information regarding land use management practices. For example, the type of crop, the date it was planted and harvested, tillage practices and dates, fertilizer applications and dates, and NRCS curve number for each period, are just some of the information that is input into SWAT's management files. This section describes how these inputs were obtained.

Agricultural Crop and Tillage Methodology: Two types of agricultural crop rotations were simulated: dairy and cash crop. A three year crop rotation consisting of two years of corn-grain followed by one year of soybean was used to simulate cash crop operations under 1992 Baseline conditions. In this rotation, soybean was assumed to represent other crops such as wheat or oats grown solely as a grain crop. Nitrogen is not a constituent of concern in this project, so differing application rates of nitrogen for the different cash crops did not cause a major problem in the model. A six year crop rotation that is typical for dairy farm operations in Northeastern Wisconsin was used to simulate dairy operations. The six year rotation consisted of: (year 1) corn grain; (year 2) corn silage; (years 3-6) alfalfa. A nurse crop such as oats was not "grown" in the third year because SWAT cannot simulate two crops growing simultaneously. The method previously used by the author of having an oats crop followed by alfalfa poses a problem whereby the just-planted alfalfa does not have sufficient protective cover present after the oats is harvested.

The agricultural HRU's consisted of two potential farming practices under 1992 Baseline conditions:

1) Dairy-based (6 year rotation: corn grain, corn silage, alfalfa, alfalfa, alfalfa, alfalfa)
2) Cash crop (3 year rotation: corn grain, corn grain, soybean).

Sugiharto et al. (1992, 1994) and McIntosh (1993a, 1993b, 1994) found that the six year dairy crop rotation and associated management practices were typical of the average dairy operations in the East River Watershed. This rotation was also recommended by University of Wisconsin (UW) Extension Agricultural Agent Kevin Erb (personal comm. 1997). The Outagamie County Land Conservation Department (LCD) used this same rotation to represent the crop patterns of the Duck, Apple and Ashwaubenon watersheds in modeling that was conducted as part of the Duck, Apple, Ashwaubenon Priority Watershed Project (Roy Burton, Outagamie County LCD Director, personal comm. 1997).

At any given time, not all farm fields are in the same year or phase of the rotation. To represent average conditions, 1/6 of the dairy farm fields were assumed to be in each of the six years of the rotation. Therefore, it was necessary to simulate the different phases of each crop rotation by representing each phase as a separate HRU. The same procedure was followed for the cash crop rotation resulting in a total of 27 agricultural HRU's, not counting the barnyard HRU. Alternatively, the results of six model simulations would have to be averaged to represent average conditions.

The Level 3 classification of the 1992 WISCLAND classified land cover image was used as the primary basis for classifying agricultural land cover into dairy and cash crop categories. In agricultural areas depicted in the WISCLAND image, corn, forage and other row crops were classified for the strata on the west side of the Fox River; whereas, only row crop and forage classifications were assigned to the east side of the river. Both images were classified using images from May 5 and July 24, 1992. The WISCLAND image of the subbasin was actually based on two separately classified scenes (east and west side of river), which decreases the reliability of the data, particularly at the detailed Level 3 classification. Therefore, the subwatershed crop percentages derived from the WISCLAND data were not directly used as inputs. Instead, the proportion of dairy and cash cropping in each of the subwatersheds was derived by generalizing the subwatershed-specific data into two agricultural categories within the subbasin: dairy and cash crop. This task was accomplished by using the WISCLAND-derived proportion of forage crops within each subwatershed as a surrogate for the proportion of dairy crops within a subwatershed. It was assumed that a dairy rotation consisted of two years of corn followed by four years of alfalfa (forage), so the remaining proportion of land area was assigned to a cash crop rotation consisting of two years of corn followed by one year of soybean.

An analysis of agricultural land cover was performed on a county-wide basis whereby the percentage of crops determined with WISCLAND was compared to that published in the Wisconsin Agricultural Statistics. For the same time period (1992), the latter showed a greater proportion of forage crops for Brown and Calumet Counties, if it is assumed that a substantial portion of the oats crop acreage was planted as a nurse crop for the first year alfalfa/forage crop, in which case, this acreage should actually be considered forage.[4] Therefore, the WISCLAND-derived forage percentage was multiplied by a factor of 1.2 to correct for an assumed under-reporting of forage due to the nurse crop potentially being classified as a row crop. This factor was based on the ratio of forage acreage to forage minus oats acreage for Brown, Calumet, Outagamie and Winnebago Counties in 1992 (Wisconsin Agricultural Statistics 1994). Again, the assumption is that much of the reported oats acreage is actually used as a nurse crop for forage, even though it is also harvested as a grain. This assumption would not necessarily be true in previous decades.

For each subwatershed, the adjusted WISCLAND-derived forage proportions were then utilized to divide the remaining agricultural acreage between the dairy rotation and cash crop rotation by assuming that the typical dairy rotation consisted of 2 years of corn and 4 years of forage. A review of the resulting subwatershed-specific proportions of cash crop rotation, dairy rotation, forage and corn showed that the values were reasonable; typically, a greater proportion of row crops (e.g., corn and soybean) were grown in subwatersheds near urban areas compared to more rural subwatersheds. The subwatershed-specific proportions of cash crop and dairy rotations were utilized as SWAT inputs for the 1992 Baseline Scenario; that is, these proportions determined the fractional inputs of the agricultural HRU's.

To determine the 2000 Scenario proportions of cash crop and dairy rotations, the WISCLAND-derived values, which were based on 1992 land cover, were adjusted to account for a 23% weighted average proportional decrease of forage acreage between 1992 and 2000. The adjustment factor of 0.77 (100% - 23%) was determined by weighting the county-specific forage reduction which occurred between 1992 and 2000 (Wisconsin Agricultural Statistics, 1993 & 2001) by each counties relative proportion of agricultural land which was in the subbasin in 2000 (Outagamie 43%, Brown 41.5%, Calumet 10%, Winnebago 5.5%).

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[4] In the WISCLAND land cover image, oats planted as a nurse crop for alfalfa/forage crop could also be classified as a row crop instead of forage because classification as forage may depend in part on the difference between the May 5 and July 24 scenes. That is, the soil may have appeared bare in the first image where the oats crop was just planted or soon to be planted, thereby suggesting that the eventual crop was not a perennial forage crop. While oats may have formed seed heads by July 24, it is uncertain whether the oats would be considered a row crop or forage crop in the classified image. It is also possible that in the southernmost counties, the nurse crop would be more likely to be reported as forage, and possibly winter wheat or other grains because they may have produced a good cover over the soil by May 5, 1992. It all depends on which factors were emphasized when classifying the different agricultural classes: spectral signature or scene differences. Note that forage crop acreage in Brown County peaks at an average of 83,000 acres/year between 1990 and 1995, and drops thereafter.

As shown in Figure 2-4 in the following section, a substantial increase in soybean acreage occurred between 1990 and 2000. Many dairy farmers have added soybean to their crop rotation and cash crop farmers have increased the relative proportion of soybean in their crop rotation.

Therefore, both the dairy and cash crop agricultural HRU's were altered to reflect increased soybean acreage and decreased alfalfa acreage under the 2000 scenario by altering the crop rotations as follows:

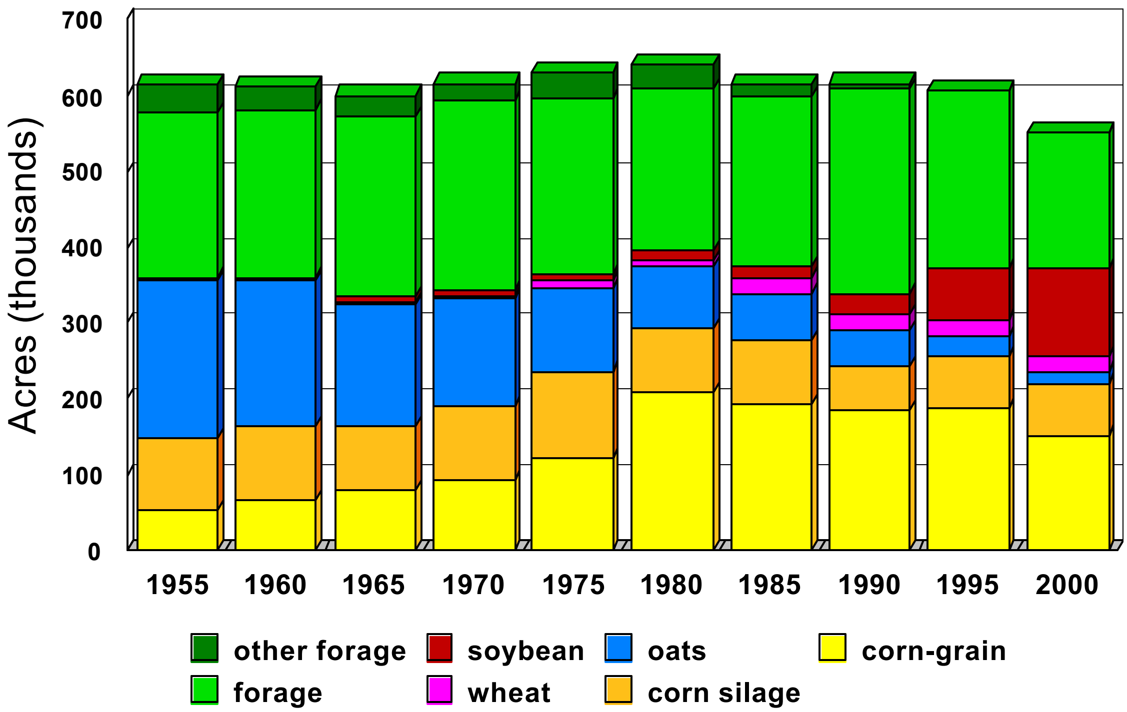
1) Dairy-based (6 year rotation: corn grain, soybean, corn-silage, alfalfa, alfalfa, alfalfa)
2) Cash crop (2 year rotation: corn grain, soybean).

With these changes in the 2000 data set, the proportions of alfalfa, corn and other row crops (primarily soybean and wheat) that were input to the SWAT model generally reflected the crop proportions listed in the Wisconsin Agricultural Statistics (2001) for the four counties in the subbasin. Greater emphasis was placed on reflecting the crop proportions in Brown County because this county is more nearly contained within the subbasin than the other three counties. However, compared to county crop statistics, soybean acreage was overstated (approx. 2-3%) with the simulated crop rotations, while corn was somewhat understated (approx. 2-3%). This discrepancy was considered acceptable because soybean under the dairy rotation has a low amount of protective residue, as does corn silage, so from the residue and associated erosion standpoint, these two crops are fairly similar as long as the discrepancy is not too large. An alternative set of crop rotations provided a closer fit with observed proportions, but it utilized a three year cash crop rotation (corn, soybean, corn) that would've required treating one of the two corn crops as if it were part of the dairy rotation. Therefore, this option posed more problems than the selected solution.

Crop Trends: Trends for major crops that are grown in Brown, Calumet, Outagamie and Winnebago counties are displayed in Figure 2-4 for a 1955 to 2001 period (Wisconsin Dept. of Agriculture 1956-2001). Each year shown in Figure 2-4 represents the average of two consecutive years (e.g., 1955-56, 1960-61....2000-2001). Tracking crop trends can be difficult because the published crop acreage may not be accurate at times. For example, in 1955-56, corn grain represented 37% of the total corn crop that was harvested in Brown, Calumet, Outagamie, and Winnebago Counties. By 1994-95, the reported percentage of corn harvested as grain had risen to 79%. However, local agricultural experts estimate that approximately equal amounts of corn-grain and corn-silage are currently harvested, with a future trend towards increased harvest of corn silage (Jim Rait, NRCS-FSA Brown County Executive Director; Doug Sutter, NRCS Brown County Ag. Agent; Kevin Erb, UW-Extension Ag. Agent; Roy Burton, Outagamie County LCD Director - personal communications, 1997).[5] Knowing the actual split between corn grain and corn silage is critical. Erosion from expected increases in the acreage harvested as corn silage could offset the effect of increased implementation of conservation tillage, because harvesting corn as silage leaves little protective residue on the ground compared to that left after corn is harvested as grain. According to UW-Extension Agricultural Agent Kevin Erb (personal comm. 1998), future gains made from increased adoption of conservation tillage may be offset by farmers harvesting a greater percentage of their corn crop as silage. Therefore, one of the scenarios simulated in this project investigated the potential impact of increasing the amount of row crops grown in place of forage crops.

[5] It is possible that the amount of corn harvested as grain may be exaggerated by farmers because monies paid out for crop insurance will be higher for corn-grain, than for corn-silage, if a hail storm or other natural disaster occurs.

Figure 2-4. Crop Trends in the Lower Fox River Subbasin Counties: 1955-2000. Total Acreage from Brown, Calumet, Outagamie and Winnebago Counties.



<u>Tillage Practices and Crop Residue Levels - Transect Survey</u>: Under each of the two potential farming practices, three tillage practices were simulated: a) conventional tillage with fall moldboard plow as the primary tillage implement for corn and fall chisel plow for soybean; b) mulch till, with chisel plow tillage in fall for corn and field cultivator in spring for soybean; and c) no-till. Field cultivator and disk tillage operations were also included prior to planting under the conventional and mulch till classes that were simulated.

The following primary tillage practices were assumed for each of the three simulated tillage classes:

Table 2-5. Primary tillage practices utilized in SWAT simulations.

tillage	corn or end of alfalfa rotation	soybean
conventional practice (CT)	fall moldboard plow	fall chisel plow
mulch till (MT)	fall chisel plow	spring field cultivator, or disk
no-till (NT)	none	none

Conservation Technology Information Center (CTIC) Conservation Tillage Reports from the four counties within the subbasin were analyzed to determine the primary tillage practice inputs to SWAT. These "Transect Survey" reports were based on statistical sampling procedures of farm fields to estimate residue levels present shortly after spring planting, as well as other information. Data was supplied by Len Olson of the Wisconsin Department of Agriculture, Trade and Consumer Protection, and it was analyzed with the Transect 2.13 software program produced by Purdue Research Foundation, Purdue University. Crop residue levels and tillage practices were summarized on a watershed basis with this program. Some of the watersheds may have contained too few points to be statistically reliable; however, most of the data seemed to be similar for adjacent watersheds. Where too few points were available, residue values were assigned on the basis of the average value from nearby watersheds.

To increase the number of available data points, data gathered in 1999 and 2000 were combined and utilized for the 2000 Scenario.[6] The earliest Transect Survey data available was from 1996, so these data were extrapolated to 1992 Baseline conditions by assuming that the increased rate of reduced tillage practices observed between 1996 and 2000 also occurred between 1992 and 1996. The level of residue present in alfalfa or forage fields, as indicated in the Transect survey data, was not used because there was limited data on this crop. Most of the time, no residue level was indicated even when the previous crop was alfalfa.

Four residue categories were initially assigned based on the percent residue present and the level of no-till or ridge-till practiced: conventional tillage (CT: 0-15%); limited mulch tillage (MT15: 15-30%); standard mulch tillage (MT30: >30%); and no-till or ridge-till (NT). Where no-till or ridge-till were present, the amount of acres which qualified as mulch-till were reduced accordingly to prevent double-accounting. The tillage/residue data were further combined into three categories for input to SWAT by reapportioning the MT15 category equally into the CT and MT30 categories. In this way, only three types of tillage practices needed to be simulated, while some credit was still given to tillage practices that left 15-30% residue, but did not exceed the 30% residue level required to be classified as mulch till.

Distinguishing between the average residue levels present in cash crop and dairy rotations was believed to be important, so additional processing of the data was performed to see if differences between these rotations could be discerned with regards to conservation tillage. To accomplish this task, Transect Survey data from 1999 and 2000 were combined from all the watersheds and summarized into two crop categories: (1) dairy rotation, represented by data from corn, small grain and other crop categories; and (2) cash crop rotation, represented by soybean. Based on this comparison, levels of MT30 and NT in cash crop operations were 1.33 and 2.25 times greater, respectively, compared to the dairy rotation in the entire subbasin. These ratios may have been larger if it was possible to separate corn grown under a cash crop rotation from corn grown under a dairy rotation. The number of soybean fields within most of the watersheds was too low to provide a reliable estimate of cash crop residue levels on a watershed basis. Therefore, the proportion of conservation tillage for the dairy rotation (represented by corn, small grains and other crops) within each

[6] Transect survey data gathered by the Brown County LCD in 1999 were not utilized in this project because it was not consistent with data from other sources and adjacent areas. For example, the percentage of present-crop corn fields in Brown County with greater than 30% residue (mulch-till) was 3%, 2%, and 4% in 1996, 2000 and 2002, respectively, based on data gathered by the NRCS. In contrast, data gathered by the Brown County LCD in 1999 indicated a level of 65% for the same category. Similar results were reported for the East River Watershed (LF01). It is important to note that approximately 50% of corn grown on dairy farms in this predominantly dairy region is silage, which typically has very little residue after it is harvested. Essentially, the same number of fields were checked in all of these surveys.

watershed was multiplied by subbasin-wide ratios of 1.33 (MT30) and 2.25 (NT) to estimate the proportion of conservation tillage under the cash crop rotation within that watershed.

Insufficient data was available in 1996 to perform this computation, so the ratios determined for the 2000 scenario were applied instead. The levels of conservation tillage for dairy operations under the 1992 baseline and 2000 scenario's are summarized in Tables 2-6 and 2-7, respectively. For cash crop operations, proportions of land under MT30 and NT residue levels are assumed to simply be 1.33 and 2.25 times greater, respectively, than the figures for the dairy operations. Alternatively, the same level of conservation tillage could have been applied to both dairy and cash crop rotations, but the method described here is believed to better reflect actual conditions. Using the procedures outlined above, watershed-specific levels of conservation tillage were modeled within the subbasin; furthermore, the levels of conservation tillage for cash crop rotations were assumed to be 1.33 and 2.25 times greater than under a dairy rotation, for MT30 and NT residue classes, respectively.

Table 2-6. Simulated level of conservation tillage for dairy operations: 1992 Baseline.

	Watershed	CT	MT	NT
LF01	East River	94.5%	5.5%	0.0%
LF02	Apple and Ashwaubenon Creeks	96.4%	3.6%	0.0%
LF03	Plum Creek	95.3%	4.7%	0.0%
LF04	Fox River/Appleton	95.3%	4.7%	0.0%
LF05	Duck Creek	78.8%	21.2%	0.0%
LF06	Lake Winnebago/North and West	93.3%	6.7%	0.0%
LFM	L. Fox Main Channel uses East River	94.5%	5.5%	0.0%

Table 2-7. Simulated level of conservation tillage for dairy operations: 2000 Baseline.

	Watershed	CT	MT	NT
LF01	East River	87.2%	11.3%	1.5%
LF02	Apple and Ashwaubenon Creeks	58.3%	41.7%	0.0%
LF03	Plum Creek	82.7%	17.3%	0.0%
LF04	Fox River/Appleton	65.9%	34.1%	0.0%
LF05	Duck Creek	77.5%	21.8%	0.7%
LF06	Lake Winnebago/North and West	77.5%	17.5%	5.0%
LFM	L. Fox Main Channel uses East River	87.2%	11.3%	1.5%

Crop Planting Dates: Statewide average crop planting and harvesting dates were obtained in digital form directly from the Wisconsin Statistical Reporting Service, Madison, Wisconsin. The average statewide dates were adapted to Northeastern Wisconsin by adding three days to the specified date. This was necessary because average dates were not available by county, even though crop yields are published by county (Kevin Erb, UW-Extension agent, personal comm. 1997; Steve Wilson, Wisconsin Agricultural Statistician, personal comm. 1997). Within the SWAT management files, crops were "planted" approximately 10 days later than the estimated planting date because SWAT assumes that the plant starts growing immediately instead of accounting for the time it takes for the seed to germinate and produce above-ground biomass.

Crop Harvest: The harvest index override (HIOVR) option was used for the alfalfa crop along with a HIOVR value of 0.89. This step was necessary because otherwise the alfalfa yields were too low early in the season because the harvested fraction is determined in SWAT as a function of the fraction of the total growing season at the time of harvest. In addition, the model code (dormant.f) was changed to reduce the fraction of biomass transferred to the residue fraction when a perennial crop (e.g., alfalfa) goes dormant at the end of the growing season. This fraction was reduced from 0.95 residue (0.05 remaining living biomass), to 0.8 residue (0.2 living biomass), which are the same fractions that are used within SWAT to assign the proportion of above ground biomass (0.8) and root biomass (0.2). Two harvests of alfalfa were simulated as occurring during the first year of the alfalfa rotation, whereas three harvests per year occurred during the remaining three years of the rotation.

During calibration of crop yields, a discrepancy was detected whereby total alfalfa biomass seemed to be overstated compared to the reported yields. Further investigation of both the standard output and subbasin model output files revealed that the model was summing the biomass present at each harvest date (3/year for alfalfa), rather than summing the increases in biomass between the harvest dates. That is, the model was double accounting with regards to the total annual biomass reported in the standard output file, so this problem only affected the reported biomass, rather than the biomass figures used to compute residue levels.

Nutrients and Nutrient Management: The following assumptions concerning commercial fertilizer and manure applications were utilized as model inputs in the 1992 Baseline and 2000 scenarios. The dairy rotation model inputs are summarized below, and the basis for these assumptions directly follows.

Dairy rotation - tillage options: moldboard plow, chisel or mulch till, no-till
1992 Baseline condition Scenario (1 year corn grain, 2nd year corn silage, 4 years alfalfa):

1 corn grain - 230 lbs/acre of 9-23-30 at planting; 9 t/acre manure in spring, 16 t/acre in fall

1 corn silage - 230 lbs/acre (258 kg/ha) of 9-23-30 at planting; 9 t/acre manure in spring

4 alfalfa - 2nd & 3rd year 190 lbs/acre (170 kg/ha) of 0-11-45 each year; apply 16 t/acre manure in fall of fourth year

2000 Current condition Scenario (1 yr corn grain, 2nd yr soybean, 3rd yr corn silage, 3 years alfalfa):

1 corn grain - 150 lbs/acre of 9-23-30 at planting; 9 t/acre manure in spring, 5.3 t/acre in fall

1 soybean - 150 lbs/acre (168 kg/ha) of 9-23-30 at planting; 10.7 t/acre manure in fall

1 corn silage - 150 lbs/acre (9-23-30 at planting); apply 9 t/acre manure in spring

3 alfalfa - 2nd year 19 lbs/acre (17 kg/ha) of 0-11-45; apply 16 t/acre manure in fall of fourth year (it was assumed that only 10% of farmers apply 180 lbs/acre of 0-11-45; hence, the rate of 19 lbs/acre of commercial fertilizer applied)

Commercial fertilizer application rates for dairy farmers under 1992 baseline conditions were based on the amount of phosphorus applied by 11 farmers in the East River Water Quality Demonstration Project to their corn grain and corn silage crops (Tables 7a and 8a, McIntosh 1994). The calculated overall average corn acreage application rate of 256 lbs/acre of 9-23-30 was reduced to 230 lbs/acre to partially account for a single large farmer that was applying fertilizer at twice the average rate. Commercial fertilizer rates were reduced to 150 lbs/acre of 9-23-30 for the 2000 Scenario. These modeled rates are consistent with estimates of 225-250 lbs/acre of 9-23-30 for the 1992 Scenario and 150 lbs/acre of 9-23-30 for the 2000 Scenario (average of areas with manure applied and not applied), provided by Kevin Erb of the UW-Extension Nutrient and Pest Management Program (personal communication, 2003). From the McIntosh (1994) data set, it was determined that an average of roughly 189 to 199 lbs/acre of 0-11-45 fertilizer was applied twice to the alfalfa crop during the typical 4 year alfalfa crop portion of the dairy rotation (or less if applied three times). On this basis, 190 lbs/acre of 0-11-45 fertilizer was applied twice to the alfalfa crop for the 1992 Baseline Scenario. According to Kevin Erb (personal communication, 2003), only about 10% of dairy farmers continued this practice of applying relatively high rates of commercial fertilizer to the alfalfa crop by 2000, so the modeled rate was reduced to 19 lbs/acre for the 2000 Scenario.

For the 1992 Baseline conditions, the total phosphorus concentration in fresh manure was assumed to be 0.11% on a wet basis. This value was based on the average nutrient content of solid dairy manure (5 lbs P_2O_5/ton wet manure) shown in Table 2 of the Wisconsin Conservation Planning Technical Note WI-1, companion document to NRCS FOTG Standard 590, Nutrient Management (August 5, 2002 release). These values are also the defaults for the Wisconsin P-Index model. For the 2000 Baseline Scenario, the total phosphorus concentration in fresh manure was assumed to be 0.1375% on a wet weight basis, which is a 25% increase over the 1992 level. This value was based on the 1998-2000 average statewide nutrient content of solid dairy manure samples that were submitted to the University of Wisconsin Soil Testing Lab (6.3 lbs P_2O_5/ton wet manure; n=799; data courtesy of John Peters, director of UW Soil and Forage Analysis Lab). As will be discussed later in this section, there is good reason to believe that phosphorus levels in dairy manure increased between 1992 and 2000, and may have peaked around 2000.

In a farm level phosphorus mass-balance study conducted by McIntosh (1994), the total phosphorus concentration in fresh manure was assumed to be 0.084% on a wet basis (14% dry matter, and 0.6% total phosphorus in dry manure , which is essentially the same as the SWAT default level of 0.08%. However, actual solid manure analysis of the farms in this same study found averages of 16.5% dry matter, 0.73% total phosphorus in dry manure, and 0.12% total phosphorus in wet manure. The latter value is close to the 0.11% value utilized in the 1992 Baseline Scenario. It was determined through back-calculations that the average total amount of dairy manure generated by 11 farmers in the East River Water Quality Demonstration Project ranged from approximately 41 to 45.6 tons/acre over a typical 6 year crop rotation (Tables 21a,b and 23a, McIntosh 1994). These amounts are consistent with expected application rates for that time period (Kevin Erb 2003).

In 1992 there were approximately 36,200 cows (Wisc. Agricultural Statistics), 176,100 acres of crop land, and 156,800 acres of harvested crop land in Brown County (U.S. Census of Agriculture[7]). Reported crop land acres include some lands that do not have harvested crops (idle, fallow, failed crops, pasture, conservation). In addition, 95,000 cattle and calves were present in 2000. If it is assumed that each cow produces 18.9 to 21.0 tons of solid manure per year (McIntosh 1994); and manure from young stock contributes an additional 25% of the manure from the cow herd only, then 855,000 to 950,000 tons of solid manure were produced (wet basis). If it is further assumed that 80% of the total acreage is under dairy production and receives manure applications, then on average, 6.1 to 6.8 tons (6.8 to 7.6 tons on harvested crop land) of manure were generated annually in Brown County per acre of crop land under dairy production in 1992. These numbers translate to a total application rate of 36.4 to 40.5 tons of manure/acre over a typical 6 year dairy crop rotation in 1992 (40.9 to 45.5 ton/acre on harvested crop land), which is close to the rate

[7] Acreage data from U.S. Census of Agriculture were not always consistent for Brown, Calumet, Outagamie and Winnebago counties. For example, 2002 data on farms included data from 1997 (Ag. Census Table 8), which for the most part had significantly higher acreage reported for most categories than were found with another data query for 1997 which also included the years 1992 and 1987 (Ag. Census Table 6).

determined for the study farms in the East River Water Quality Demonstration Project (McIntosh 1994). Therefore, the dairy management SWAT input files were created to reflect a total dairy manure application of 38 tons/acre over the entire 6 year crop rotation for dairy farms under the 1992 Baseline Scenario.

In contrast, there were approximately 41,500 cows in the most recent years (2000-2003, Wisc. Agricultural Statistics), and 164,600 acres of crop land in Brown County in 2000 (U.S. Census of Agriculture, extrapolated). Cow numbers rose sharply between 1999 and 2001, so the 2000 to 2003 period was believed to be more appropriate for the 2000 Scenario than just considering the number of cows in 2000. In addition, 92,000 cattle and calves were present in 2000. Using the same assumptions as in 1992, 980,000 to 1,090,000 tons of solid manure were produced, and on average, 7.7 to 8.5 tons of manure were generated annually in Brown County per acre of land under dairy production in the most recent years. These rates translate to a total application rate of 46.0 to 51.0 tons of manure/acre over a typical 6 year dairy crop rotation for this same period. Therefore, manure application rates were proportionally increased to 50 t/acre over a typical 6 year dairy crop rotation for the 2000 Scenario due to the decreased land area and increased cow numbers.

<u>Manure incorporation (depth of application)</u>: In 1992, roughly 70% of manure was on a daily haul basis (20% of the 70% was spread on sacrifice fields or hay fields), and 80-90% of the solid manure was not worked in within the three days (Kevin Erb of the UW-Extension Nutrient and Pest Management Program, personal communication, 2003). The remaining 30% or less was either liquid or semi-solid manure that was either spread, or injected and/or incorporated within three days (50-70%). On this basis, the following calculations were made to provide an overall estimate of the fraction of manure that was incorporated in 1992.

70% (daily haul) * 15% (worked in) = 10.5% mixed, 59.5% surface applied
30% (liquid/semi-solid) * 60% (injected or incorporated soon after) = 18% mixed deep, 12% surface applied
<u>TOTAL: 28.5% mixed deep, 71.5% surface applied in 1992</u>

Mike Mrychinski of the Brown County LCD estimated that in 1992 roughly 75% was not incorporated; whereas, about 50% of manure was incorporated in 2000 (personal communication 2003). Therefore, it was assumed that under the 1992 Baseline Scenario, 71.5% of the manure was surface applied; that is, not incorporated within three days of application. This value was changed to 50% for the 2000 Scenario. Within the SWAT model, surface applied fertilizer/manure is assumed to be mixed into the surface soil layer (10 mm, or 0.4 inches thick in SWAT), and the remainder mixed into the next layer, which was set to a thickness of 203 mm (8 inches) for this project.

<u>Cash crop rotation (corn & soybean)</u> - tillage options: moldboard plow, chisel mulch till, no-till

The following information summarizes the crop rotations and fertilizer applications for cash crop farms modeled under the 1992 Baseline and 2000 Baseline scenarios. The basis for these model inputs directly follows the summary.

1992 Baseline condition Scenario (2 years corn, 1 year soybean):

2 years corn: 125 lbs/acre ammonia prior to planting; 270 lbs/acre of 9-23-30 at planting

1 year soybean (soybean serves as legume crop or fragile crop in the cash crop rotation)[8]: 196 lbs/acre 9-23-30 at planting (note that this amount of nitrogen was not necessary for soybean but was kept to ensure the growth of modeled crops was correct

2000 Current condition Scenario (1 year corn, 1 year soybean):

1 year corn: 125 lbs/acre anhydrous ammonia prior to planting; 240 lbs/acre of 9-23-30 at planting (269 kg/ha)

1 year soybean: 150 lbs/acre of 9-23-30 at planting(168/258 kg/ha)

In Erb's (2000; Table 5) mass balance study within the Apple and Ashwaubenon watersheds, he found that cash grain farms had an average annual net loading of 3 kg/ha, or 2.7 lbs/acre of phosphorus during the 1996-98 study period. Commercial fertilizer was the only primary input of phosphorus, so the total average application of fertilizer must have been the net loading plus the amount needed by the crops. Assuming annual net loading of 2.7 lbs/acre phosphorus, crop phosphorus harvest removal rates of 24 lbs/acre from corn (130 to 150 bu/acre, UW-Ext. 1997) and 15.3 lbs/acre from soybean (35 to 45 bu/acre, UW-Ext. 1997), and a 50% corn and 50% soybean rotation, the average annual application rate would be about 22.4 kg/ha of phosphorus or about 225 lbs of 9-23-30 per year. This rate translates to 270 lbs and 180 lbs of 9-23-30 per year for corn and soybean, respectively (alternatively 250 and 200 lbs/year). With a two year corn and one year soybean rotation, the average application rate would be 240 lbs of 9-23-30 per year. Nutrient management practices increased by the year 2000, so fertilizer application rates should be lower than the 1996-98 rates. Conversely, fertilizer application rates may have been somewhat higher under the 1992 Baseline conditions, which is reflected in the assumed model inputs summarized above. Under the 2000 Scenario, it was therefore assumed that on average, commercial phosphorus fertilizer was applied at crop harvest removal rates for cash-crop operations, which is a reasonable assumption according to Jim Hunt of NRCS (personal communication 2004), and it is 2.7 lbs/acre/year lower than Erb (2000) found in 1996-98.

Under a nutrient management alternative scenario whereby soil test phosphorus and total phosphorus in the soil were assumed to no longer increase, no commercial starter fertilizer was added to soybean and a minimal rate of commercial starter fertilizer was added to corn at planting to boost corn yields under certain conditions.

Milk Production and Number of Dairy Cows: Milk production has been steadily increasing for several decades within the subbasin, as can be seen in Figure 2-5, which shows annual milk production and annual average number of milk cows in Brown County. Increases in annual milk production are likely to be associated with somewhat larger animals, and increased manure production per milking cow. In Brown County, milk production increased 52% from 532 million lbs in 1992 to 809 million lbs in 2003, even though the number of cows increased only 14.6% during this same period. There was a sharp increase in the annual numbers of milk cows in both 2000 and 2001. If these trends continue, the impact on dairy farmers in the subbasin may be substantial given that the amount of land currently available for manure application is expected to decrease as urbanization continues.

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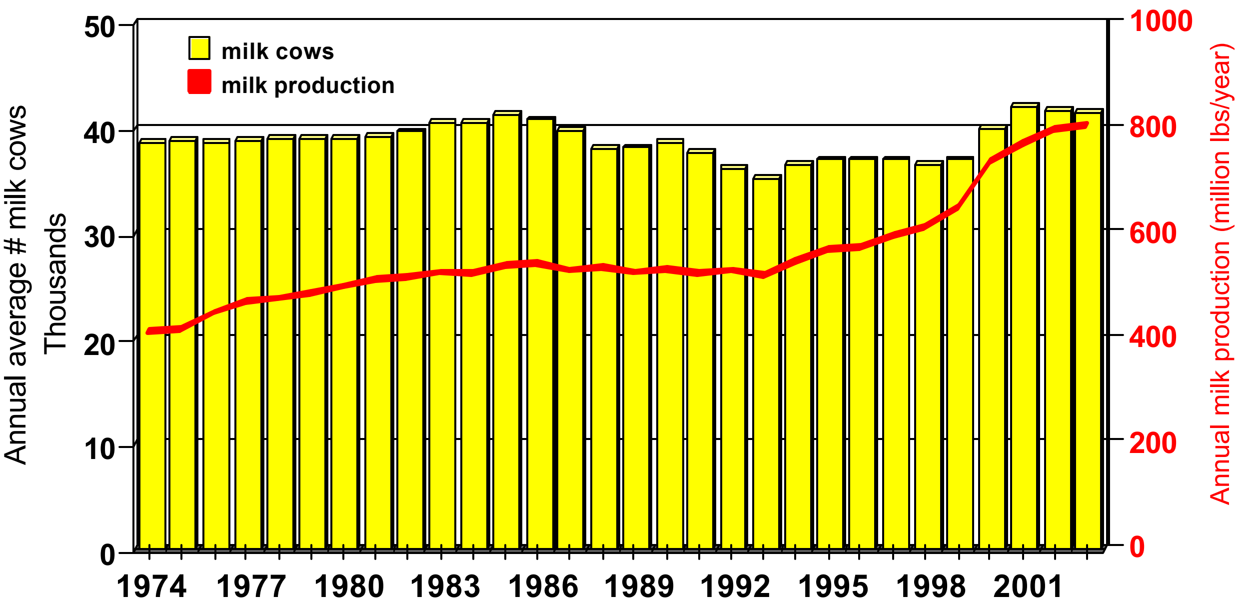
[8] Although some winter wheat and other crops are also grown, it is not possible to model all these crops in a single representative rotation.

Figure 2-5. Annual Milk Production and Number of Milk Cows in Brown County, Wisconsin (1974-2003)



Dietary Phosphorus Management Scenario (25% reduction in dietary phosphorus): Dietary phosphorus in dairy livestock frequently exceeds the required amount. Recent research has shown that existing levels of phosphorus in the diet of dairy cows can be substantially reduced without adverse health effects to cows. In a Wisconsin study, Ebeling et al. (2002) found that reducing phosphorus in the diet of dairy cows by avoiding excess supplements in the dairy ration greatly reduced the measured level of phosphorus in the corresponding manure, and the subsequent load of dissolved phosphorus in runoff from fields where the manure was applied.

The precise average dietary phosphorus level of dairy cattle in the subbasin is not known for either the Baseline 1992 or the 2000 scenarios. However, in a nutrient mass balance study conducted by Erb (2000) within the Apple and Ashwaubenon watersheds, most of the dairy farms aimed for 0.52% to 0.54% phosphorus in the milking cow ration. Examples of a 25% reduction could be: 0.55% to 0.41%, or 0.51% to 0.38%, or 0.41% to 0.34%. These examples of dietary phosphorus reductions still provide a margin of safety, for Wu et al. (2001) found that most studies showed that problems in dairy cattle don't appear until dietary phosphorus falls below 0.3%. Also, the U.S. National Research Council (NRC 2001) recommends dietary phosphorus levels of between 0.32% (55 lbs/day of milk) to 0.38% (120 lbs/day of milk) for dairy cows.

According to Kevin Erb (personal communication 2004), dietary P levels for dairy cows were 0.42% in 1992, 0.52% in 1997-98, and 0.42-0.46% in 2003 (on a per cow basis). As of 2003, small farms were generally around 0.52%, while larger farms were below 0.42%. On a per acre basis, the average in 2003 was around 0.46%. For the dietary phosphorus management scenario, a drop of 25% from the 0.52% level observed in 1997-98, to 0.39% seemed achievable. Therefore, a 25% reduction in the level of phosphorus in the dairy cow feed was selected for this scenario.

Reducing the phosphorus levels in the ration fed to dairy cows by a set percentage can result in an even greater reduction of phosphorus in manure produced. For example, in dairy cattle diets with phosphorus concentrations of 0.32% (no supplemental P) and 0.48% (P added), manures with average dry-weight phosphorus concentrations of 0.48 and 1.28%, respectively were produced (Ebeling et al. 2002). From this example, the concentration of phosphorus in manure could theoretically be reduced by about 62% with a 33% reduction of phosphorus fed to the dairy herd. However, the manure that was applied to the field plots in this study actually contained total phosphorus concentrations of 0.49% for the low phosphorus diet, and 0.89% for the high phosphorus diet. This difference translates to a 45% decrease in manure phosphorus with the same 33% reduction in dietary phosphorus. Morse et al. (1992) found that a 40% decrease in dietary phosphorus lowered manure total phosphorus by 23%, while Metcalf et al. (1996) reported a 30% decrease in manure phosphorus with a reduced phosphorus diet. Powell et al. (2001) estimated that dietary phosphorus levels of 0.35%, 0.38%, 0.48% and 0.55% fed to lactating cows, corresponded to average manure phosphorus amounts of 18.9, 21.4, 29.4, and 35.3 kg/cow/year, respectively, on a dry weight basis. In most studies, it appears that monosodium phosphate was added to raise the phosphorus levels fed to cows in the different treatments. However, organic feed inputs are frequently added to raise protein levels in the feed ration, which often raises the phosphorus level above the required rates. But excess organic phosphorus fed to cows may not translate to excess levels of total phosphorus in manure, at least not to the same degree as inorganic inputs. Therefore, the relationship between total phosphorus in the feed ration and corresponding manure content, as reported by Powell et al. (Figure 1, 2002), was utilized in this project because the relationship was more conservative than measured by Ebeling et al. (2002). On this basis, the effect of reducing the dietary phosphorus levels in dairy cattle by 25% (e.g., 0.51% to 0.38%) was simulated by reducing the fraction of total phosphorus in manure produced by lactating cows by 33%. This reduction roughly translates to a 25% decrease after non-lactating cows and other dairy livestock are accounted for. Therefore, for the “2000 Dietary P Reduction Scenario” the 25% decrease in dietary phosphorus fed to lactating cows was assumed to correspond to a reduction in manure phosphorus from all dairy livestock of 25%. The precise form of phosphorus that would be reduced under the scenario is unknown, so both mineral and organic phosphorus levels in applied manure were reduced by 25%, to a wet weight concentration of 0.103%.

RIPARIAN BUFFER STRIPS IN RURAL AREAS

Vegetative buffer strips (VBS), also known as vegetative filter strips, or riparian buffers strips, or filter strips can reduce sediment and nutrient loads to waterways. A VBS is defined as "a strip or area of herbaceous vegetation situated between crop land, grazing land, or disturbed land (including forest land) and environmentally sensitive areas" (Natural Resource Conservation Service, 1999). Although the primary goal of installing a VBS may be to reduce sediment and nutrient loadings to waterways, additional potential benefits include the ability to moderate water temperature, maintain and improve wildlife distribution and diversity, and to reduce human impact in urban environments.

A measure of a VBS's impact and effectiveness in reducing sediment and nutrient delivery to waterways is its trapping efficiency. Trapping efficiency measures the percentage of a given constituent load (e.g., sediment, phosphorous) which the VBS prevents from reaching the adjoining waterway. For example, if a VBS receives a sediment load of 100 kg from adjacent agricultural land and retains 70 kg its trapping efficiency would be 70%.

Default Method for Simulating Reductions due to VBSs in Rural Areas: Except as described below, the trapping efficiency and overall impact from VBS's along streams and road ditches in rural areas was simulated using essentially the same method as applied in the Restoration and Compensation Determination Plan for the Lower Fox River/Green Bay Natural Resource Damage Assessment, or NRDA-RCDP (Stratus 2000; Baumgart 2000b). The effect of VBS's along streams in urban areas was not simulated explicitly; rather, the urban calibration process implicitly incorporated potential effects of buffer strips which would be expected in a typical urban environment.

a) Reductions from agricultural areas due to VBS's were simulated internally within the SWAT model by modifying the code. It was therefore possible to vary the VBS trapping efficiency according to the average soil hydrologic group within each subwatershed, rather than use a single fixed value.

b) The width of installed VBS's was assumed to average 10 meters per side. Trapping efficiencies were assumed to be the same as the 15 meter VBS's outlined in the NRDA-RCDP Report, but the load reduction related to displaced agricultural land was decreased proportionately to reflect the smaller VBS width.

c) Simulated phosphorus and suspended sediment yields within the VBS 90 meter "effectiveness" zone are assumed to be 1.2 times greater than the average yield within the subwatershed (instead of assuming the yield throughout a subwatershed is spatially homogenous). With this load factor of 1.2, the VBS-affected load to non-affected load ratio is 1.5 when the two areas are equal. The effect of this change is to increase the simulated impact and overall effectiveness of a VBS compared to the method applied by Stratus (2000).

d) Potential VBS impact zones along streams were identified as areas within 100 meters of a stream under agricultural landuse (90 meter effectiveness zone, plus 10 buffer width). Only streams identified in the 1:24k hydrology layer provided by the WDNR were included in this portion of the analysis. A GIS method similar to that outlined in the NRDA-RCDP to estimate the potential agricultural land that could be impacted by adding VBS's was utilized for this purpose. However, areas that were already buffered were not included, because they had already been accounted for as natural buffers.

The potential to add VBS's along road ditches was also incorporated into the model. However, the impact area affected by buffered roads was assumed to be 2/3 that of a comparable stream. The factor of 2/3 was based on the assumption that road ditches are not as efficient as streams with regards to intersecting surface water runoff. The increased loading factor of 1.2 was applied to roads, as it was for streams. Including road ditches had the effect of roughly doubling the potential VBS-impacted area. Other surface drainage channels (i.e., the extended drainage network) were not included directly in this VBS analysis. However, the total length of such channels that could be buffered (excluding grass waterways) is substantial. Visual inspection of 1992 digital ortho-photographic images indicate there are many drainage channels beyond that indicated solely by the 1:24k hydrology layer. Based on this first-cut analysis, the extended drainage network could be roughly the same length as the road ditch or 1:24k stream networks. If a high resolution DEM was available, a better approximation of the extent of this network could be estimated.

e) The effects from existing "natural" forested and wetland VBS's were accounted for in this application of the model. The linear extent of existing forested and wetland VBS's was determined through GIS analysis by intersecting the stream hydrology with the combined landuse/land cover layer (Brown County landuse, ECWRPC landuse, WISCLAND land cover). A 90 m VBS-impact zone was then created around existing forest and wetland buffer areas, which were identified as those areas which intersected the stream network. Landuse/cover within the VBS-impacted areas was then tabulated for each subwatershed, which was used to estimate the relative proportion of agricultural land in the VBS-impacted areas compared to agricultural land, and the corresponding reduction potential. The increased loading factor of 1.2 was not used with "natural" existing VBS's, because the impact zone of the GIS-identified "natural" buffers extends 90 meters further than it should at the upstream and downstream ends of the VBS. In addition, visual inspection of the GIS-estimated buffer impact zone indicated that this zone was apparently too large in some areas. To correct for this problem, the estimated zone was decreased by 20% whenever the area was over two times greater than the average of two other existing buffer impact zone estimation methods, which were not utilized directly for this project.

f) In addition to existing "natural" VBS's along streams, existing forested and wetland VBS's along road ditches were accounted for in the model by increasing the impact area affected by buffered streams by an additional amount equivalent to 2/3 of the proportional increase in the combined road ditch and stream lengths that were intersected by wetland and forested landuse/cover (additional area of 11%). The factor of 2/3 was based on the assumption that road ditches were not as efficient as streams with regards to intersecting surface water runoff. Including road ditches had the effect of increasing the natural VBS impacted area by about 11% * 2/3, or 7.7%. More importantly, the agricultural area that could be impacted by adding a VBS increased substantially compared to only adding buffers along streams.

g) VBS's that were installed by the Brown County LCD were accounted for in the model under the 2000 Baseline Scenario and alternative scenarios using the aforementioned methods. The installed VBS GIS layer supplied by the Brown County LCD was utilized for this purpose. Other installed VBS's including those installed by the NRCS, and the Outagamie, Calumet and Winnebago County LCD's were not accounted for in the model. As of the year 2000, the extent of these VBS's within the subbasin did not appear to be significant. It was assumed that the installation of VBS's under the 1992 Baseline Scenario was negligible.

h) Trapping efficiencies for soluble and insoluble phosphorus were not assumed to be the same. Instead, insoluble phosphorus was set to that of TSS, and soluble phosphorus was set to 0.24 for C soils and 0.27 for B soils. The model was calibrated such that these latter values produced the same total phosphorus trapping efficiency as the NRDA-RCDP project (Stratus 2000; Baumgart 2000b) under the assumption that 30% of phosphorus delivered to rural streams is soluble. These assumptions may overstate the VBS reduction potential for soluble phosphorus, at least relative to that assumed for insoluble phosphorus and TSS. However, with these trapping efficiencies the model was less sensitive to potential errors in the underlying assumption of 30% soluble phosphorus.

With the methodology employed in this VBS-modeling effort, the combined suspended sediment reduction within the assumed VBS-affected area that is associated with both the up slope VBS impact zone (90 m assumed effective range) and the VBS width of 10 m on each side of a stream, is about 54%, 49% and 45% for hydrologic group A, B and C soils respectively. The combined phosphorus reduction is about 49%, 45% and 40% for hydrologic group A, B and C soils respectively. The assumed increased loading factor of 1.2 effectively increases these simulated impacts compared to reductions calculated without the factor. Slightly higher reductions would be associated with a cash crop rotation compared to a dairy rotation as the proportion of smaller particle sizes associated with runoff with the former rotation would lower, but this difference was ignored. In addition, the VBS trapping efficiencies utilized here are based on an analysis of studies which utilized relatively high tillage practices (Baumgart 2000b), so the reduction potential from VBS's may be lower for reduced tillage practices.

With the approach utilized in this VBS-modeling effort, the combined potential VBS-impacted area is about 56.3% for the calibration subwatershed LF01-15 when all 1:24,000 streams and roads are buffered (including existing natural buffers). If the assumed 1.2 increased loading factor is added to account for greater transport capacity attributed to these more efficient flow paths, then these areas would account for approximately 68% of the total load contribution from this example subwatershed, which is fairly representative of average conditions in the subbasin. If the buffer impact zone is increased another 90 m, then the combined potential VBS-impacted area would be about 81% of the total area within the subwatershed, plus adding the 1.2 increased loading factor would essentially mean that the entire load is within this 180 m impact zone. But these assumptions do not include the extended stream network which is not delineated in the 1:24,000 hydrology network. Therefore, the high proportion of the subwatershed that is calculated to be impacted with the 180 m zone assumption suggests that the average effective impact zone is likely to be less than 180 m.

Alternative VBS Simulation Methods: Two alternative methods of estimating the impact of installing VBS's were applied, and the results compared to the default method described above. In the first alternative, the same methods mentioned above were utilized except the assumed trapping efficiency was raised from 45% to 65% for hydrologic group B soils, and a proportional increase for the other soils. Ideally, the model would first be recalibrated after this change was made, and then another Baseline 2000 scenario would be developed to compare to the alternative scenarios. However, it was much easier to instead determine the relative change between Baseline 2000 results with this alternative VBS method, and an alternative scenario where 100% of the streams were buffered (without recalibrating the model). The relative difference between the loads determined with this method and the default method under the Baseline 2000 Scenario was then used to estimate the absolute loads for the alternative scenario where 100% of the streams were buffered.

In the second alternative VBS simulation method, a different approach was utilized to estimate the impact of installing VBS's. Initially, two local assessments which investigated the potential impact of installing vegetative buffer strips (VBS) were considered for application in this project. The Brown County Land Conservation Department (1999) estimated a 70-80% sediment trapping efficiency based on the removal rates reported for buffer strips in a literature review conducted by Desbonnet et al. (1994), as well as many other supporting documents (Brown County LCD 1995). An alternative approach proposed by Baumgart (2000b) was applied to the Green Bay Drainage Basin through the National Resource Damage Assessment Resource Compensation and Determination Plan for the Lower Fox River and Green Bay (Stratus 2000). As previously mentioned, a modified version of the latter method was employed throughout this project. However, the second alternative VBS simulation method was developed to estimate the impact of installing VBS's using a method somewhat similar to that proposed by the Brown County LCD (1999). On this basis, trapping efficiencies of 80% for suspended sediment and sediment-attached phosphorus were assumed to occur when VBS's were either installed, or already present as wooded, wetland or grassed areas within agricultural landscapes (for hydrologic group B soils). A soluble phosphorus trapping efficiency of 48% was assumed to occur when a VBS was present. Trapping efficiencies of 75% for suspended sediment and 45% for phosphorus were assumed for hydrologic group C soils. These impacts were simulated by modifying the model code.

Overall reduction potential within each subwatershed was directly related to the proportion of the total stream network with installed or natural VBS's, as indicated in the 1:24,000 stream coverage. Road ditches and the extended stream network were not considered in this analysis. The linear extent of existing forested and wetland VBS's was determined through GIS analysis by intersecting the stream hydrology with the combined landuse/land cover layer (Brown County landuse, ECWRPC landuse, WISCLAND land cover). The first option under this alternative VBS simulation method included the impact of these existing forested and wetland VBS's, whereas the second option did not. The effect of VBS's along streams in urban areas was not simulated explicitly; rather, the urban calibration process implicitly incorporated potential effects of buffer strips which would be expected in a typical urban environment, which was also done with the default VBS simulation method,

MODELING LOADS FROM URBAN AND URBANIZING AREAS

Urban Areas: The buildup and washoff option was selected as the method to simulate urban loads from impervious surfaces in SWAT. The buildup/washoff method incorporated in SWAT is similar to that usesd in the Storm Water Management Model (SWMM, Huber and Dickinson 1988). Because measured loads from different urban sources were not available within the project area, all urban areas were lumped into one class and simulated as medium density residential areas. For the pervious portion of the urban HRU, phosphorus and sediment loadings were simulated by assuming that these areas are in grass, and an appropriate SWAT management routine was developed to simulate the runoff and loadings from these areas.

The urban component of the SWAT model was initially calibrated for suspended sediment and total phosphorus by adjusting the urban management file and associated files to obtain a representative suspended sediment concentration of about 88 mg/L and a total phosphorus concentration of 0.18 mg/L during a 1977-2000 climatic period (representative concentration = total long-term load/total long-term water volume). These calibration concentrations and corresponding yields were based on a review of the following urban runoff data which is summarized in Table 2-8: (1) four urban Milwaukee, Wisconsin streams with a median and mean of 107 mg/L and 152 mg/L TSS, respectively, and median and mean of 0.18 mg/L and 0.21 mg/L total phosphorus, respectively (Bannerman et al. 1996); (2) 8 Wisconsin and two Upper Michigan storm sewer sites with a median and mean of 120 mg/L and 237 mg/L TSS, respectively ,and median and mean of 0.29 mg/L and 0.45 mg/L total phosphorus, respectively (Bannerman et al. 1996); (3) 8 Lake Superior Basin cities storm sewer sites with a median and mean of 284 mg/L and 433 mg/L TSS, respectively, and median and mean of 0.44 mg/L and 0.47 mg/L total phosphorus, respectively (Steuer et al. 1996); (4) Marquette, Michigan storm sewer site with a geometric means of 159 mg/L TSS and 0.29 mg/L total phosphorus (Steuer et al. 1997); (5) seven stormwater sites in Madison, Wisconsin with a median and mean of 93 mg/L and 106 mg/L TSS, respectively, and a median and mean of 0.32 and 0.38 mg/L total phosphorus, respectively (Waschbusch 1995); (6) stormwater from 25 runoff events within residential basins in Madison, Wisconsin had a median and mean of 136 mg/L and 171 mg/L TSS, respectively, and a median and mean of 0.40 and 0.59 mg/L total phosphorus, respectively (Waschbusch et al. 1999); (7) stormwater from 15 runoff events that entered a treatment chamber installed below the pavement surface at a municipal maintenance garage and parking facility in Milwaukee, Wisconsin contained median event mean concentrations of 232 mg/L TSS, and 0.262 mg/L total phosphorus (Corsi et al. 1999); (8) 43 samples of stormwater that entered an urban stormwater treatment unit which collected runoff from a 4.3 acre municipal maintenance yard in Madison, Wisconsin contained median and mean concentrations of 251 mg/L and 345 mg/L TSS, respectively (Waschbusch 1999); and (9) during 64 runoff events, stormwater entering a wet detention pond in Madison, Wisconsin from a 0.96 km^2 residential area had median and average event mean concentrations of 144 mg/L and 239 mg/L TSS, respectively, and median and average event mean concentrations of 0.45 mg/L and 0.57 mg/L total phosphorus, respectively (House et al. 1993).

Table 2-8. Summary of phosphorus and suspended sediment/TSS concentrations measured in urban streams and storm sewers within Wisconsin and neighboring states.

	Sediment/TSS (mg/L)				Total Phosphorus (mg/L)			
	storm sewer		urban stream		storm sewer		urban stream	
Reference	median	mean	median	mean	median	mean	median	mean
Bannerman, et al. 1996	120	237	107	152	0.29	0.45	0.18	0.21
Steuer et al. 1996	284	433			0.44	0.47		
Waschbusch, R.J. 1995	93	106			0.32	0.38		
Waschbusch, R.J. 1999	251	345						
House et al. 1993	144	239			0.45	0.57		
Steuer et al. 1999	159				0.29			
Corsi et al. 1999	232				0.26			
Waschbusch et al. 1999	136	171			0.45	0.59		
median	**152**	**238**	**107**	**152**	**0.32**	**0.47**	**0.18**	**0.21**
average	**177**	**255**	**107**	**152**	**0.36**	**0.49**	**0.18**	**0.21**

After calibration, the 1977-2000 average annual simulated urban suspended sediment yield was 0.32 t/ha from a typical urban subwatershed in LF01, and the total phosphorus yield was 0.64 kg/ha. These yields compare to observed median unit-area loads of 0.46 t/ha TSS (15 watersheds ranged from 0.06 to 1.58 t/ha) and 0.56 kg/ha total phosphorus (4 watersheds ranged from 0.23 to 2.12 kg/ha) from urban watersheds in Southeastern, Wisconsin till plains ecoregion (Corsi et al. 1997). In addition, simulated yields were similar to the range of urban yields that were generated with the SLAMM model by RUST (1999, now Earth Tech) for the City of Green Bay Stormwater Management Plan. Urban yields of sediment that were generated for Green Bay with the SLAMM model ranged from 0.10 t/ha in residential areas to 0.86 t/ha in commercial areas. Urban yields of phosphorus that were generated for Green Bay with the SLAMM model ranged from 0.40 kg/ha in residential areas to 2.9 kg/ha in industrial areas. For all SLAMM modeled urban landuses (excluding open space and wetland), the combined sediment yield was 0.33 t/ha and the combined phosphorus yield was 1.05 kg/ha.

Suspended solids in storm sewers are expected to have a higher proportion of large particles than in urban streams as larger particles are more likely to settle out in streams, or before reaching the stream, than in storm sewers. Larger particles are not associated with reduced water clarity in Green Bay, nor are they expected to be a major component of runoff from rural areas. Therefore, greater emphasis was given to water quality data collected from streams than data from storm sewers when selecting the calibration concentrations.

Urbanizing Areas: Urbanization is a transitional change from rural to urban land use. Urbanization and associated land use changes for the simulation period are by nature continuous. Therefore, the problem of constructing a model framework for simulating the spatial and time dependant nature of this change throughout the simulation period did not render a simple or obvious solution. Perhaps understandably, the current version of SWAT does not directly model continuous changes in land use over time in a single model simulation.

An HRU could have been used to represent urbanizing areas, but potential technical problems precluded using this method for this project. Instead, rural to urban transition was simulated by adding a separately calculated urbanized load to the SWAT-simulated loads and assuming that urban areas increased at a steady rate of 2.6% per year, or an amount that was directly estimated (e.g., 1992 vs. 2000). Loads were computed for each subwatershed by assuming that the annualized change in urban area from 1992 to 2000 remained the same for the simulated periods. To derive the load associated with urbanizing, the average annual increase in urban area within each subwatershed was then multiplied by an assumed yield (mass/area), which was based on two separate Wisconsin studies that are described below. In an alternative scenario (Chapter 11, Scenario 11), it is assumed that the urban area has doubled. Therefore, the annual amount of land undergoing urbanization is also assumed to double in this scenario, compared to the 1992 and 2000 Baseline scenarios.

In a study conducted from spring 1977 to summer 1978, Madison et al. (1979) found that the mean and median TSS concentrations from rapidly urbanizing watersheds in Germantown, Wisconsin were approximately 6,900 and 5,100 mg/L, respectively during monitored runoff events. The mean and median total phosphorus concentrations were about 4.5 and 2.9 mg/L, respectively. The mean sediment yield was roughly 16.3 t/ha and the mean phosphorus yield was approximately 10.7 kg/ha. While erosion controls were implemented in the non-control watershed, they were judged to be ineffective due to drought conditions. Owens et al. (2000) studied soil erosion from two small construction sites in Dane County, Wisconsin. Both sites were less than 5 acres. During the active construction phase the flow-weighted average concentration of suspended sediment from a commercial site was 12,700 mg/L (n=8), and 2,600 mg/L (n=3) from a residential site. However, they noted that few of the storms produced runoff at the residential site because most of the construction took place in winter; whereas, construction at the commercial site primarily took place during the summer months. Furthermore, they suggested that there was evidence which indicated that the suspended sediment concentrations could have been as high at the residential site as they were at the commercial site if construction had instead taken place during the summer months. Sediment yields of 6,750 lbs/acre (7.6 t/ha) were measured during the summer construction season at the commercial construction site, whereas 1,650 lbs/acre (1.8 t/ha) was measured during the winter construction season at the residential construction site. Some level of erosion controls are currently being implemented in the subbasin, and both total precipitation and rainfall intensity are generally expected to be lower in Northeastern Wisconsin. An HRU was created to roughly simulate fallow conditions similar to what might occur during urbanization. Subsequent SWAT simulations with this HRU produced a sediment yield of 2.8 t/ha and a phosphorus yield of 4.0 kg/ha under the 1977-2000 climatic period.

Based on the Wisconsin construction site runoff data and associated caveats, and SWAT simulations under fallow conditions, a sediment yield of 4 t/ha and a phosphorus yield of 4.5 kg/ha were utilized to calculate subwatershed-specific loads due to urbanization (as routed to the watershed outlet). These representative sediment and phosphorus yields are roughly 8 times and 3 to 4 times higher, respectively, than yields generated with the SWAT model for comparable agricultural areas.

OTHER MODEL INPUTS

NRCS (SCS) Curve Numbers: Default condition II curve numbers furnished with SWAT were initially utilized for all crops and landuses. These curve numbers are the same as those recommended by NRCS in their National Engineering Handbook (USDA 1972). The curve numbers were altered throughout the rotation periods according to changes in crop type, crop growth and tillage practice. Curve number is also affected by the soil hydrologic group, which ranges from A (most permeable) to D (least permeable). During the calibration process, all curve numbers were adjusted by a single factor to get the best fit between observed and simulated water yields, although a slightly lower value was utilized in the Duck Creek watershed. As previously stated, area-weighted average values for the NRCS hydrologic group were generated for each subwatershed. This was accomplished by representing soil hydro groups A to D as real numbers from 1.0 to 4.0, and utilizing the GIS to compute area-weighted average values for each subwatershed based on the area of each soil within the subwatershed. These average values, were then converted to NRCS curve numbers based on the crop type, crop maturity, tillage practice and time of season.

Instead of having a separate management file for each hydro group classification, an equation was added to the model code which adjusts the curve numbers associated with tillage operations according to the soil hydrologic group. Thus, only a single management file is required for each management practice; otherwise, each management practice requires a separate management file for each soil hydro group category to be represented. In addition, rather than being limited to representing only a small number of soil hydro group categories, this method allows the use of the actual weighted-average soil hydro groups for each subwatershed.

Subwatershed Channel Width and Depth: The channel widths for main channels and routing reach channels, and the channel depths for routing channels, were estimated using a modified form of the following equations which were adopted from Theurer and Comer (1992).

$$\text{channel width} = 1.29 * DA^{0.6} \qquad \text{(Eq. 2)}$$
$$\text{channel depth} = 0.131 * DA^{0.4} \qquad \text{(Eq. 3)}$$

The modified equations used for SWAT model inputs were:

$$\text{channel width} = (1.29 * DA^{0.6})/1.8 \qquad \text{(Eq. 4)}$$
$$\text{channel depth} = 0.15 * DA^{0.5} * (0.001/\text{slope}_{\text{channel}})^{0.4} \qquad \text{(Eq. 5)}$$

where:

channel width and depth are in meters

DA = drainage area in km^2 (routing reach cumulative area)

Channel depths and widths estimated with the above equations were similar to measured values for the main stem of the East River (Quinlan 1989), and for the Upper East River (unpublished transect data collected by the USGS, Madison, Wisconsin in 1994). Subwatershed main channel width is not a critical parameter within the SWAT model. A review of the FORTRAN source code revealed that this parameter only affects transmission losses through the stream bed. Therefore, TSS is not affected by this parameter because only surface runoff affects TSS yields; whereas, water yield is only slightly affected by subwatershed channel width. Substitution of channel width values ranging from 0.1 m to 120 m confirmed that simulated TSS yields from a subwatershed were not affected by channel width values.

Manning's n: A Manning's n value of 0.065 was used for all main channels and routing reaches, except routing reaches near the outlet of large watersheds were assigned a value of 0.04. For overland n, a value of 0.1 was assumed for all subwatersheds. Changes in overland n values had virtually no effect on simulated runoff and TSS export, even over an extreme range of values.

Groundwater Inputs: Both the revaporation storage and revaporation coefficient parameters were set to zero for all simulations. Revaporation is water from the shallow aquifer or sub-surface water which is drawn to the surface and evaporates. Because revaporation does not contribute to surface water or recharge, it can be considered an overall loss that can be accounted for elsewhere in the model. The aforementioned problems with evapotranspiration implied that revaporation could be ignored, for it simply decreased the water available for surface water runoff or recharge.

The groundwater recession alpha factor (ALPHA_BF) and groundwater delay value (GW_DELAY) were adjusted so that the simulated hydrograph recession matched the observed recession. The alpha factor characterizes the groundwater recession and the rate at which groundwater flow is returned to the stream; whereas, the delay is the time it takes for water leaving the bottom of the root zone to reach the shallow aquifer where it becomes groundwater flow (Arnold et al. 1996). The alpha factor can be thought of as defining the slope of the recession curve. ALPHA_BF and GW_DELAY were set at 0.6 and 6.0 days, respectively. These values were chosen to fit the measured hydrograph, rather than reflect what the anticipated groundwater recession curve would actually look like.[9] The chosen set of values worked relatively well for the calibration subwatershed (Upper Bower Creek - 36 km^2), which is rather "flashy", but they did not work as well on the much larger Duck Creek watershed (276 km^2), at County Highway FF. This matter will be discussed in the following section. The selection of appropriate deep percolation coefficients is discussed later in the "Model Calibration - Hydrology" section.

The concentration of soluble phosphorus associated with groundwater was set at 0.06 mg/L for both urban and agricultural-impacted areas, 0.07 mg/L for wetlands, and 0.04 mg/L for forested areas. These levels essentially act as the lowest possible phosphorus concentration that can be simulated in a stream (assuming no water losses in the stream).

Biological Mixing Efficiency: According to the SWAT users manual: "Biological mixing is the redistribution of soil constituents as a result of the activity of biota in the soil (e.g. earthworms, etc.). Studies have shown that biological mixing can be significant in systems where the soil is only infrequently disturbed. In general, as a management system shifts from conventional tillage to conservation tillage to no-till there will be an increase in biological mixing." Therefore, biomixing was set at the default value of 0.2 for all areas with some exceptions. For no-till management files, the BIOMIX parameter was increased from 0.2 to 0.4 for the cash crop rotation and to 0.5 for the dairy rotation. BIOMIX was increased to 0.6 for the intensive rotational grazing rotation (IRG), which was modeled under an alternative scenario and is expected to have the greatest biological activity. These changes were made to simulate additional biological mixing that is expected to occur when tillage is reduced substantially. Jimmie Williams, one of the developers of the EPIC, APEX and SWRRB models confirmed that a BIOMIX value of 0.4 to 0.5 is acceptable for reduced-tillage management.

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[9] If the goal was to reflect groundwater inputs from recharge due to the upper aquifer only, then the ALPHA_BF should be set lower and the GW_DELAY should be longer. However, the objective was to instead match the overall hydrograph recession after a runoff event, recognizing that some processes, such as stream bank storage of groundwater, are not directly accounted for by SWAT (J.G. Arnold, personal comm. 1998).

Sediment and Phosphorus Routing Sub-models: Routing of sediment and phosphorus was done with the SWAT model except for the main channel of the Fox River (described later in this section). SWAT currently uses a routine to determine routing channel sediment deposition/resuspension that is now documented in the most recent user's manual (Neitsch et al. 2001b). This new routine is based on the concept of stream carrying capacity for sediment.[10] Except for Duck Creek (LF05), the following values which affect sediment deposition and resuspension in the routing reaches were used for all simulations: (1) SPCON = 800 mg/L TSS; (2) SPEXP = 1.4; and (3) PRF = 1.25. These values were within the acceptable range recommended by SWAT model developers (J.G. Arnold and J.R. Williams, personal comm. 1998). The calibrated version of the model was not sensitive to minor adjustments to these parameters since most of the sediment exported from the individual subwatersheds was routed to the watershed outlet by the model with the selected parameter values. During the calibration phase, it was determined that a SPCON value of 300 mg/L, and simulated sediment concentrations derived with this value, more closely matched observed concentrations at the USGS Duck Creek monitoring station (at CTH FF), compared to a SPCON of 800 mg/L. Lowering the SPCON value effectively reduced the capacity of the system to transport higher concentrations of sediment, as is expected where streams flow through wetland complexes.

An alternative procedure was considered whereby the process of trapping sediment in wetlands as water is routed through the system is directly simulated by the SWAT model by treating the wetland complexes along the tributary channels as reservoirs. However, this procedure is far more complex than altering the SPCON value because it involves providing model inputs for the storage volumes and associated trapping efficiencies of the wetland complexes in the watershed, and treating the wetlands as a reservoir. The simpler procedure of lowering SPCON for LF05 is believed to be more suitable given the uncertainty involved in estimating the required model inputs. In addition, although modeling of wetlands as ponds was utilized for subwatersheds LF01-5 and LF01-12 where tributaries pass through large wetland complexes, it was not implemented elsewhere because doing so might duplicate (double account) the modeling process that already accounted for the potential buffer impact of wetland areas along the tributary network. Long-term simulations indicated that the SWAT wetland routine removed about 10% of the phosphorus and 35% of the sediment in LF01-12, and 19% of the phosphorus and 47.5% of the sediment generated in LF01-5.

Phosphorus transport in stream reaches was modeled using the QUAL2E submodel within SWAT. The QUAL2E inputs were altered slightly for Duck Creek (LF05) because somewhat more phosphorus is expected to be trapped through the LF05 wetland complexes compared to the other watersheds (RS-5, P settling increased from 0.15 to 0.25, and BC4 organic P mineralization decreased from 0.02 to 0.01).

Model simulations indicated that the routing reach channels produced unreasonably high amounts of channel degradation even with very low USLE soil erodability (K) and USLE cover (C) factors. Therefore, the routing channel degradation component of the model was essentially "turned off" by assigning a value of zero to both the K and C factors within a channel routing reach. Hence, within routing reach channels, only deposition was tracked by the model. The lack of sufficient measured TSS loads from watershed-scale areas precluded attempts to fully calibrate this important aspect of the model. However, starting in October, 2003,

[10] Both model developers J.G. Arnold and J.R. Williams (personal comm. 1998) stated that the equations currently used in the SWAT model produced better results than the previously used method which relied on computing deposition based on fall velocity and Stoke's Law, and also applying Bagnold's (1977) stream power equation to compute degradation.

the Lower Fox River Watershed Monitoring Program (www.uwgb.edu/watershed) implemented a three year watershed monitoring project in the East River, Duck Creek, Apple Creek, Ashwaubenon Creek and Baird Creek watersheds which will provide valuable sediment and phosphorus loading data that can be used to validate the model.

As previously described, the SWAT model was used to route phosphorus and TSS non-point rural and urban loads to the Fox River. However, SWAT was not used to transport sediment and phosphorus loads from watershed outlets through the main channel of the Fox River to Green Bay. Instead, point source loads, urbanizing loads and SWAT-simulated non-point urban and rural source loads at watershed outlets were routed to the lower Green Bay using the approach applied by Baumgart (2000a). Based on a relationship between trapping efficiency and the reservoir capacity/average annual inflow ratio that was developed by Brune (1953) and extended by Dendy (1974), an estimated 5% of the Fox River suspended sediment was assumed trapped between the Lake Winnebago outlet and Little Lake Buttes des Morts (LLBDM), another 5% in the reach from the Little Rapids dam (10.6 km upstream from the DePere dam) to the DePere dam, while an additional 15% was assumed to be deposited between the Little Rapids dam and Fox River mouth. For phosphorus, 2.5%, 2.5% and 7.5% of the non-soluble fraction was assumed to be trapped between these two river reaches, respectively (½ the net sediment deposition rate). Soluble phosphorus was assumed to be conservative, so it was transported through the system with no losses (i.e., no net deposition).

For purposes of routing urbanizing loads from watershed outlets to Green Bay, it was assumed that 90% of phosphorus is not soluble and is treated as sediment, while the remaining 10% of the phosphorus is soluble and is transferred down the system with no net loss. This assumption is based in part on the results of a study involving monitored runoff from urbanizing sites in Wisconsin in which Madison et al. (1979) found that the proportion of dissolved phosphorus ranged from 0.5% (29 kg/ha) to 14% (2.8 kg/ha) of the total phosphorus load during six events. Moderate events which had yields that more closely resembled the assumed yield of 4.5 kg/ha for this project had higher proportions of dissolved phosphorus. The assumed proportion of dissolved phosphorus is also the same as that used in the Fox-Wolf Basin NEWWT analysis (White et al. 1995).

For routing purposes, it was assumed that 65% of the point source phosphorus load was soluble, and therefore routed directly through the system with no net loss; whereas, the remaining fraction was routed as sediment-attached phosphorus. Little difference in the total phosphorus load routed to Green Bay was observed when the soluble fraction ranged from 50% (-1.4%) to 75% (+ 1.0%), compared to the assumed value of 65%. Point source contributions of suspended sediment to the lower Fox River were routed in the same manner as non-point sources.

CHAPTER 3. BARNYARD CONTRIBUTIONS

SIMULATING BARNYARD CONTRIBUTIONS

To simulate daily runoff and phosphorus export from barnyards within the SWAT model, an HRU was established in each subwatershed to represent barnyard areas. Existing barnyard load estimates served as a basis for model calibration. The estimated barnyard phosphorus load from the 36 km^2 Upper Bower Creek calibration subwatershed LF0115 was 448 kg, or 988 lbs (Wierl et al. 1996), which was updated to 376 kg, or 828 lbs (Rappold et al. 1997), presumably due to best management practice (BMP) implementation and/or other changes. These phosphorus loads were estimated with the Wisconsin Barnyard Runoff Model (BARNY) under a simulated 10 year, 24 hour storm event (WDNR 1996). BARNY is a modified version of the USDA Agricultural Research Service Feedlot Runoff Model (Young et al. 1982). For LF0115, the event-based load was converted to an annual load by multiplying by a factor of 2.61, which was based on the event to annual load ratio of Bower Creek subwatersheds. BARNY-derived phosphorus loads were provided on an annual basis by the Brown County LCD and the Outagamie County LCD, so no conversion was necessary elsewhere in the East River and in the Duck, Apple, and Ashwaubenon watersheds.

For calibration purposes (1992 Baseline conditions), it was assumed that the 1977-96 long-term average annual phosphorus load from the Upper Bower Creek calibration subwatershed was 1,172 kg, or 2,580 lbs (988 lbs * 2.61 / 2.2 kg/lb). Simulated daily manure loads from the barnyard HRU were then adjusted until the long-term annual load was equal to this value. For other areas in the subbasin, subwatershed-specific manure loads were based on BARNY estimates reported in the East River Priority Watershed Project (WDNR 1993b) and the Duck, Apple, Ashwaubenon Priority Watershed Project (WDNR 1997).[11] Little data was available for subwatersheds outside of these two watersheds, so barnyard phosphorus loads from these subwatersheds were assumed to be proportional to the barnyard load from the LF0115 calibration subwatershed on an agricultural land areal basis.

Again, these estimates were used for the 1992 Baseline Scenario. Updated BARNY loads obtained from the Brown County and Outagamie County LCD's were utilized for the Baseline 2000 Scenario, with one modification. Stuntebeck and Bannerman (1998) measured an 85% and 87% reduction in the phosphorus loads contributed to Otter Creek and Halfway Prairie Creek, respectively, from two barnyards after BMPs were installed to control manure runoff. Based on this research, and the fact that some measurable losses will occur unless a barnyard is replaced with a confined operation, an 85% reduction in phosphorus load was assumed whenever barnyard BMP controls were implemented. Stuntebeck and Bannerman (1998) also noted that: (1) "...none of the 18 runoff events occurring with frozen ground were sampled during the post-BMP period at Otter Creek."; and (2) "The percentage reduction in loads for Otter Creek might have been lower if sampling had included all runoff periods occurring with frozen ground, when filter strips are not expected to work efficiently (Schellinger and Clausen, 1992)." On the other hand, potential reductions may be greater for some operations. Therefore, the presumed 85% reduction that was applied in this project may need to be

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[11] Alternatively, a single representative unit-area barnyard load could be assigned to the barnyard HRU in all the subwatersheds so that the total barnyard load within each subwatershed would be proportional to the amount of land operated as a dairy operation.   The representative barnyard load would be based on the typical barnyard load/dairy farm land area ratio that was estimated with the BARNY model in the East River and Duck, Apple, Ashwaubenon watersheds, but it would be applied uniformly throughout the entire subbasin.

revisited. A 100% reduction in phosphorus loads was assumed when either the barnyard was no longer in use, or it was replaced with a confined system.

POTENTIAL CONTRIBUTIONS FROM BARNYARDS

Investigations by USGS and WDNR indicate that barnyards may still contribute a significant portion of the phosphorus load in watersheds with relatively high numbers of barnyards (Stuntebeck 1995, Stuntebeck and Bannerman 1998, Wierl et al. 1998). Previous BARNY-estimated loads for the Otter Creek watershed in Wisconsin indicated only 71 lbs due to a 10 year, 24 hour storm event (Wierl et al. 1996). However, during a single event in this watershed, these researchers measured about the same amount of phosphorus from a single barnyard with 50 cows; furthermore, 5 out of 12 measured events from this barnyard exceeded 20 lbs of phosphorus during the April 1994 to October 1995 pre-BMP phase of the study (see Figure 2 in Stuntbeck and Bannerman 1998). According to Wierl et al. (1998), controlling phosphorus from barnyards appears to be as important as reducing phosphorus in crop land runoff in watersheds where the ratio of farm fields to barnyards is about 20:1 or less. The ratio for Bower Creek, in the East River Watershed, is 15:1 (Wierl et al. 1998).

Based on pre-BMP data from barnyard studies conducted by the USGS and WDNR (Stuntebeck 1995, Stuntebeck and Bannerman 1998), the total phosphorus load from 11 events in the 42 km^2 Halfway Prairie Creek watershed increased from 187 lbs to 294 lbs due to the contribution from a single barnyard (57% increase in total event load). Similarly, the total phosphorus load from 12 events in the 24 km^2 Otter Creek watershed increased from 360 lbs to 616 lbs due to the contribution from a single barnyard (71% increase in total event load). An upstream-downstream approach was utilized to measure the contribution from each of the barnyards. In both cases, the barnyards were located near the bottom of the watersheds, so a substantial portion of the phosphorus load from each watershed was attributable to runoff from a single barnyard during the measured events. Importantly, installation of barnyard BMP's reduced the load of total phosphorus by 85% at Otter Creek and 87% at Halfway Prairie Creek, based on the Hodges-Lehmann estimator (Stuntebeck and Bannerman 1998).

If historical barnyard loads were actually much higher than predicted, then improvements in barnyard operations within the subbasin may have produced greater phosphorus load reductions than anticipated; plus, the contributions from sources such as crop land may have been greater than previously estimated. Further investigation is recommended to ensure that historical and current estimated phosphorous loads associated with crop land and barnyard runoff are correct. Furthermore, a methodology to track trends in barnyard runoff is recommended in the watershed load allocation monitoring phase.

In 2003, Jeff Kreider of the WDNR was debugging the Barnyard Confined Animal Loading Model (BCALM), which will serve as a replacement for the BARNY model in Wisconsin. BCALM is a daily time step mechanistic model with significantly more sophistication than BARNY, and it is likely that predictions from BCALM will be substantially different than from the BARNY model (John Panuska, WDNR 2003, personal communication).

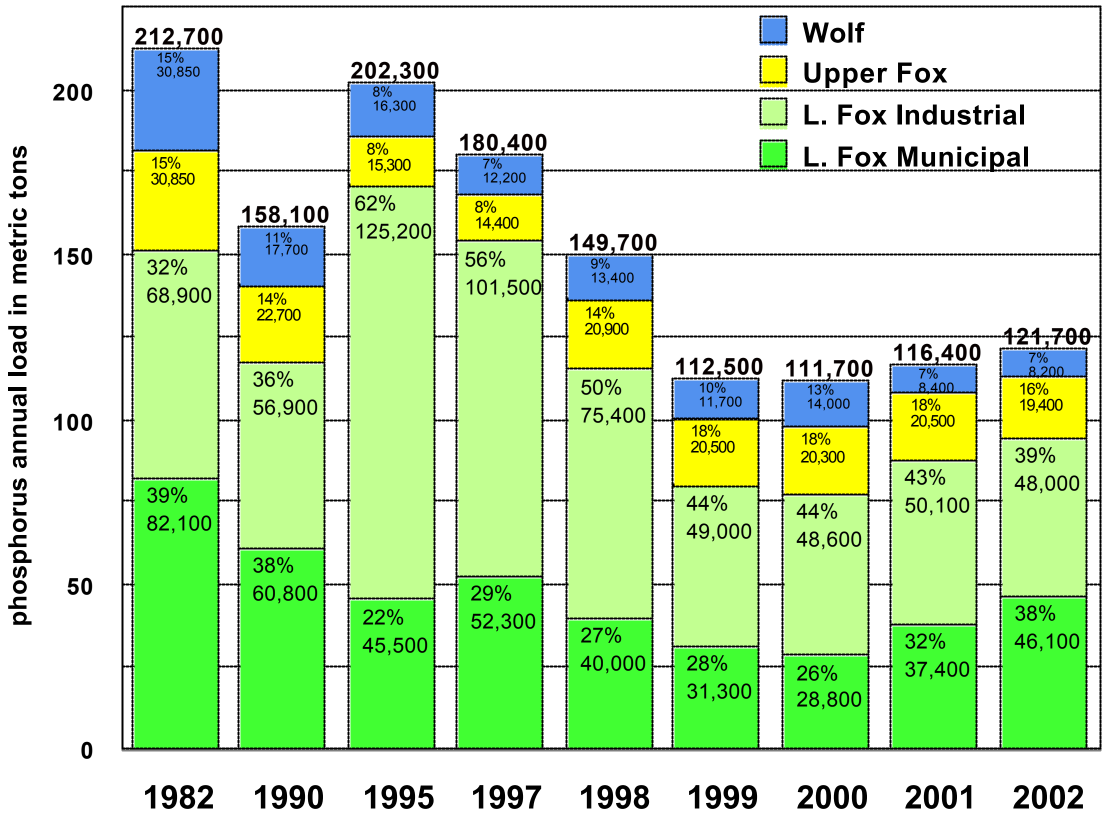
CHAPTER 4. POINT SOURCE CONTRIBUTIONS

Phosphorus: The WDNR compiled phosphorus concentrations and loads from point source facilities within the subbasin for the following years (1982, 1990, 1993-2003). For this project, this data was merged into a single database which will serve as a basis for tracking trends and allocating current and projected loads of phosphorus from point sources. Phosphorus loads from industrial and municipal point sources within the Lower Fox, Upper Fox and Wolf river subbasins are compared for selected years in Figure 4-1. The loads represented in this figure are not routed to the outlet of the Fox River. A marked decrease in loads from municipal sources has occurred in the last 18 years. During this same period, total phosphorus contributions from industrial sources have increased dramatically at times, such as in the mid-1990's, but recent loads were lower than in 1982. Point source discharges of phosphorus between the Lake Winnebago outlet and the Fox River mouth varied from about 117,700 kg in 1990 to 170,700 kg in 1995, and to a low of 77,400 kg in 2000 before increasing in more recent years (Figure 4-1).

Loads for the Baseline 1992 Scenario were assumed to be an average of available data from 1990 to 2000. Loads derived under this scenario were primarily used as a comparison to measured loads (i.e., model assessment). Loads for all Baseline 2000 Scenarios were assumed to be an average of data from 1999 to 2002. Where no current data existed, the most recent data was typically utilized. With these assumptions, the estimated phosphorus contribution from point sources in the Lower Fox River Subbasin was 134,000 kg/yr for the 1990 to 2000 period (Baseline 1992 Scenario) and 85,000 kg/yr for all Baseline 2000 Scenarios. These load estimates are not routed to Green Bay. The estimated phosphorus contribution routed to Green Bay from point sources in the Lower Fox River Subbasin was 131,000 kg/yr for the Baseline 1992 Scenario and 83,000 kg/yr for the Baseline 2000 Scenario.

Suspended Sediment: Total suspended solids loads from point sources were derived from a 1989 to 1995 data set summarized by the Wisconsin Dept. of Natural Resources (WDNR 2001). This data set was not updated with more recent information because the contribution of TSS from point sources is relatively small compared to other sources. On this basis, point sources along the lower Fox River contributed an average of 3,500 t/yr of TSS from 1989 to 1995, or 3,000 t/yr of TSS routed to Green Bay.

Figure 4-1. Fox-Wolf Basin Point Source Phosphorus Loads (1982-2002).
Average annual load from Fox River is 450,000 to 600,000 kg/year.



** 1982 combined load from from Wolf and Upper Fox basins divided equally as precise fractions not found
Annual loads obtained from the Wisc. Dept. of Natural Resources.

CHAPTER 5. MODIFICATIONS TO SWAT FORTRAN CODE

Modifications of the SWAT2000 program code (4/18/2001 version) were performed to create the version of the model that was applied in this project. Major modifications to SWAT and some problems with the SWAT code are described in this chapter. SWAT model developers and water quality modelers with the WDNR were informed of any possible problems discovered with the model code.

Sediment Equations: The "ysed.f" file was modified to permit the selection of different sediment yield equations, and to allow user inputs for sediment equation coefficients and exponents so that the model could be calibrated. This change simply reflects the same approach used in EPIC (Sharpley and Williams 1990), APEX (Williams et al. 1995) and the NRCS version of SWRRBWQ (Arnold et al. 1994).

Evapotranspiration Routine: Balancing the water budget to provide the expected long-term surface volume runoff of about 200 mm/year (Gebert et al. 1987) was not readily feasible without questionable modifications to key parameters. Using default values, water yields from 41 year simulations (1956-96) with the uncalibrated SWAT model were approximately 80 to 95 mm too low, depending on which potential evapotranspiration (PET) method was selected. With the Hargreaves and Samani (1985) PET option, simulated water yield for a 1990-1994 period was only 137 mm with default parameters, compared to an expected 200 to 220 mm. Increasing the NRCS curve number did not greatly affect the total water yield to the stream; instead, surface runoff increased at the expense of recharge to the stream. Changing key soil parameters such as available water capacity (AWC) and saturated conductivity within acceptable ranges also produced an insufficient effect on water yield. In addition, the soil parameters were obtained using a relatively robust approach, so it did not seem wise to greatly change them without strong evidence which would indicate that the values were wrong. It was only possible to obtain sufficient simulated water yields by altering both the available water capacity of the soil (AWC) and plant uptake compensation factor (EPCO) to near the minimum values of their respective acceptable ranges. However, reducing the EPCO to the lowest permitted value had an undesirable effect on simulated water yields after precipitation events following extended dry spells. This problem will be discussed in more detail later. Reducing the EPCO to nearly zero also caused crop biomass and yields to drop dramatically, for the ability of a crop to extract soil moisture from lower levels was sharply reduced as EPCO approached zero. In addition, the seasonal range in soil moisture dropped sharply; which is opposite of the desired change. Therefore, a different approach was evaluated and utilized in this project.

In the SWAT2000 version of the model, it is now possible to have the model read in daily PET values (Neitsch et al. 2000a, 2000b). After reviewing SWAT documentation, SWAT code and published articles concerning evapotranspiration equations, the simulated water budget was balanced by simply reducing the SWAT-computed potential evapotranspiration (PET) by an adjustment factor. This step was simpler than having a separate program compute daily PET. Instead, a variable called the evapotranspiration coefficient (ETCOEF) was added to each of the three PET equations used by SWAT (in etpot.f), which simply reduced the PET by a factor that was input by the user in the control-input/output file:. This is the same approach used by Baumgart (1998) with a previous version of SWAT (SWAT1996). However, the PET levels computed by SWAT2000 are lower and produce water yields that are closer to expected values than the SWAT1996 version, so the PET did not require as much adjustment. To calibrate the model, the ETCOEF factor was adjusted until simulated water yields to the stream were approximately the same as the observed stream volume. A value of 0.806 was chosen for the selected Hargreaves and Samani (1985) PET equation, and this value was used in all simulations.

Priestley-Taylor Potential Evapotranspiration Methods: When the Priestley-Taylor evapotranspiration option was tried with the model, simulated water yields were much higher and more similar to expected values, when compared to the other methods. Unfortunately, an apparent shortcoming was discovered whereby evapotranspiration during winter months was essentially zero. Therefore, this method was not utilized for this project. A further explanation of the problem is described below.

In the model code, and documented revisions made to SWAT2000 (from SWAT1999), the same net radiation equation that is used for Penman-Monteith (PM) is now used in the Priestley-Taylor (PT) method to estimate potential evapotranspiration. Prior documentation (Arnold, et al. 1996), and the model code from earlier SWAT versions, both indicate that solar radiation had been used to estimate net radiation for the Priestley-Taylor. However, when the revised net radiation equation is utilized with the PT method, problems can occur whenever net radiation is computed as less than zero (fairly frequent during snow cover conditions). For Green Bay, Wisconsin, the method now used for Priestley-Taylor results in essentially zero ET and PET during both January and February, and these values are nearly always zero in December. Removing snow cover by artificially setting temperature for snowfall to occur to minus 30 degrees Celsius produces a major change for the PT method, but a much less drastic change for the PM method. This problem doesn't occur in the PM method because the equation is additive; whereas for the PT method, net radiation is multiplied by other factors. While net daily radiation can be less than zero, it doesn't mean that no evapotranspiration occurs. A potential solution that seemed to work was to modify the model code so that net daily radiation was not allowed to go below zero.

Hargreaves-Samani Potential Evapotranspiration Method: Preliminary model runs indicated that with SWAT2000, simulated crop biomass was much lower for the Hargreaves and Samani method than with either the Penman-Monteith or Priestley-Taylor PET methods. Crop biomass and yields were found to be identical for the latter two methods after the code was temporarily modified to remove the effect of all stress factors (i.e., water, temperature, nutrients). Crop yields and biomass are important because they affect crop residue production, which indirectly affects comparisons between different tillage systems.

The problem was traced to a query in the model code whereby relative humidity was not computed when the Hargreaves and Samani PET method has been selected. Model documentation coincides with this finding. However, even though relative humidity is not needed for PET with HG, it affects the vapor pressure deficit (VPD) and therefore biomass production of the crop (in the grow.f file). To correct this problem, the clicon.f code file was changed so that the model generates relative humidity regardless of which PET method is selected, unless measured relative humidity values are read into the model. After this change, crop yields were identical with all three PET methods when all stress factors were ignored. Some earlier versions of SWAT only computed relative humidity for the Penman-Monteith method, but the differences in crop yields and biomass were not as large as the current version of SWAT.

Perennial Crops (alfalfa): A problem occurred whereby the alfalfa crop kept growing regardless of "kill" or "harvest and kill" commands in the management file. Thus, even though corn was planted, the model was actually using the plant characteristics associated with the perennial crop alfalfa. This problem was traced to a recent code change, so the code in the readmgt.f file was altered to reflect that in the SWAT1998 version (nkill = 0 instead of 1, in two lines). After this change, perennial crop growth was stopped by using the "kill" command after a separate "harvest" command instead of a combined "harvest and kill" command because the latter command did not stop growth. This temporary "fix" to the model code was not ideal, but worked well nonetheless.

Crop Harvest: The model code (dormant.f) was changed to reduce the fraction of biomass transferred to the residue fraction when a perennial crop (e.g., alfalfa) goes dormant at the end of the growing season. This fraction was reduced from 0.95 residue (0.5 remaining living biomass), to instead reflect the model's computed values of total biomass minus the root biomass (living biomass represented as the root biomass). Generally, SWAT assumes that 20% of the mature crop's total biomass is composed of root biomass, leaving the remaining 80% to be transferred to the residue fraction when the perennial crop goes dormant.

Minimum Base Temperature: At times, crop biomass and yield were lower than anticipated. Further investigation revealed that the corn crop did not grow during certain years. This problem only occurred when air temperature were low when the crop was planted. Apparently, the average air temperature was below the base temperature of the crop at these times. Normally, when the air temperature is below the base temperature for a crop, no growth will occur. However, the plants should still grow once temperatures warm up, but this did not occur with the model. To overcome this problem, the code was modified to have the crop grow slightly (1/100th of the potential growth), even when the temperature stress factor, or other plant stressors, would normally dictate zero growth.

HRU Versus Subwatershed Channel Lengths and Areas: In the unmodified version of SWAT2000, the subwatershed ("subbasin" in SWAT) channel length is multiplied by the HRU/subwatershed area ratio to obtain an area-weighted channel length that is supposed to be associated with that HRU. The HRU area is also used to compute time-of-concentration. For this project, the SWAT 2000 code was modified so that both the subwatershed channel length and subwatershed area are used to calculate the time of concentration for all HRU's within a subwatershed. The rationale for this modification is provided below.

Because an HRU area-weighted channel length is used in the unmodified version of SWAT, simulated subwatershed sediment yields are strongly affected by the number of HRUs assigned in the SWAT simulation, as well as the HRU size (total phosphorus yields are less affected). The reason for this is that the channel length greatly affects time-of-concentration, which affects peak flow, which then affects the simulated sediment yield. The HRU area is also used to compute time-of-concentration in the unmodified version of SWAT, so similar effects are observed.

However, there is no real physical basis for either the HRU channel length or area being used to generate peak flow. HRU's are abstract representations that can be discontiguous within a subwatershed, so scale isn't applicable to an HRU. The area of the HRU is relevant only as it is applied to the fractional area of the subwatershed within which it belongs. The Modified Universal Soil Loss Equation (MUSLE) operates at the subwatershed scale when sediment yield and load are computed. HRU's simply offer greater flexibility within that framework. Therefore, subwatershed channel length and subwatershed area ought to be applied when generating the proportion of sediment that is contributed from each HRU within a subwatershed. Consequently, the SWAT 2000 code was modified so that the subwatershed channel length and area are used instead of the HRU values to calculate the time of concentration for all HRU's within a subbasin.

In addition to the above changes, consistency also required that the model code be modified by substituting subwatershed area (sub_ha) for HRU area (hru_ha) in the unit-area form of MUSLE as follows:

$$sedyld = 1.586 * Qsurf^{0.56} * Qpeak^{0.56} * sub\_ha^{0.12} * CKSLP * CFRG \quad \text{(Eq. 6)}$$

where:

sedyld = metric tons of sediment per hectare from HRU, at subwatershed outlet (Mg/ha)

Qsurf = surface runoff volume in mm

Qpeak = peak flow rate in mm/hr (in ysed.f: peakr * 3.6/(sub_km))

Channel flow time-of-concentration uses subwatershed channel length & area, not HRU values.

sub_ha and hru_ha (subwatershed and HRU respective areas in hectares)

CKLSP = USLE multiplier factors ©, K, LS, P)

CFRG = coarse fragment fraction

To calculate the HRU sediment load at the subwatershed outlet, the above yield (Mg/ha) is simply multiplied by the HRU area (ha). In essence, "hru_area^1.12", as applied in MUSLE with the unmodified version of SWAT, is separated into subwatershed area^0.12 and HRU area^1.0 to reflect the premise that MUSLE is operating at the subwatershed scale, rather than the abstract HRU scale. Importantly, these modifications ensure that a user can select any number of HRU's, or vary the size of the HRU's, without concern as to how these factors will affect the simulated load at the subwatershed outlet. Without these modifications, increasing the proportion of mulch-till and reducing the proportion of conventional-till within a single subwatershed over time may produce odd results, as the respective HRU area-weighted channel lengths and HRU areas also change, thereby affecting sediment and phosphorus yields. However, if the peak flow and drainage area exponents in the unit-area form of MUSLE had been set to zero (defaults are 0.56 and 0.12, respectively), the changes recommended here would have no affect on simulated sediment yields because HRU "channel length" and HRU area would no longer have any direct influence on sediment yield.

To compensate for the above changes, the tran.f file was altered so that HRU area-weighted channel length was calculated and utilized only to compute the HRU transmission losses, as done in the unmodified version of SWAT. Without this change, transmission losses would be too high because subwatershed channel length would be used in the modified version of SWAT to calculate the transmission loss from each HRU within that subwatershed.

Changing the time-of-concentration equation by using subwatershed channel length instead of the HRU area-weighted value also affects the delay of stream flow and constituents such as sediment and phosphorus. It is expected that this change should result in an improvement as the subwatershed scale is a more appropriate operating scale compared to the HRU scale. However, if the above changes are made to the model code, SWAT model users should also increase the surface water lagging factor input, which relies on the time-of-concentration. Otherwise the delay of surface water and associated constituents will be too long if the lagging factor had been based on the unmodified version of SWAT with a data set with many HRU's. Flow volume and total constituent loads remain the same over a long period of time with the modified version of SWAT, but the simulated amount on a particular day is different due to the lagging factor change. With the modified version, the time-of-concentration and the lagging factor now coincide with the size of the subwatershed, and not the HRU. In the unmodified version of SWAT, time-of-concentration varied from about 0.68 hours for no-till HRU's to 1.4 hours for conventional till HRU's in the 33 HRU LF0115 subwatershed. These values could vary even more widely if this subwatershed had been divided into an HRU which consisted of all agriculture, and the remaining HRU's were for other landuses (probably from over 5 hours for the agricultural HRU to about 0.7 hours for the remaining HRU's). With the modified version of SWAT, the time-of-concentration of 5.9 hours was the same for all agricultural HRU's within the 36 km^2 LF0115 subwatershed, and this value is about what would be expected for a subwatershed of this size.

Simulations which divided a subwatershed into either 1 or many urban HRU's confirmed that with the modified version of SWAT2000 there was no difference in the sediment or phosphorus yields from a theoretical urban subwatershed (LF0115 as urban) with the modified version of SWAT2000. However, substantial differences were observed when the unmodified version of SWAT2000 was applied to the same data sets.

Transmission Loss: The location in the code where the transmission loss subroutine is called was moved to directly after the sediment subroutine is called. In the unmodified version, transmission losses were calculated before the sediment loads were calculated; thereby, affecting sediment yields which is not appropriate. While this change seems reasonable in the project area, it may cause problems in areas where sediment generated from upland areas frequently never reaches the subwatershed outlet because the transmission losses in the stream are so great that there is no flow at the outlet (personal communication, Jeff Arnold March 2003).

NRCS Curve Number: To reduce the number of management files required in the SWAT model, a simple equation was added to the readmgt.f file. The following equation adjusts curve numbers associated with tillage operations according to the soil hydrologic group.

$$CNa = CN + ((CN\_soil-78)/7 * (101 - CN)/3) \quad \text{(Eq. 7)}$$

Where:

CNa =	NRCS curve number associated with tillage practice and crop conditions, but adjusted for the actual soil hydrologic group
CN =	NRCS curve number associated with the tillage practice and crop conditions, assuming B hydro group soil
CN_soil =	NRCS curve number associated with actual soil, assuming standard crop of corn (67 for A, 78 for B, 85 for C soils); new variable added to readsol.f

With this change, only a single management file is required for each management practice; otherwise, each management practice requires a separate management file for each soil hydrologic group category that is represented. In addition, rather than being limited to representing only a small number of categories, this method allows the use of the actual weighted-average soil hydrologic group for each subwatershed. These averages were calculated by having hydrologic groups A to D represented as real numbers from 1.0 to 4.0, which were then converted to NRCS curve numbers associated with the assumed standard crop conditions (CN_soil).

Maximum Phosphorus Enrichment Ratio: The enrichment ratio for sediment-attached phosphorus and organic phosphorus was changed to allow an upper limit for the phosphorus enrichment ratio (ERORGP) instead of the default fixed value. This change allows the enrichment values to vary according to model computations, but only up to the maximum value set by ERORGP (default is 3.5).

Crop Residue Change: The model code was changed to separate the effect on the USLE C factor of above ground living biomass from the effect of any remaining ground residue. Otherwise, the above ground live biomass dwarfs any remaining ground residue that might remain with conservation tillage; in effect, this is the same as assuming that no-till corn is not much different than moldboard plow once the crop is well underway (on a C-factor basis). While a number of methods could be used as a remedy (e.g., crop canopy cover vs ground residue), the same methodology that is used in SWAT was utilized, but the effect of the two forms of erosion protection were separated in the USLE C factor calculation.

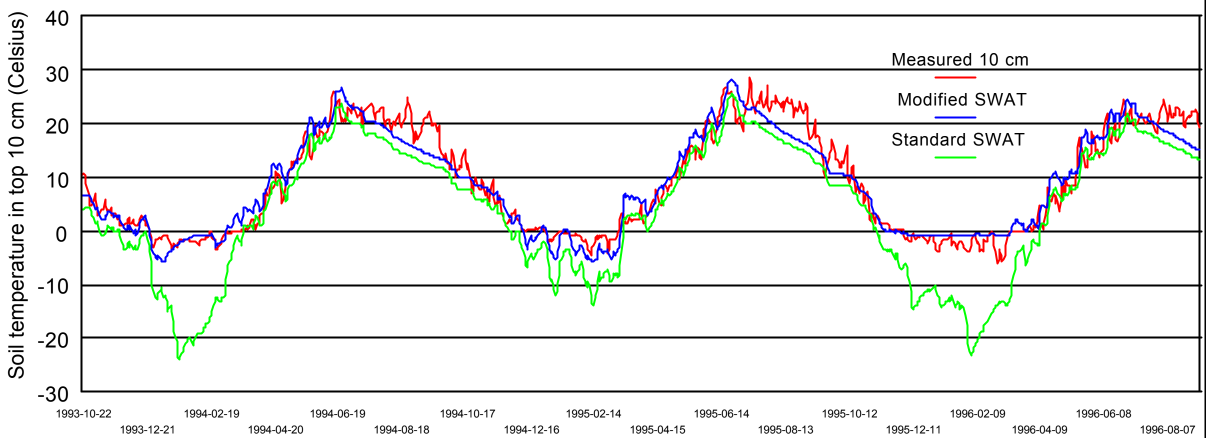
Denitrification and Nitrogen Fixation: The April 2001 version of SWAT2000 differed from the November 2000 version in that the soil water denitrification point parameter (SNDCO) was changed from 1.1 to 0.95. Prior to this change, little or no denitrification occurred, which is not correct. Unfortunately, this change produced too much denitrification with the data sets that were utilized in this project. In a typical cropped system, about 15% of the applied fertilizer and 5% of total mineralized N could be lost through denitrification (Miller and Gardiner 1996, Tables 10-3 and 10-4). To approximate the expected amount of denitrification, the model code was changed to allow adjustments of the SNDCO and denitrification (CDN) parameters. More recently, model developers have also now included these parameters as inputs in the "readbsn" file of the March, 2003 version of SWAT to give model users greater flexibility. Therefore, the modification utilized in this project simply reflects that there is a range of acceptable values. The final values for these parameters were changed from initial defaults of 0.95 to 0.99 for SNDCO, and from -1.4 to -0.3 for CDN. Without these changes, too little soil nitrogen was available for residue decay so decay rates were lower than expected. If residue decay rates are too slow, the model may simulate little difference in soil losses between moderate levels of conservation tillage and no-till.

When soybean growth was simulated, odd effects were noticed whereby increasing the harvested fraction of the crop actually decreased sediment yield instead of increasing it. This problem was traced to too little nitrogen being present because either there was too little fixation or too much removal of nitrogen at harvest. Insufficient nitrogen translated into slow decay rates and apparently, too much residue buildup. To remedy this situation, the nitrogen fixation coefficient (fixco) was changed from the default of 1 to 0.95 by modifying the code. Altering this coefficient had several effects, including increasing nitrogen fixation, and the overall nitrogen available for residue decomposition. An alternative correction would be to artificially introduce additional nitrogen to the soybean crop. In addition, the proportion of nitrogen removed during harvest of soybean (0.065), which is set in the crop data file, should be investigated to determine if it is too high because this value may need to be lower (see Miller and Gardiner 1996, Table 10-3). Alternatively, the 0.065 value is consistent with data listed by Miller and Gardiner (1996, Table 13-1), depending on whether this source was for the whole crop or just the harvested portion.

The ability of the model to adequately track residue decomposition deserves further investigation because there should be a fair amount of year-to-year carryover of residue in no-till corn fields. Thus, it may be that decomposition should indeed be slower to allow sufficient year-to-year carryover of residue under no-till conditions. However, this aspect of the model is fairly sensitive so it may be difficult to adjust all the necessary parameters correctly without creating an excessive buildup of residue.

Soil Temperature: As shown in Figure 5-1, simulated soil temperatures with the unmodified model code did not adequately reflect observed soil temperatures at a Greenleaf, Wisconsin site, which is located near the centroid of the East River watershed. Importantly, the simulated duration of frozen soil is often longer than it should be. Review of the model code and documentation showed that although the effect on soil temperature of significant amounts of snow cover and residue were reflected through a lagging function, the insulating qualities of these two factors were not directly considered. Thus, soil temperatures during winter months were underestimated and the duration of frozen soil was too long, which had the effect of reducing surface residue decay too much during the predicted cold or frozen soil conditions. Simple changes were made to the model code which: (1) ensured that soil temperatures did not fall below -1.5 °C whenever 9.0 mm of snow cover (as water equivalent), or 800 kg/ha of surface residue were present; and (2) set a minimum soil temperature of -7.5 °C. As shown in Figure 5-1, simulated soil temperatures more closely matched observed values with this modification. The duration of frozen soil was also better simulated by the modified version of SWAT.

Figure 5-1. Measured and SWAT-simulated Soil Temperatures.



Snowfall: Currently, the SWAT model does not utilize measured snowfall. Instead, the snowfall threshold temperature input set by the user is used to determine whether precipitation is either rainfall or snowfall. Obviously, this method is not perfect. The model code for measured climatic inputs is somewhat complicated, so it was not changed to allow snowfall inputs. Instead, the input temperature files were altered by increasing the maximum daily temperature just enough to force the precipitation to fall as rain whenever the average daily temperature was below 1.5 Celsius (snow/rain threshold) and less than 50% of the measured precipitation occurred as snow (assumed 1:10 precipitation/snowfall ratio). Between 1976 and 1997, approximately 60 daily temperatures were changed as a result of this modification.

Measured Temperature Input: The clicon.f file was modified to remove a problem whereby SWAT automatically substituted model-generated temperature values whenever the sum of the measured maximum and minimum daily temperatures was zero degrees Celsius. In the past, the model substituted a generated temperature whenever the maximum or minimum daily temperature was zero degrees Celsius. Although the current code results in less frequent substitutions (about three per year), the code was changed to accept all daily values in the input files (unless the input is -97 degrees Celsius or less, i.e., a "flag" in the data set). Alternatively, the model could only substitute daily values when both the minimum and maximum inputs are zero, or null values. This problem is not as insignificant as it first seems. Precipitation that occurs when the average measured temperature is zero should typically be simulated by default as snowfall, but this is often not the case when simulated temperatures are substituted.

HRU Area Inputs: The readhru.f and readbsn.f files were modified to allow either absolute or fractional area inputs to the HRU files (unmodified model uses fraction of basin area only). Absolute areas in km^2 were utilized in this project to simplify the model input process, and increase the flexibility of input data sets that were created.

WETLAND Routine: The wetland code seemed to have an error. The entire phosphorus contribution from the fraction of the HRU that went into the wetland seemed to be removed, regardless of the values used for pertinent variables, including the phosphorus settling rate. The following modification to the SWAT-2000 "wetlan.f" file was made, and the model seemed to work correctly after this change. Confirmation of this modification has not been received yet from SWAT model developer Jeff Arnold.

```
      !! equation 29.1.1 in SWAT manual
      xx = 0.
!    ** hru_ha is already equal to da_ha * hru_fr, so leave out hru_fr(j)***
!       xx = wet_fr(j) * hru_ha * hru_fr(j)     (note: modified by commenting out and using below instead)
        xx = wet_fr(j) * hru_ha
```

CHAPTER 6. MODEL CALIBRATION

CALIBRATION OVERVIEW

All results reported here were simulated with a version of the SWAT2000 model (USDA-ARS version 4/18/2001) that was modified for this project. Model calibration involves adjusting model inputs within acceptable ranges to obtain a good fit between observed and simulated values. The model was first calibrated for crop yields and biomass, after which it was calibrated for stream flow. After the model was successfully calibrated for flow, the model was calibrated for suspended sediment and phosphorus. In this chapter, calibration of the model for crop yields, stream flow, suspended sediment and phosphorus are discussed. The next chapter describes the model validation/assessment phase where the model is tested to see if it can provide reasonable estimates during other time periods and at other locations. The Upper Bower Creek watershed (LF01_15, 36 km^2) was used as the primary calibration site for stream flow, suspended sediment loads and phosphorus loads (Figure 1-1). This monitoring site is located in the East River Watershed, and it was jointly funded by the USGS and WDNR (USGS Station #04085119). A 1990 to 1994 data set was used for calibrating the model (50 events), while the data set from 1996-97 (17 events), along with data from other sites, were used in the model assessment phase. Most of the data obtained for USGS stations was provided in digital format, both on an event basis, and on a daily basis by the USGS, Madison, Wisconsin.

CALIBRATION - CROP SUB-MODEL

Calibration: Brown County crop yields from 1989 to 1996 were used for calibrating the crop sub-model. Crop yield data from 1993 were not used to calibrate the model because the model was unable to predict the low crop yields that resulted from the unusually wet 1993 weather. Initially, simulated crop yields were much higher than published Brown County values during the 1989 to 1996 crop calibration period (Wisconsin Dept. of Agriculture 1956-96). This observation is important because excessive simulated biomass production also overstates the amount of residue left after the crop is harvested, thereby over estimating the amount of residue that is available to reduce soil erosion. Nutrient cycling would also be adversely affected. Crop yields were calibrated by adjusting each crop's biomass energy factor (BE) in the SWAT crop database file. BE's were reduced from default values of 39, 25, 20 and 35; to 30, 20, 9 and 30 for corn, soybean, alfalfa and oats, respectively, to correspond more closely with published crop yields for Brown County from 1989 to 1996.

Adjusting BE seems reasonable because the daily potential increase in biomass is simply a linear function of BE and photosynthetic active radiation (PAR) within the SWAT model. In addition, stress factors that SWAT does not directly account for such as excessive soil moisture (although denitrification is addressed), herbicide-induced stress, competing vegetation, pests, molds, fungi and other diseases are indirectly addressed by lowering BE to obtain smaller yields. The alfalfa BE of 9 is much lower than the 14 which was utilized by Baumgart (1998). However, in that project, the Hargreaves PET method was utilized which unknowingly understated crop yields and biomass because relative humidity was inadvertently set to zero by the version of the model used at that time. The problem with relative humidity has been corrected by the author of this project (see Chapter 5).

Potential Heat Units (PHU's) were adjusted so that the average date of maximum biomass coincided with the expected maturity date: 90, 85, 60 and 90 days after germination for corn, soybean, alfalfa and oats respectively. The PHU's were set to 1300 for corn and soybean, and 1000 for alfalfa.

After calibration, the simulated average crop yield from 1989 to 1996 (excluding 1993) closely matched the measured yield of the crops simulated in this study. However, further analysis showed that the simulated average annual yields were only slightly correlated with the measured annual yields from 1989 to 1996 (R^2 = 0.41 for corn grain). Apparently, factors affecting crop yields from year to year were not fully accounted for by the calibrated model. This is not necessarily surprising, for many factors that determine annual crop yield were not utilized in this modeling effort, including daily measured solar radiation.

Average 1989-96 crop yields were closer to the observed values when the Penman-Montieth PET method was used (R^2 = 0.65 for corn grain). Total denitrification losses were greater with this PET method, thereby reducing yields and biomass during wet years and providing a better match between observed and simulated crop yields. However, water yields did not correspond as well with this PET method so the Hargreaves-Samani (1985) PET option was utilized instead.

Additional Input Modifications: In addition to modifications that affect crop yield, the alfalfa minimum erosion ratio value (Cmin) was increased from the default of 0.01 to 0.10. This change was made because the former value produced essentially no erosion from alfalfa fields, which is unlikely. Modification of the "C factor" routine in SWAT tempered this increase in Cmin by accounting for residue and above ground biomass separately.

CALIBRATION - HYDROLOGY

Upper Bower Creek: The first calibration step for hydrology was to match the simulated 25 year (1976-2000) average annual water yield to an expected value of about 200 mm for the Upper Bower Creek subwatershed, which was based on the regional runoff values published by the USGS (Gebert et al. 1987). Initial calibration was accomplished by altering the evapotranspiration coefficient (ETCOEF) until the simulated and expected long-term annual water yields were similar. As described in Chapter 5, the ETCOEF variable is the leading coefficient that was added to the SWAT FORTRAN code to permit adjustments of the potential evapotranspiration (PET) equations utilized by SWAT. Therefore, an ETCOEF value of 1.0 produces the same runoff results as the un-altered version of SWAT.

Calibration was further refined to match the USGS-measured total water yield for the October 1990 to December 1994 calibration period, as well as statistical measures from 52 runoff events which occurred within this period. An ETCOEF value of 0.81 was selected for all simulations, along with the Hargreaves and Samani (1985) PET equation. All three optional PET equations were tested for fit with the Bower Creek 1990-94 hydrograph. Although the different methods did not produce great differences, the selected method produced a somewhat better match with observed runoff events and overall water yields. The selected ETCOEF value of 0.81 decreased evapotranspiration and increased runoff. Stream flow in the Bower Creek watershed is very "flashy", so no attempts were made to alter the initial values used in the groundwater module, which assumed that percolation to the deep aquifer was low.

Base flow analysis of the calibration data set was conducted using a computer program developed by Arnold et al. (1995). Based on the second iteration with this computer program, base flow comprises approximately

15% of stream flow in Upper Bower Creek. Simulated total base flow for the calibration period was 15.2% with the aforementioned ETCOEF value of 0.81, and default values for available water capacity (AWC), curve number (CN), plant uptake compensation factor (ESPO) and soil evaporation compensation factor (ESCO).

Additional attempts were also made to calibrate the model by altering different parameter combinations such as the available water capacity (AWC), curve number (CN) and plant uptake compensation factor (EPCO), instead of ETCOEF. Values were adjusted to increase water yield which was substantially below the expected total yield with the defaults. The soil evaporation compensation factor (ESCO) was left at 1.0 because lowering this value decreases water yield, which was already too low. All of the parameters had to be modified to near the maximum or minimum end of their respective acceptable ranges to obtain the required total water yield. However, the results were not nearly as good as when only the ETCOEF was modified. For example, the measured water yield in September 1992 was 15 mm. Calibrations involving the ETCOEF produced a water yield of 26 mm for this same period; whereas, calibrations involving different combinations of the other set of parameters produced water yields from 50 to 63 mm. The EPCO value needed to be decreased to nearly zero to obtain adequate water yields without using the ETCOEF modification. It appears that when the EPCO was set this low, the model did not allow a sufficient reduction in soil moisture during dry periods, resulting in over-stated water yields when rainfall fell shortly after these dry spells. Adjusting these other parameters to get the expected total water yield had the undesirable effect of simulating essentially no groundwater recharge to the stream when EPCO was reduced to nearly zero. In addition, very low values of EPCO decreased crop yields substantially as plants did not get sufficient water under dry conditions.

Two other variations were also tried: (1) seasonally varying evapotranspiration according to the day of the year, with peak evapotranspiration occurring at the end of July; and (2) reducing ETCOEF, AWC and CN. Both of these variations produced comparable results to simply reducing ETCOEF below one, but with greater complexity. The second method produced a better match with both observed annual water yields and observed runoff events following dry spells. Therefore, the ETCOEF was the primary means of calibrating the water yield, and small adjustments to the other parameters were made to get a slightly better fit between simulated and observed values. The final hydrologic calibration parameters were set as follows: ETCOEF = 0.806; AWC = 0.97; CN = 0.985; ESCO = 1.0; and EPCO = 1.0. With these final calibration values, base flow comprised the same proportion of total stream flow during the calibration period as the observed flow. A slightly better fit was produced with another set of hydrologic calibration parameters (ETCOEF = 0.803; AWC = 0.98, CN = 0.98), but base flow was higher than desired.

With the exception of the subbasin-wide curve number adjustment factor, recommended default NRCS curve numbers were used for all model simulations. Throughout the rotation, default curve numbers were input to reflect changes in crops and tillage. Attempts were made to improve the fit between the simulated and observed hydrographs by altering the default curve number for alfalfa, but no improvements were observed.

Simulated and observed stream flow volumes from 52 events which occurred between 1990 and 1994 are compared in Figure 6-1. A total of 50 runoff events were computed by USGS from 1990 to 1994 for the purposes of evaluating non-point contamination (Walker et al. 2001). However, USGS flow data from two very large 1993 runoff events were also included in the data set for a total of 52 runoff events, although these events could not be used for phosphorus and TSS analysis because no water quality samples were collected during these events. Total water yields from these two large runoff events were 58% and 108% greater than the largest event in the data set with only 50 events. The coefficient of determinations, (R^2), as determined through linear regression analysis, was 0.81 for the 52 event stream flow volumes (0.66 for 50 events).

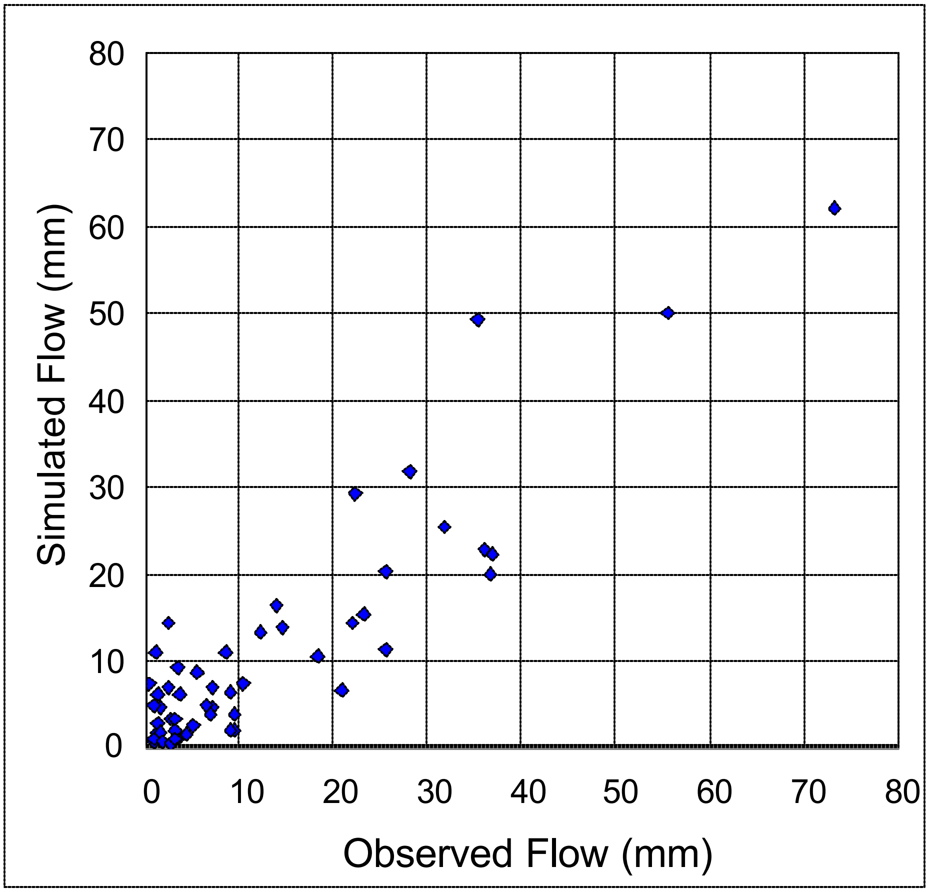


Figure 6-1. Observed and simulated flow events at Upper Bower Creek, 1990-94 calibration period.

The Nash-Sutcliffe coefficient of efficiency (NSCE) was also used to assess the ability of the model to match observed values (Nash and Sutcliffe 1970).[12] The following equation is used to compute the coefficient of efficiency:

$$NSCE = 1 - \frac{\sum_{i=1}^{n} (xo_i - x_i)^2}{\sum_{i=1}^{n} (xo_i - xo^*)^2} \quad \text{(Eq. 8)}$$

---

[12] The Nash-Sutcliffe coefficient method has been recommended as a goodness-of-fit criterion by the American Society of Civil Engineers Task Committee on Evaluation Criteria for Watershed Models (ASCE Task Committee 1990).

where n is the total number of events, x_i is the simulated flow or TSS load for an event, xo_i is the observed event flow or load, and xo^* is the mean flow or load for all observed events. A NSCE value of 1 indicates a perfect fit. The NSCE for 52 total event stream flow volumes was 0.80 (0.64 for 50 events). These statistical measures, along with the relationship shown in Figure 6-1, indicates that there was an acceptable level of correspondence between simulated and observed events.

Annual simulated and observed stream flows were: 1991 (201 vs 180 mm), 1992 (210 vs 230 mm), 1993 (344 vs 370 mm), and 1994 (132 vs 102 mm), respectively. The maximum relative difference was 30% in 1994, when the lowest flow occurred; thereby, suggesting that the model may have greater difficulty simulated water yields during dry periods. Total simulated water yield during the entire October 1990 to December 1994 calibration period was 906 mm compared to the observed total of 902 mm.

Monthly simulated and observed stream flows from 1991 to 1994 are shown in Figure 6-2, along with monthly precipitation from the USGS weather stations. The precipitation scale in Figure 6-2 is twice that of the scale for flow. The NSCE and coefficient of determination (R^2) were 0.86 and 0.87, respectively, for monthly flows during this period, indicating a good correspondence between observed and simulated flows. However, the simulated flow of 62 mm in April 1993 was much less than the observed value of 111 mm. As shown in Figure 6-2, total observed precipitation during April, 1993 was 95.5 mm, <u>which is less than the observed flow</u>. While ice conditions may contribute to overstated measured flows in earlier months, it is less likely to occur in April. However, ice conditions may have played some role during the first half of this month, as all of March, 1993 was affected by ice (USGS Water Resource Data Book, 1993 Water Year). In addition, above average groundwater recharge, frozen soil surface and thawing sub-soil may all contribute to the high measured flow exhibited in April 1993, which the model was not able to predict. Given that the measured flow was greater than the measured precipitation in April 1993, it is doubtful that the model could be expected to match the simulated and observed flow in this month without introducing faulty data or assumptions somewhere in the model framework.

Daily simulated and observed stream flows during the October 1990 to December 1994 calibration period are shown in Figures 6-3a through 6-3c. Average daily precipitation from the USGS weather stations within Upper Bower Creek is also shown in these figures. With some exceptions, general peaks and recessions were tracked by the model. However, simulated flows during the May-June, 1991 period were overstated (33 mm vs 6 mm, see Figure 6-3a). An extended period with much higher than normal temperatures was experienced from May 10-31, 1991, with the average daily temperature 10 degrees Fahrenheit above the 30 year normal. The calibrated model and data set may be overstating runoff when soil moisture has actually been greatly depleted due to high temperatures, clear skies, and high evapotranspiration. For this project, daily solar radiation values were simulated by the model. Results may have been better if measured values were utilized. Importantly, the simulated water yield for the May-June, 1991 period was only slightly better when the ETCOEF was set to the default value of 1.0 (30 mm), thereby indicating that reducing evapotranspiration by a factor of 0.806 through the model modification was not primarily responsible for overstating water yields during this period.

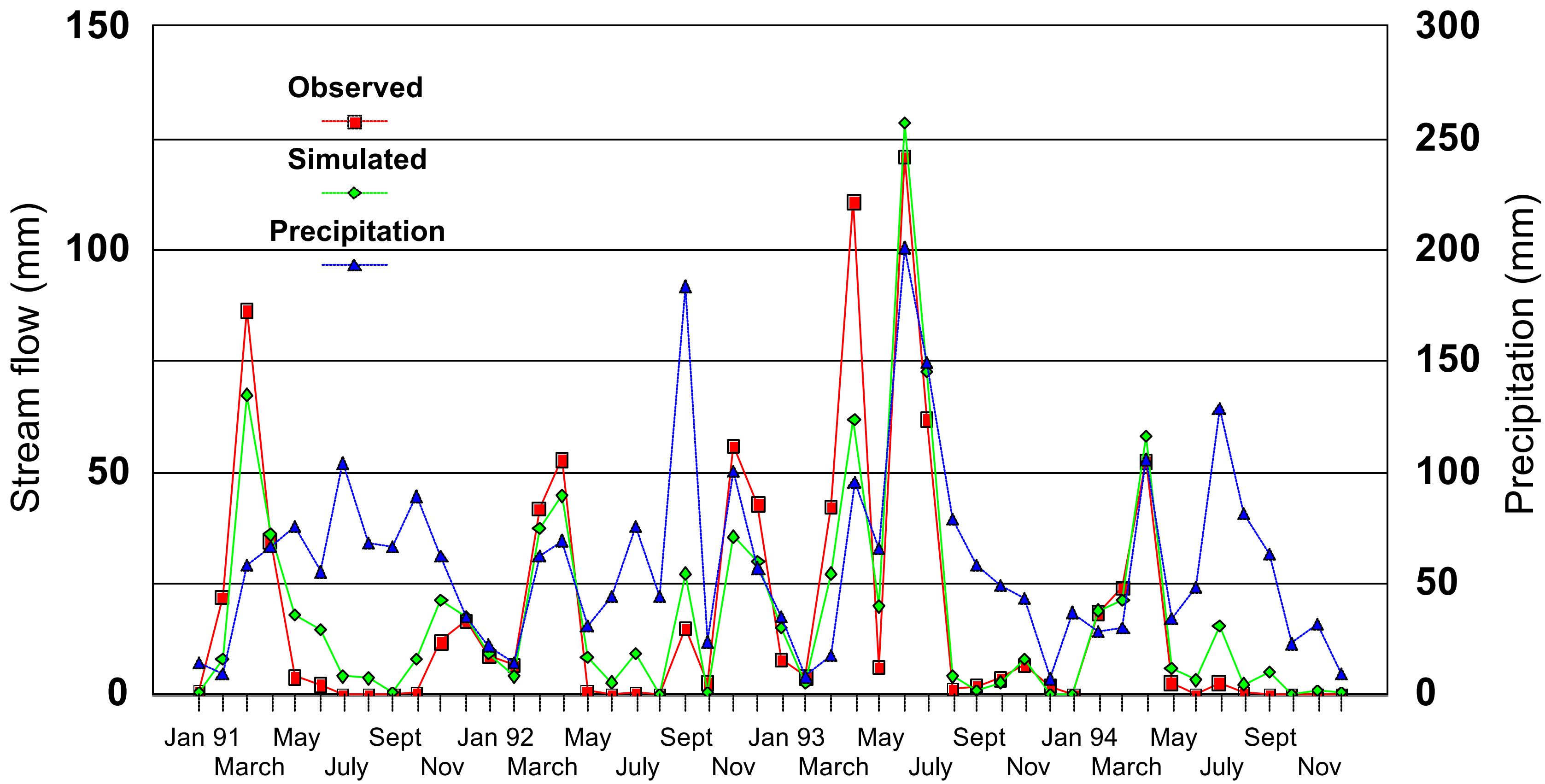


Figure 6-2. Observed and simulated monthly stream flow - Upper Bower Creek. 1990-94 calibration period. Precipitation from USGS weather stations is also shown.

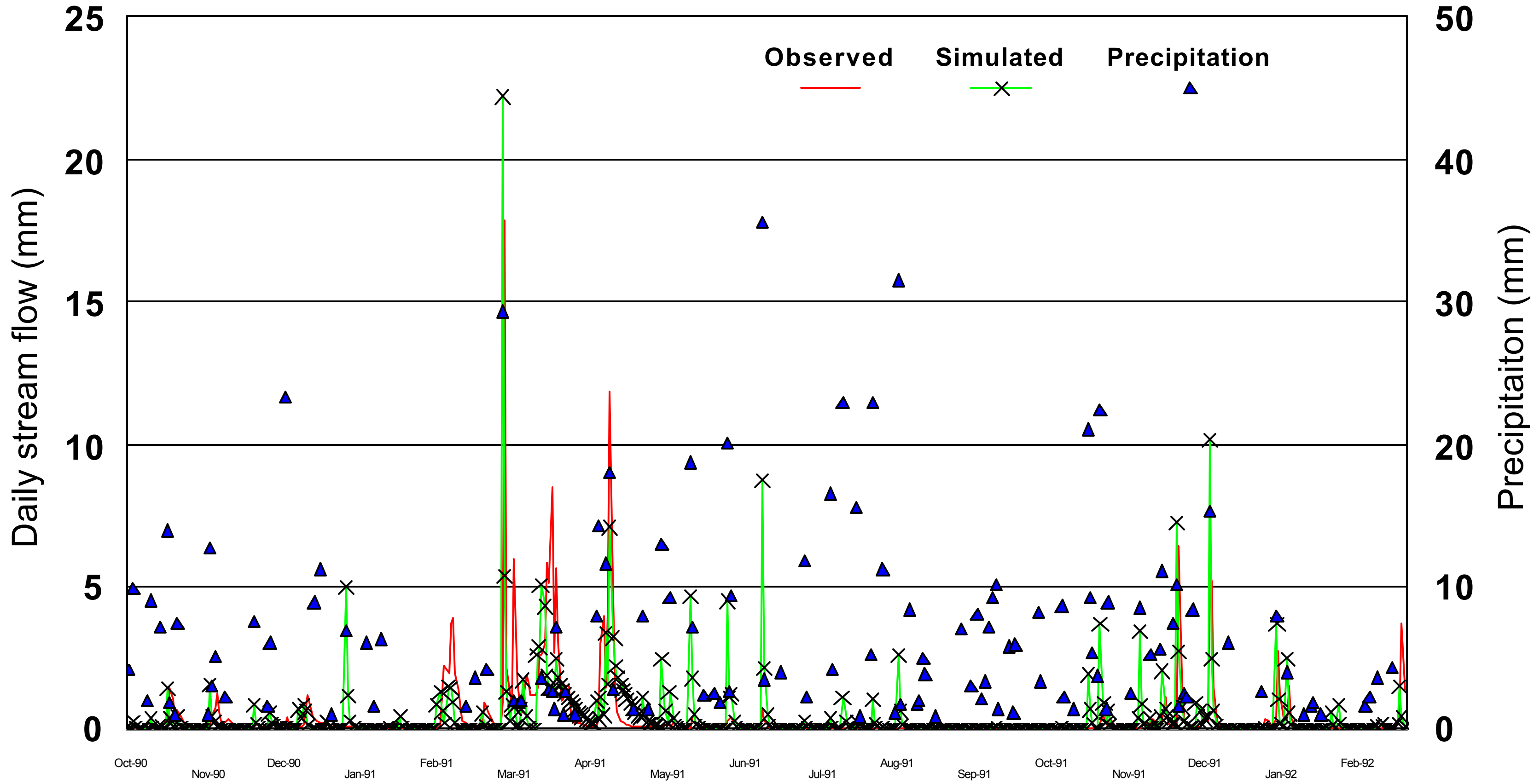


Figure 6-3a. Observed and Simulated Daily Stream Flow - Upper Bower Creek. 1990-94 calibration period. Precipitation from USGS weather stations is also shown.

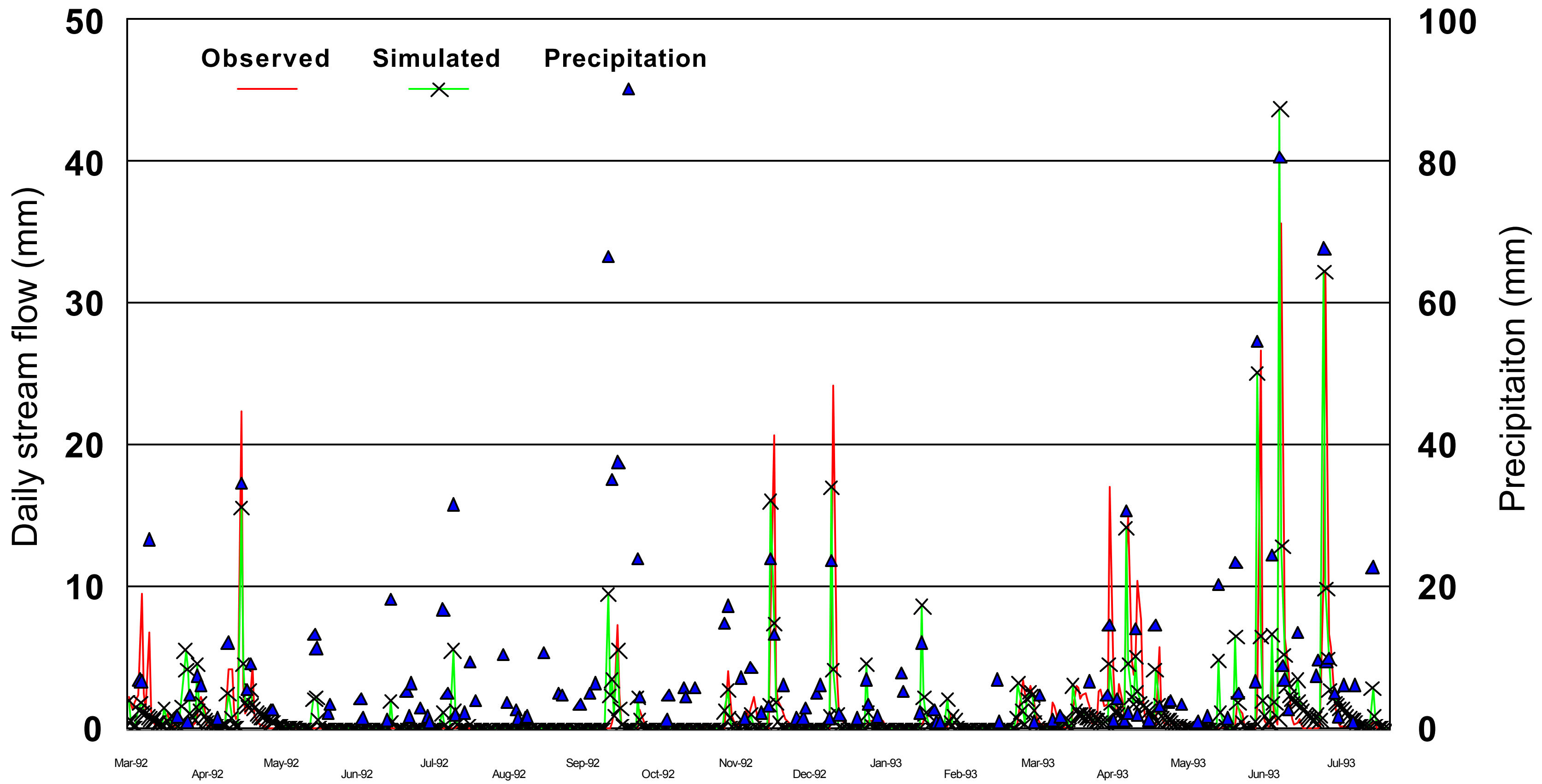


Figure 6-3b. Observed and Simulated Daily Stream Flow - Upper Bower Creek. 1990-94 calibration period. Precipitation from USGS weather stations is also shown.

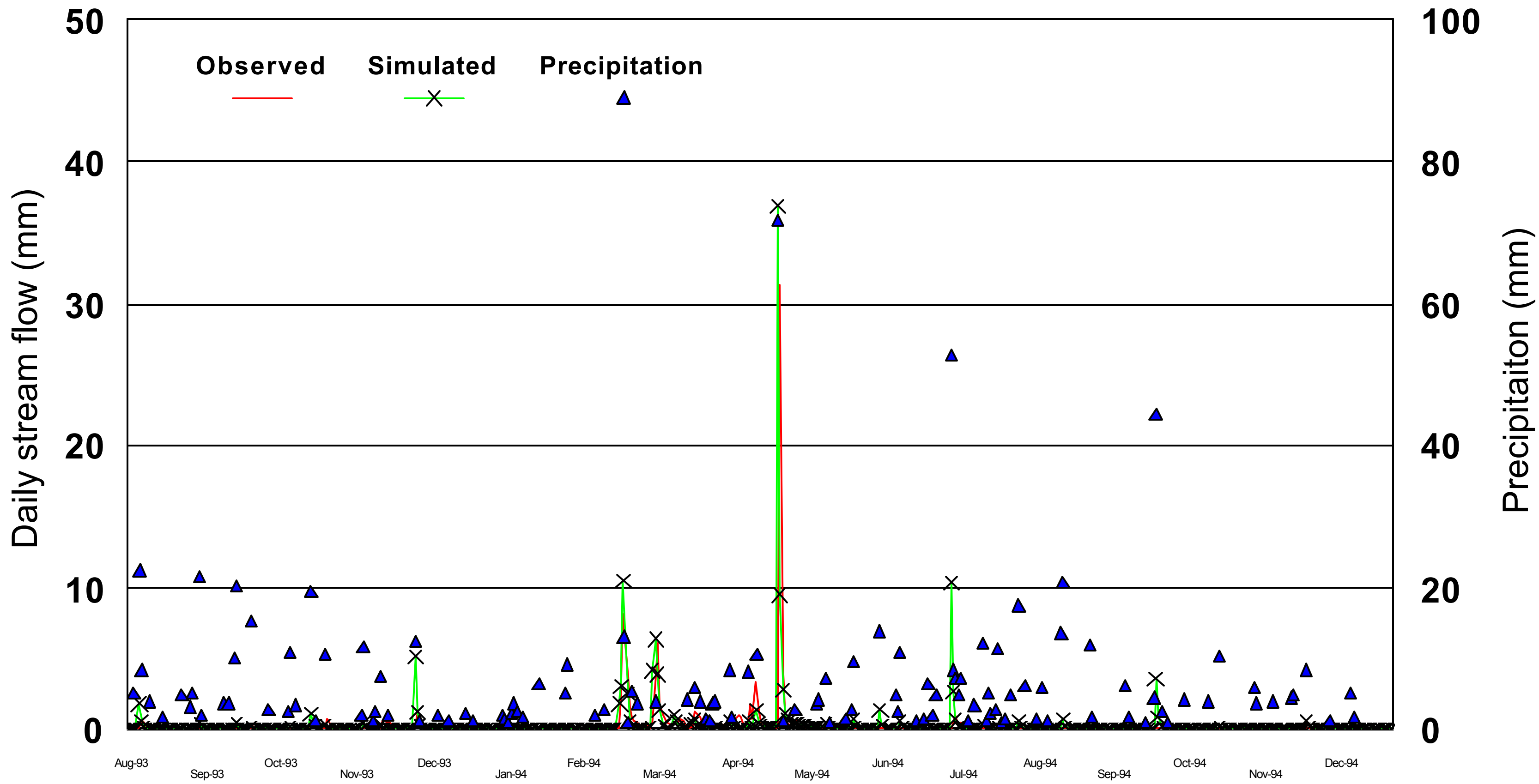


Figure 6-3c. Observed and Simulated Daily Stream Flow - Upper Bower Creek. 1990-94 calibration period. Precipitation from USGS weather stations is also shown.

While the previous example implies that the model overstated soil moisture during dry periods, other evidence suggests that the model is capable of reasonable predictions after long dry spells. For example, simulated flows during a rather wet period in September, 1992 period matched observed values fairly well (Figure 6-3b). Approximately 140 mm of rainfall occurred from Sept. 14 to Sept. 18, 1992 (66.7, 0.0, 35.2, 0.1, and 37.6 mm, respectively). Despite this high precipitation event, the calibrated model predicted only 23.8 mm of total stream flow from this series of events, which compares relatively favorably to the observed value of 12.3 mm. After an additional 28.5 mm of rainfall occurred approximately one week later, the additional observed total stream flow was 2.9 mm compared to a simulated flow of 3.1 mm. Given the relatively impermeable nature of the soils in Bower Creek, it appears that the model was able to reasonably track soil moisture, even during an extreme period. Overall, a comparison between precipitation, observed flow and simulated flow indicates that the model was capable of adequately estimating stream flow over a fairly wide range of soil moisture conditions (Figure 6-3).

Duck Creek at County Highway FF: The USGS monitoring station located on Duck Creek at County Highway FF (station #04072150) was used to calibrate the model to the hydrologic characteristics of this watershed. This site is located at the outlet of subwatershed LF05_7, and the drainage area at this point is 276 km^2 (Figure 1-1). The calibration period was Jan. 1, 1989 to Dec. 31, 1996.

The amount of water lost through percolation to the deep aquifer and transmission losses from subwatershed tributaries were altered to match simulated and observed total stream volumes. The curve number was also reduced by a factor of 0.97 compared to a factor of 0.985 for all other watersheds. Percolation to the deep aquifer is considered a loss from the system, and it is determined by multiplying computed soil percolation by the deep percolation coefficient. To calibrate the model, the deep percolation coefficient in the groundwater file for Duck Creek was set to a value of 0.4, in contrast to the 0.04 value used for other watersheds. The resulting 17 mm/year of water lost to the deep aquifer in the Duck Creek watershed corresponds to data reported by Krohelski (1986; Figure 9), who estimated a recharge rate to the lower aquifer of 0.4 inches/year for Brown County, while western areas in the vicinity of Duck Creek were presumed to exceed the average. Krohelski (1986) found that stretches of the main branch of Duck Creek were losing. Therefore, the subwatershed tributary transmission losses were increased until the simulated average annual water yield matched the observed flow. Elsewhere in the subbasin, net transmission losses were assumed to be negligible. Ideally, net transmission losses in the main routing reaches would have instead been raised to calibrate the model for the LF05 watershed. However, unintended effects precluded using this means of simulating expected increased transmission losses in this watershed.

Based on the second iteration of the base flow analysis program, base flow comprised about 30% of the USGS-measured 1989-96 stream flow. The first iteration of the computer program calculated a baseflow proportion of 48%. It is likely that the second iteration is much closer to the actual value, so it is estimated that about 30 to 37% of stream flow is base flow. With the final calibration parameters, simulated base flow for the calibration period was also 30% on the second iteration.

Average annual simulated and observed water yields during the Jan. 1, 1989 to Dec. 31, 1996 calibration period are listed in Table 6-1. The average annual simulated and observed water yields for the entire period are essentially identical. Simulated annual flows were relatively close to the observed flows except in 1989, 1995 and to a lesser extent 1994. Both 1989 and 1995 were relatively hot, dry years. The model overstates annual flows during dry years, just as it did in the Bower Creek subwatershed on an event basis.

Table 6-1. Annual Flow at Duck Creek (1989-1996 Calibration period).

	Observed Flow (mm)	Simulated Flow (mm)	Relative difference
1989	70	99	42.0%
1990	226	191	-15.6%
1991	164	146	-11.1%
1992	235	181	-23.1%
1993	331	322	-2.6%
1994	151	192	27.1%
1995	94	160	70.0%
1996	230	203	-12.1%
average	**188**	**187**	**-0.5%**

Average monthly simulated and observed flows during the calibration period are listed in Table 6-2. A NSCE of 0.80 was determined for the monthly average, while R-squared was 0.83. On average, March and April simulated flows are depressed; whereas, May and July through October simulated flows are substantially higher than the observed values.

Table 6-2. Average Monthly Flow at Duck Creek (1989-96 calibration period).

	Observed Flow (mm)	Simulated Flow (mm)
Jan	2.3	3.0
Feb	3.7	3.3
March	43.4	26.0
April	49.8	37.9
May	15.2	22.6
June	31.0	36.1
July	12.7	19.5
August	2.3	5.8
Sept	2.8	5.4
Oct	4.2	6.4
Nov	12.9	14.5
Dec	7.3	6.1

Monthly simulated and observed water yields for Duck Creek at CTH FF are compared in Figure 6-4. NSCE and R-squared values of 0.82 were determined for the 96 months in the calibration period. Simulated and observed monthly stream flows were generally close. The large flood event of June 23 and 24, 1990 was predicted fairly closely on a monthly total basis. However, simulated flows were much lower than observed flows in March 1990 and April 1992, and at times substantially higher than observed flows in late summer and fall. Some of these differences may be attributed to the model or how it was applied. Ice conditions and snow melt may also contribute to discrepancies. However, it is important to note that only a single weather station was close to any of the subwatersheds in this monitored watershed. Even this station is actually just outside of the drainage area.

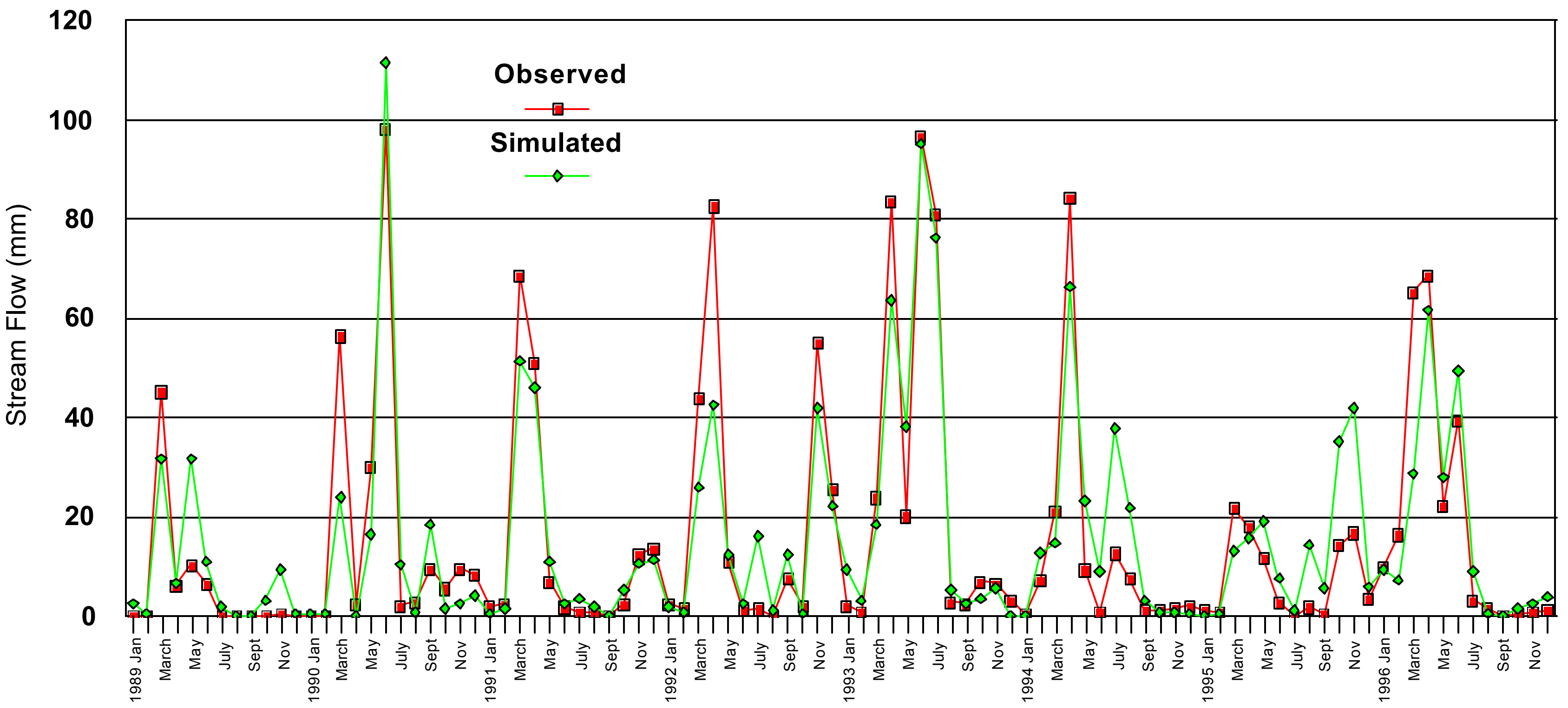


Figure 6-4. Observed and simulated monthly stream flow - Duck Creek at CTH FF (1989-96 calibration per

Simulated and observed daily flows were also compared by delaying the daily flows by one day to bring them in phase with the observed flows; plus, a running average was used to smooth the hydrograph somewhat. A 3-day running average was selected on the basis of a combination of the highest NSCE, and the best visual fit between the observed and simulated hydrographs. Weighted, running average factors of 0.30, 0.6, and 0.1 were used, along with the one day delay mentioned earlier. Most of the largest daily flows were close to the measured values, even without the smoothing operation. However, the smoothing operation helped provide a closer fit for very large events, such as occurred on June 23 and June 24, 1990.[13] Simulated daily stream flows generally tracked the observed values during the 1989-96 calibration period (set of 5 figures available upon request). However, there was a tendency for the model to over-predict small events, particularly during the summer and fall months. A NSCE of 0.69 was determined for the calibration period, while R-squared was 0.69. R-squared was 0.61 when only a single day delay between simulated and observed values was utilized (without any smoothing of peaks).

CALIBRATION - TOTAL SUSPENDED SEDIMENT

The SWRRBWQ (Arnold et al. 1994), EPIC (Sharpley and Williams 1990), SWAT (Arnold et al. 1996), and APEX (Williams et al. 1995) models all use a form of the Modified Universal Soil Loss Equation (MUSLE) shown below to compute sediment yield from a watershed on a daily basis.

$$\text{MUSLE: } Y = a\,(Q)^b (q_p)^c (DA)^d\ [(K)\,(C)\,(PE)\,(LS)] \qquad \text{(Eq. 9)}$$

where:

Y	=	sediment yield in metric tons/ha (Mg/ha)
Q	=	surface runoff volume in mm
q_p	=	peak flow rate in mm/hr
DA	=	drainage area in hectares
K	=	soil erodability factor
C	=	crop management factor
LS	=	slope-length and slope-steepness factor
PE	=	erosion control practice factor
a,b,c,d	=	constants normally set at a = 1.586, b & c = 0.56, d = 0.12 (user-specified values can be used where there are sufficient data for calibration)

However, the sediment yield equation in the unmodified version of SWAT does not allow user-specified values for the a, b, c and d coefficients. The MUSLE equation in SWAT is also in a slightly different form; whereby, the variables are not in unit area format (e.g., Q is in cubic meters instead of mm and Y is in metric tons instead of tons/ha). To permit calibration of the model to site-specific conditions, Equation 9 was therefore inserted into the SWAT code (in ysed.f).

TSS loads from the Upper Bower Creek USGS monitoring station were used to calibrate the model. While a total of 50 runoff events were computed by USGS from 1990 to 1994, only 29 of these events were used for calibration of the sediment equation. This partial data set was selected primarily on the basis of the largest measured events, as well as events which occurred directly after or before the major events (for simplicity).

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[13] For the Duck Creek site, peak flow during the June 23, 1990 event could only be estimated by USGS. This estimate was based on: (1) a gage height of 21 ft., which was indicated by flood marks; and (2) the rating curves was extended above 1,500 cfs on the basis of contracted flow measurements.

The excluded events had individual TSS loads of less than 50 tons[14]. Information presented later in this section will show that excluding these data points had no meaningful effect on the statistical fit of the calibrated model; in addition, the calibration coefficients were also virtually unaffected.

The Nash-Sutcliffe coefficient of efficiency was used as the primary criterion to optimize the values for a, b, c, and d in the MUSLE sediment equation. The following calibration data set for MUSLE variables was selected on the basis of maximizing the NSCE, and visual comparison of simulated and observed TSS loads: a = 0.0245, b = 1.6, c = 0.0 , d = 0.0. Another data set produced slightly better results for suspended sediment (a = 0.0178, b = 1.7), but the fit for phosphorus was not as good, so the selected parameter set was a compromise solution which provided the best overall fit for both sediment and phosphorus, while also producing the correct fraction of dissolved phosphorus. The variable "c" was set to zero, which eliminates the simulated peak flow from affecting sediment yield.[15] In SWAT, peak flow is a function of simulated flow volume and average monthly precipitation intensity, rather than measured intensity. Although it should still be theoretically better to have the monthly precipitation intensity included as a factor in MUSLE, little difference was detected during the calibration phase.

Figures 6-5a to 6-5c compare simulated and observed TSS loads for the entire TSS calibration data set of 50 events. These figures represent the same data, but the scale on the vertical axis is different. The NSCE and R-squared values for the entire data set were both 0.93; whereas, NSCE and R-squared were 0.95 for the 29 data points used to calibrate the model. Therefore, excluding some of the events from calibration had little effect on the statistical fit of the data. The calibrated model did not simulate TSS loads during moderate and small events (less than 250 tons) as well as during larger events. The greatest relative deviations occurred at low levels (Figure 6-5c). Monthly simulated and observed TSS loads were also compared for the 1991-94 calibration period. The NSCE and R-squared values for these months were both 0.91, indicating a good correspondence between simulated and observed loads. Two large events in June and July 1993 were excluded from this analysis because the suspended sediment loads were only estimated by USGS; that is, no samples were collected during these events. Importantly, the USGS did not measure TSS or phosphorus concentrations at a sufficient frequency to precisely determine loads for all events; rather, this was only done for the 50 measured events. Therefore, monthly loads based on the USGS daily load data should be considered very good estimates, rather than absolute observed values.

There was insufficient water quality data at the Duck Creek site to calibrate the model for TSS at this location because sampling by the USGS primarily occurred during low flow events. This was unfortunate, for the sediment routing component of the model needs additional refinement to better define the user-adjustable parameters in the sediment deposition/degradation sub-model.

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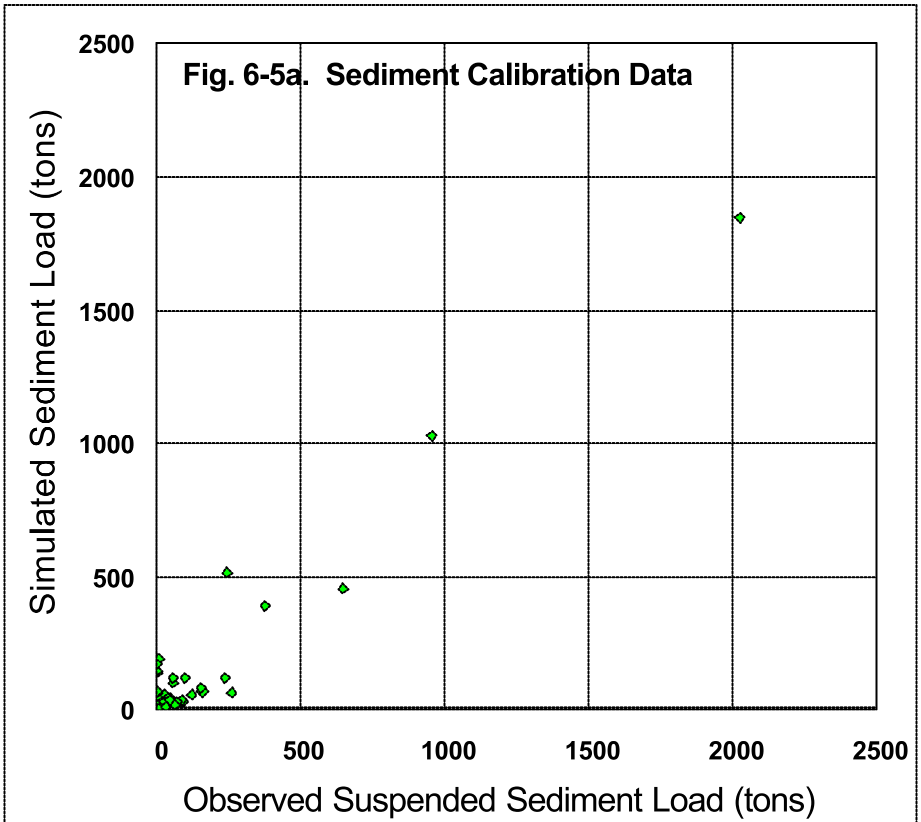
[14] All TSS loads are reported in metric tons (1,000 kg or Mg).

[15] Comparisons between simulated sediment reductions associated with switching from conventional tillage to mulch till produced a large range in reductions (36% to 56%) when HRU areas were changed from a large area (CT) or small area (MT) to the reverse (small area CT and large area for MT). If there was no difference in the areas, then the reduction should have been 46%. By not including the peak flow in MUSLE, the range was greatly reduced to 42% to 53%. However, this observation is no longer applicable to the version of SWAT applied here, which was modified to use subwatershed channel lengths in place of HRU channel lengths when time of concentration is computed in the model.



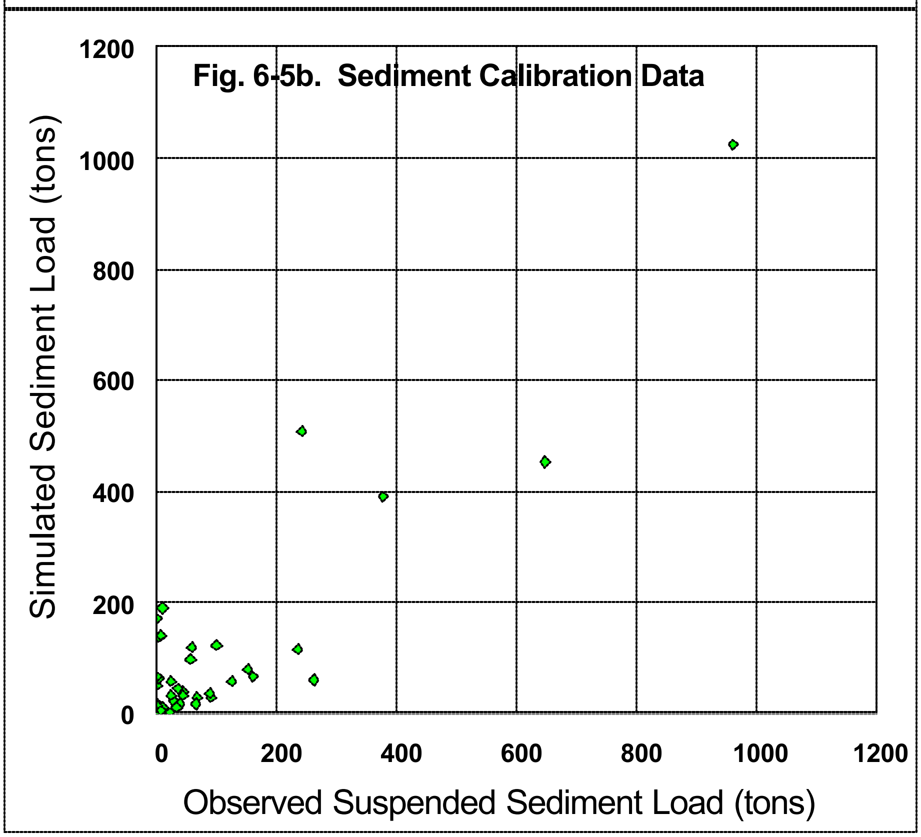


Figure 6-5. Observed and simulated suspended sediment loads at Upper Bower Creek. 1990-94 calibration period.

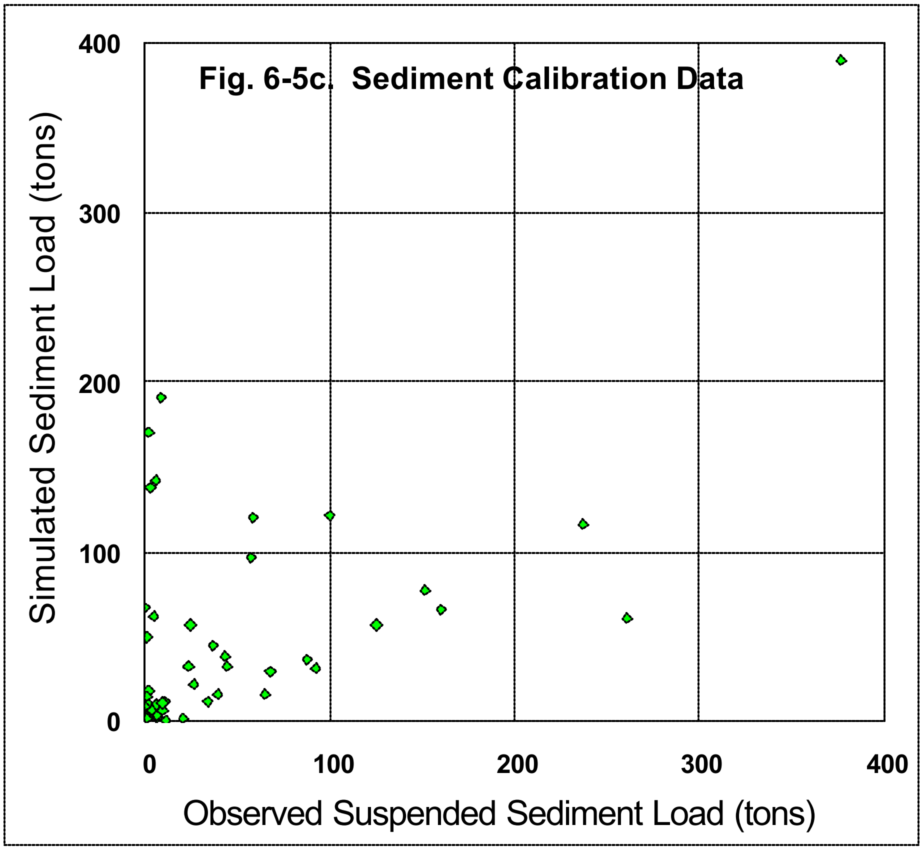


Figure 6-5. Observed and simulated suspended sediment at Upper Bower Creek. 1990-94 calibration period.

CALIBRATION - PHOSPHORUS

<u>Upper Bower Creek</u>: After the model was calibrated for flow and TSS, calibration could proceed. However, calibration of the phosphorus component of the model also took place during earlier phases because the amount of phosphorus and nitrogen fertilizer applied to a field affect crop growth, residue formation and decay, potential evaporation, and therefore flow and TSS loads. Hence, model calibration is truly an iterative process, rather than a step-by-step process.

To calibrate the phosphorus component of the model, the phosphorus availability index (PSP) and the phosphorus soil partitioning coefficient (PHOSKD) were the primary parameters which were adjusted to obtain total annual phosphorus loads that closely matched observed values. These parameters were then fine-tuned to obtain a close match between simulated and observed event loads, while also obtaining a simulated dissolved phosphorus load that was 30% of the total phosphorus load. The final calibration parameters were set as follows: PSP = 0.390; PHOSKD = 185; phosphorus percolation coefficient (PPERCO) = 20; and the phosphorus uptake parameter was set to 14. A better fit between observed and simulated phosphorus loads could've been obtained by setting the PSP to 0.445 and PHOSKD to 130, which also increased the fraction of dissolved phosphorus. However, the percentage of dissolved phosphorus that resulted was about 48%, compared to an estimated expected value of about 30%, which is intentionally lower than the concentrations and loads that have been measured in streams within the subbasin[16].

Simulated and observed phosphorus loads for the entire calibration data set of 50 events are compared in Figure 6-6. The NSCE and R-squared values for the entire data set were 0.75 and 0.79, respectively; whereas, NSCE and R-squared were 0.80 and 0.87, respectively for the 29 data points used to calibrate the model. The calibrated model did not simulate phosphorus loads during moderate and small events as well as during larger events. Five of the eight largest absolute deviations between observed and simulated load events occurred during the month of April; two occurred during snow melt events in March and the other event occurred in mid December. With the exception of one of these eight events, simulated runoff was also markedly under-predicted. Simulated suspended sediment was also markedly under-predicted for all but one of the eight events. Had runoff been better predicted during these eight events, simulated suspended sediment and phosphorus loads would also have been closer to the observed values. The data suggests that the model generally under predicts runoff during the month of April, when the soil is often highly saturated; conversely, the model overstates runoff after long dry spells when soil moisture has been greatly depleted. The precise cause for this problem has not yet been determined, but it may be related to the same reason that average annual runoff is greatly understated by the un-modified version of SWAT (without using extreme parameter values to calibrate the model).

Monthly simulated and observed phosphorus loads were also compared for the 1991-94 calibration period. The NSCE and R-squared values for these months were both 0.71 and 0.72, respectively, indicating a fair correspondence between simulated and observed loads. As was done for suspended sediment, two large

[16] For this project, it is assumed that the dissolved phosphorus fraction coming off the fields under current conditions is lower than what is observed in stream, which is typically between 45 and 70% dissolved phosphorus. This is an important assumption because many BMP's are not as effective at reducing dissolved phosphorus as they are at reducing phosphorus attached to soil particles (e.g., conservation tillage, buffer strips). If this assumption is too low, then the effectiveness of some BMP's in reducing total phosphorus may be lower than predicted by the model.

events in June and July 1993 were excluded from the monthly statistical analysis because the phosphorus loads were only estimated by USGS; that is, no samples were collected during these events.

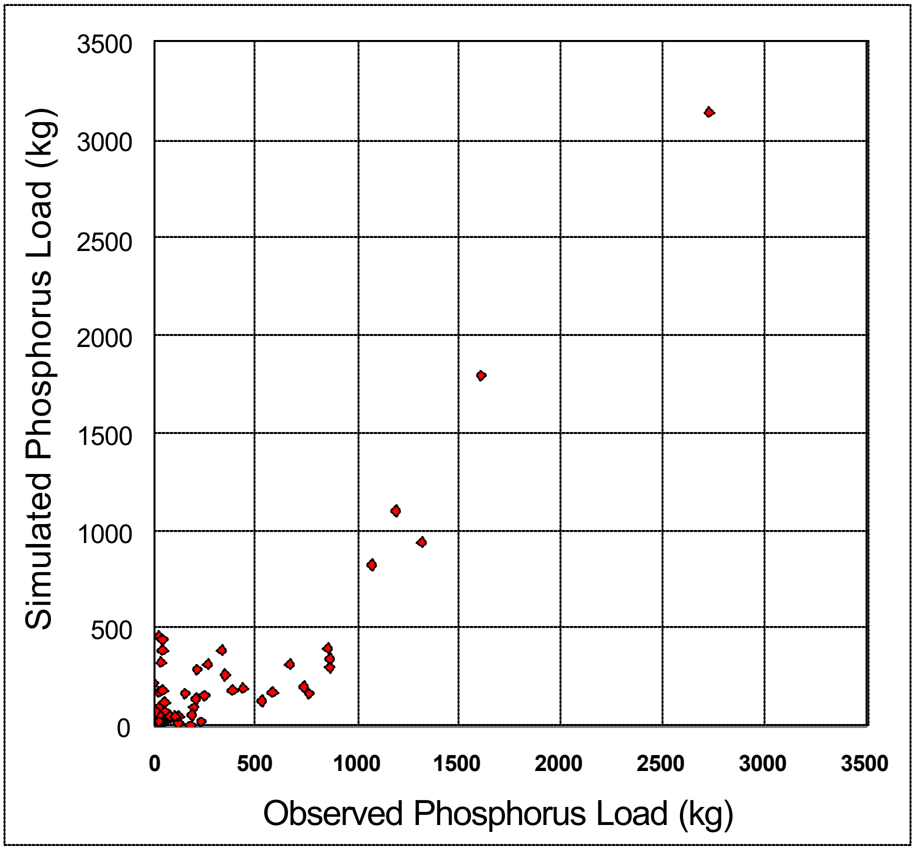


Figure 6-6. Observed and simulated phosphorus loads at Upper Bower Creek. 1990-94 calibration period.

CHAPTER 7. MODEL VALIDATION/ASSESSMENT

A model must be able to provide reliable estimates (validated) during the flow assessment phase before it can be further assessed to determine whether the model can be applied to provide reliable estimates and predictions of suspended sediment and phosphorus loads. This chapter describes how well the model was able to estimate flow, suspended sediment and phosphorus yields and loads. All validation comparisons utilized the modified and calibrated SWAT model. No adjustments of parameters were made to obtain a better fit between simulated and observed values in the model assessment phase. At any particular site, simulations for the model assessment phase were conducted during different time periods than during model calibration.

ASSESSMENT - HYDROLOGY

<u>Upper Bower Creek</u>: A total of 17 discharge events were computed by the USGS for the April 1, 1996 to June 30, 1997 model validation period at the USGS monitoring station on Upper Bower Creek. Stream volumes from these events are compared to simulated volumes in Figure 7-1. However, only 12 of these events were unaffected by ice according to the 1997 USGS Water Resource Data Book (USGS 1997). Ice-affected events are labeled in Figure 7-1. For the 12 observed and simulated flow events that were not affected by ice, R^2 was 0.95 (0.78 for all 17 events), and the NSCE was 0.94 (0.78 for all 17 events). These statistical measures, along with the relationship shown in Figure 7-1 indicates that there was an acceptable level of correspondence between simulated and observed events.

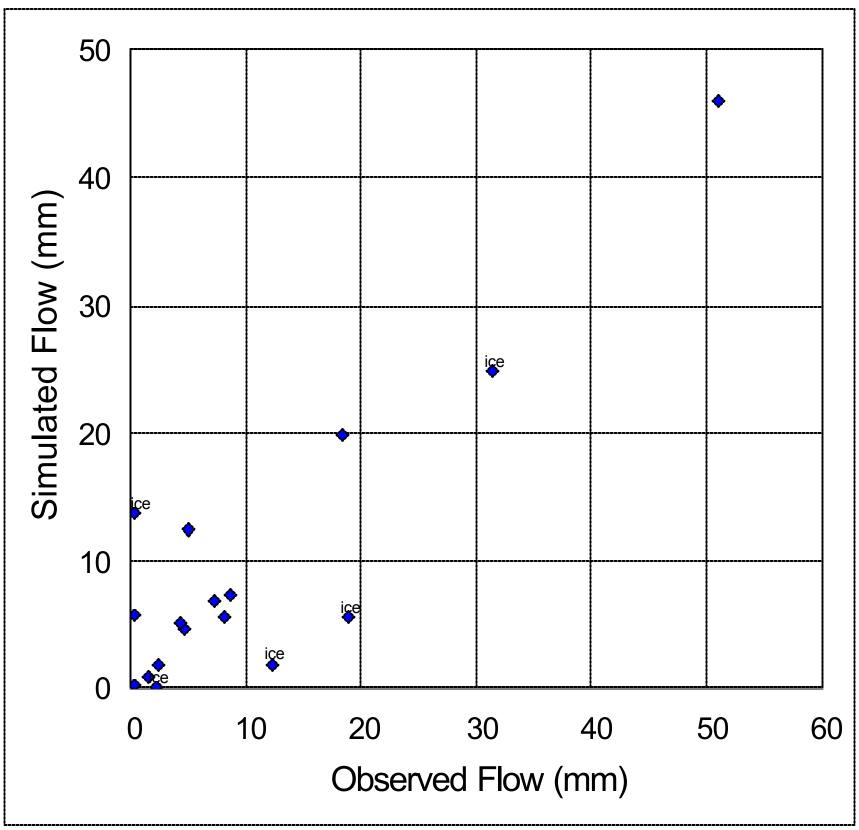


Figure 7-1. Observed and simulated flow events at Upper Bower Creek, 1996-97 validation period.

Simulated and observed daily stream flow at the USGS monitoring station on Upper Bower Creek during the April 1, 1996 to June 30, 1997 model assessment period are compared in Figure 7-2. Monitoring at this site was suspended April 1, 1995, and resumed April 1, 1996. Average precipitation from the USGS weather stations in the Bower Creek subwatershed is also shown in Figure 7-2. Total simulated water yield during this period was 316 mm compared to the observed total of 330 mm. In 1996, rainfall after the month of June had little effect on either the simulated or observed hydrographs, thereby indicating that soil moisture was being adequately tracked by the model during this period. As indicated in Figure 7-2, there is general agreement between the observed and simulated daily flows over a wide range of precipitation events. Notable exceptions include the ice-affected period from December 23 to March 29, 1997 (Bower Creek water discharge records, USGS 1997), and the last day of record: June 30, 1997. Only the first portion of the latter runoff event was measured before the daily measurements were discontinued, so comparisons between the simulated and observed flows for this event are not entirely appropriate.

Monthly simulated and observed stream flows from the validation period are shown in Figure 7-3, along with monthly precipitation from the USGS weather stations. The precipitation scale in Figure 7-3 is twice that of the scale for flow. The NSCE and coefficient of determination (R^2) were both 0.76 for monthly flows during this period, indicating a fairly good correspondence between observed and simulated flows. However, the simulated water yield of 41 mm in March 1997 was much less than the observed value of nearly 84 mm. As shown in Figure 7-3, total observed precipitation during March 1997 was 48 mm, <u>which is much less than the observed flow</u>. The model may have been unable to predict snow melt and associated runoff very well during this period. In addition, ice conditions may have contributed to this discrepancy because discharge estimates were ice-affected from December 23, 1996 to March 29, 1997 according to the USGS (USGS Water Resource Data Book, 1997 Water Year). Other factors such as above average groundwater recharge, frozen soil surface and thawing sub-soil may also contribute to the discrepancy between the simulated and measured flow in March 1997. Given that the measured flow was greater than the measured precipitation in this month, it is doubtful that the model could be expected to match the simulated and observed flow in this month.

The event, daily and monthly statistical measures, along with the relationships shown in Figures 7-1 to 7-3 indicate that there was an acceptable level of correspondence between simulated and observed events.

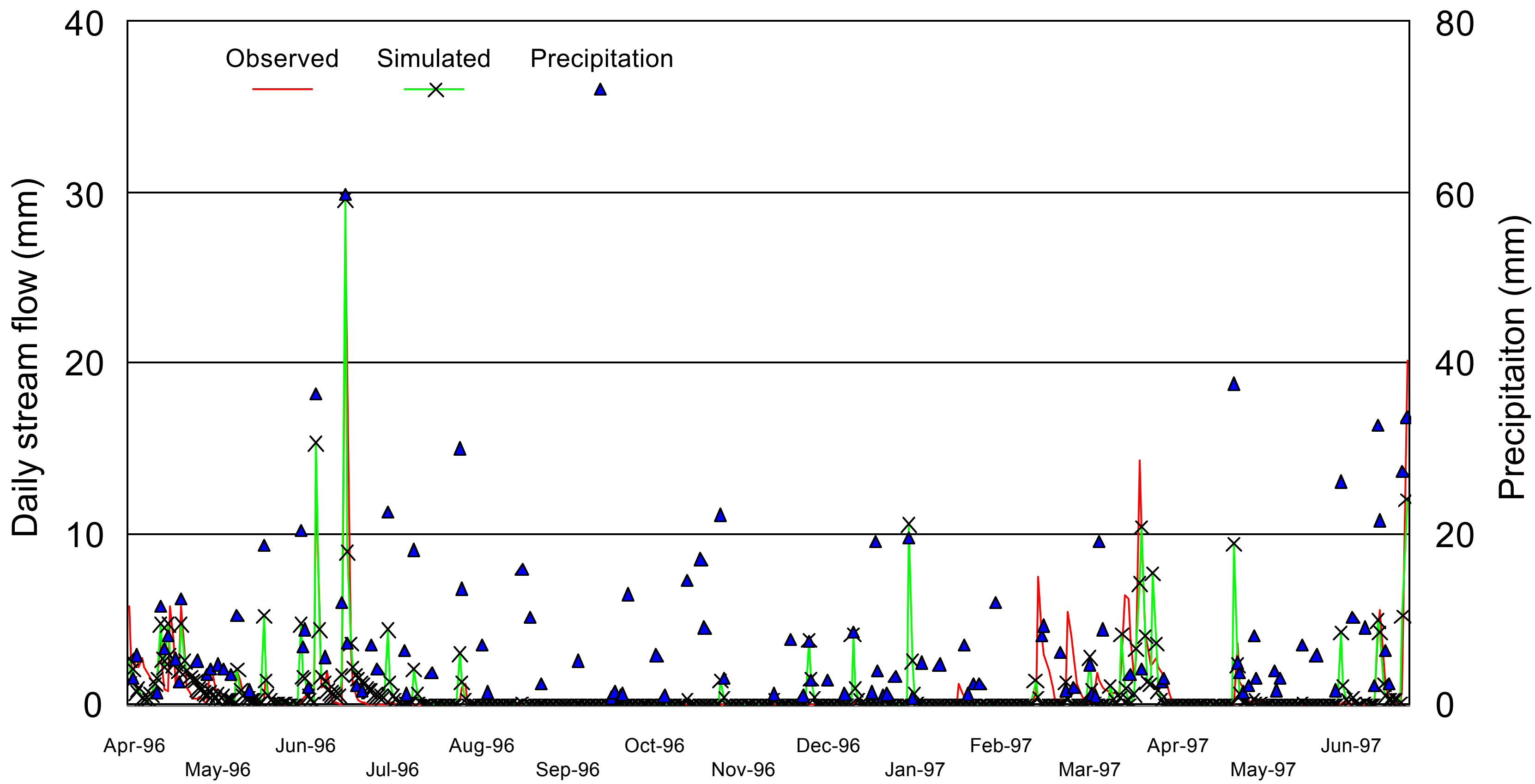


Figure 7-2. Observed and simulated daily stream flow - Upper Bower Creek. 1996-97 validation period. Precipitation from USGS weather stations is also shown.

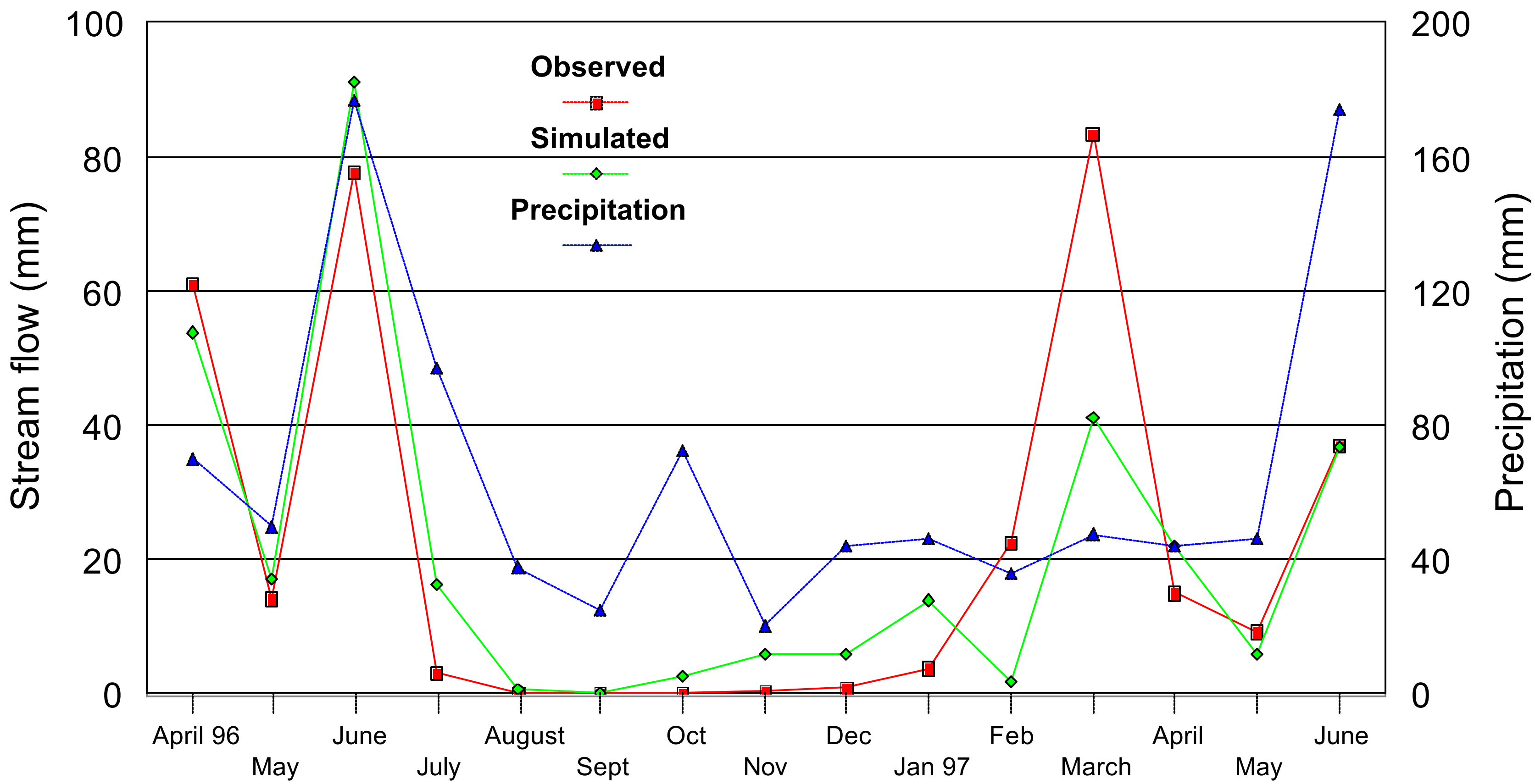


Figure 7-3. Observed and simulated monthly stream flow - Upper Bower Creek. 1996-97 validation period. Precipitation from USGS weather stations is also shown.

Duck Creek at County Highway FF: Average annual stream flow during the January 1, 1997 to June 30, 2001 model assessment period for simulated and observed runoff was 140 mm and 142 mm, respectively (counting partial year as ½). As shown in Table 7-1, with the exception of the year 2000, simulated average annual flows were relatively close to the observed flows.

Table 7-1. Simulated and observed average annual stream flow at Duck Creek (1997-June 2001).

	Observed Flow (mm)	Simulated Flow (mm)
1997	149	114
1,998	179	163
1,999	77	85
2,000	49	126
2001 (partial)	178	148

Simulated and observed monthly average stream flows for Duck Creek at CTH FF are compared in Figure 7-4 during the model assessment period. The NSCE was 0.36 and R-squared was 0.41 for the 54 months in this period, indicating only a fair fit between observed and simulated monthly flows. Observed flows greatly exceeded simulated values in March 1997, February 1998 and March 2001. Discharge measurements during these periods are typically affected by ice. Some of the departures may be due to errors in measured flows due to ice conditions in Duck Creek (i.e., stage height affected by ice), inability of the model to predict flow during these conditions or snow melt runoff periods, or other factors.

In addition, the simulated flow of 61 mm for July, 2000 greatly exceeded the observed flow of 7 mm. This large departure is due to a very localized extreme rainfall event of 105 mm which occurred on July 8, 2000. The localized nature of this event can be seen in a radar image of cumulative rainfall for that storm from the NOAA National Weather Service in Green Bay (available from author upon request). This image shows a narrow NW to SE oriented band of very high precipitation directly in line with the Green Bay NWS station. For comparison purposes, Appleton recorded 49 mm for the same day. If the July 8, 2000 event were excluded, the simulated annual water yield of 64 mm in 2000 would have been much closer to the observed flow of 49 mm. This example shows that a larger number of measured precipitation sites within the Duck Creek watershed would have greatly improved model results for the year 2000, and possibly other years. After the July 8, 2000 event was excluded, the NSCE and R-squared improved to 0.56 for the 54 months in this period, which indicates a better match with observed flows.

On a daily flow basis, a NSCE of 0.14 and R-squared of 0.30 were determined for this period, indicating a poor fit between observed and simulated daily flows (set of daily hydrographs available upon request). However, the NSCE and R-squared both improve to 0.48 when the July 8, 2000 event is excluded from the statistical comparison. Still, simulated stream flow did not track observed flows nearly as well as during the previous calibration period. Trends illustrated by the hydrograph suggest that the model seems to be overstating soil moisture and potential runoff during, or after extended dry periods. This finding was also indicated by the calibration data sets.

Base flow analysis of the 1997-2000 data set showed that on the second iteration of the computer program, base flow comprised about 29% of the USGS-measured stream flow in Duck Creek at CTH FF, which was essentially identical to that found for the simulated base flow during this same period.

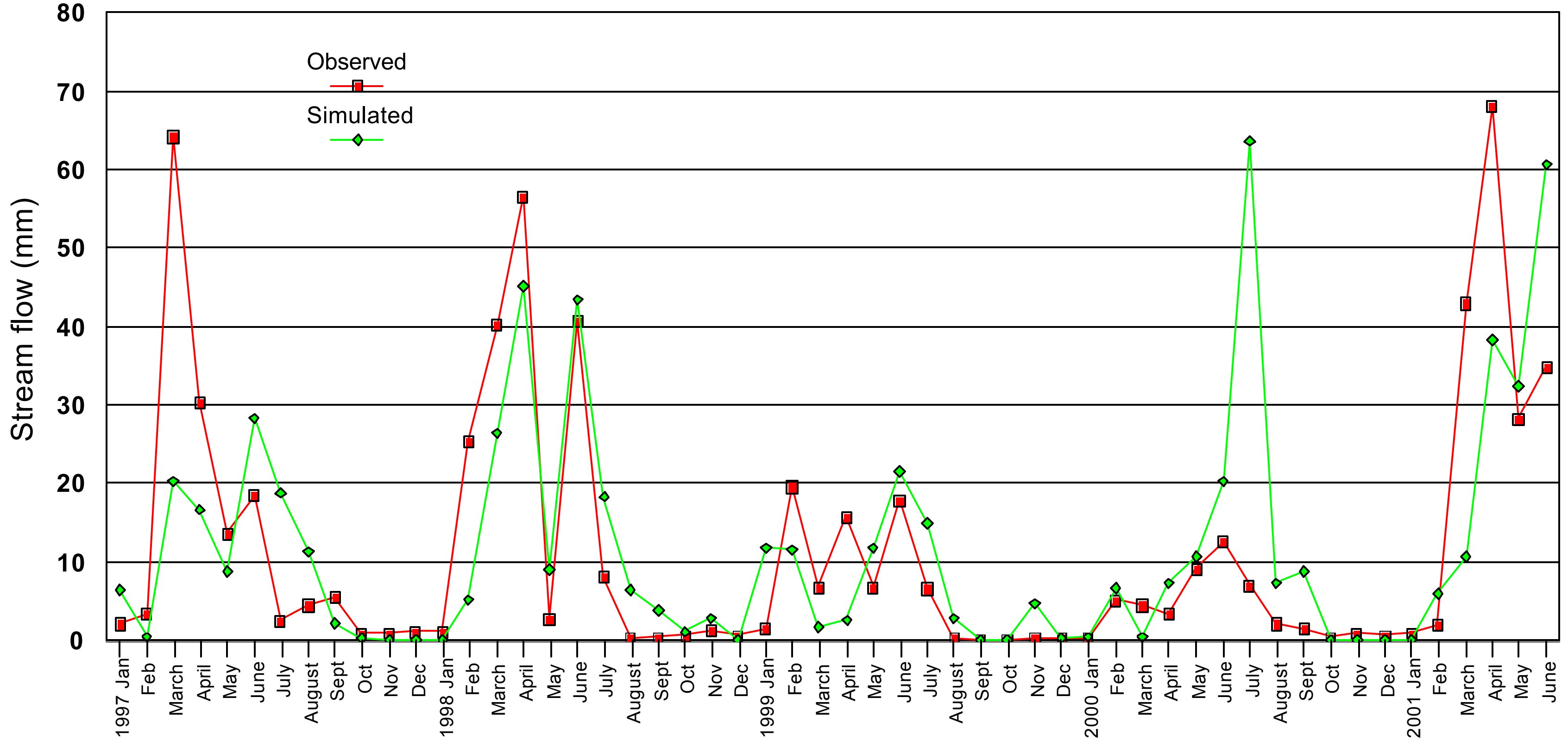


Figure 7-4. Observed and simulated monthly flow: Duck Creek at CTH FF (1997-2001 validation period).

Upper East River Watershed, East River at Midway Road: Simulated and observed daily stream flow at the USGS Station #04085109 are compared in Figure 7-5. As shown in Figure 1-1, this site is located at the outlet of LF01-1 on the Upper East River at Midway Road (122 km^2). The simulated hydrograph generally preserved the observed peaks and recessions during the 4/01/93 to 4/04/94 period (Figure 7-5). A NSCE of 0.74, and a R^2 coefficient of determination of 0.74 were computed for this period by comparing observed flow with simulated flow that was delayed by one day. This delay factor was added to account for phase differences between the model and actual events, but it was not used in Figure 7-5. Total simulated flow for the first period was 394 mm compared to the observed flow of 439 mm. Overall, simulated flow adequately tracked observed flows during the first period. The combined water yields from the first and second periods were 427 mm for the simulated flow, compared to an observed flow of 477 mm.

Stream flow volume during the second period (10/01/94 to 9/30/95) was substantially lower than the first period with an observed flow of 38 mm compared to the simulated flow of 32 mm (Figure 7-5). The peaks and recessions of the simulated flows generally occurred at the same time as the observed events. A NSCE of 0.30, and a R^2 coefficient of determination of 0.39 were computed for this period by comparing observed flow with simulated flow that was delayed by one day. At first glance, these statistics do not seem to indicate a robust fit between observed and simulated daily flows during the second period. However, peak daily flows are nearly an order of magnitude less than during the first period, so low correlation is not critical. In general, the calibrated model performed reasonably well during the second period, in the summer of 1995, which was extremely hot and fairly dry period. An average expected water yield for the watershed would be approximately 200 mm (Gebert et al. 1987). Therefore, this was an unusually dry period, and TSS loads should be expected to be quite low.

Simulated and observed monthly average stream flows for East River at Midway Road are compared in Figure 7-6 during the same model assessment period. NSCE and R-squared values of 0.92 were determined for the 24 monitored months, indicating a good fit between observed and simulated monthly flows.

Base flow analysis of flow data from the 1993-95 validation period (both periods combined) showed that on the second iteration of the computer program, base flow comprised approximately 22% of the USGS-measured stream flow in Upper East River at Midway Road. Simulated total base flow during this same period was determined to be about 27% using the same method. This period was fairly short because observed data were missing from 4/04/94 to 9/30/94. Therefore, baseflow determined with this method may not be indicative of long-term base flow. For example, the fraction of base flow indicated by direct model output was actually 17% for a 1989 to 2000 period (i.e., simulated recharge divided by total simulated flow).

Overall Model Assessment - Hydrology: As stated above, simulated flows tracked observed flows at all three validation sites on a daily, monthly and annual basis reasonably well. Therefore, model assessment could proceed for suspended sediment and phosphorus. Without this positive assessment for predicting flows, it would be unreasonable to expect that the model could be applied to predict suspended sediment or phosphorus loads.

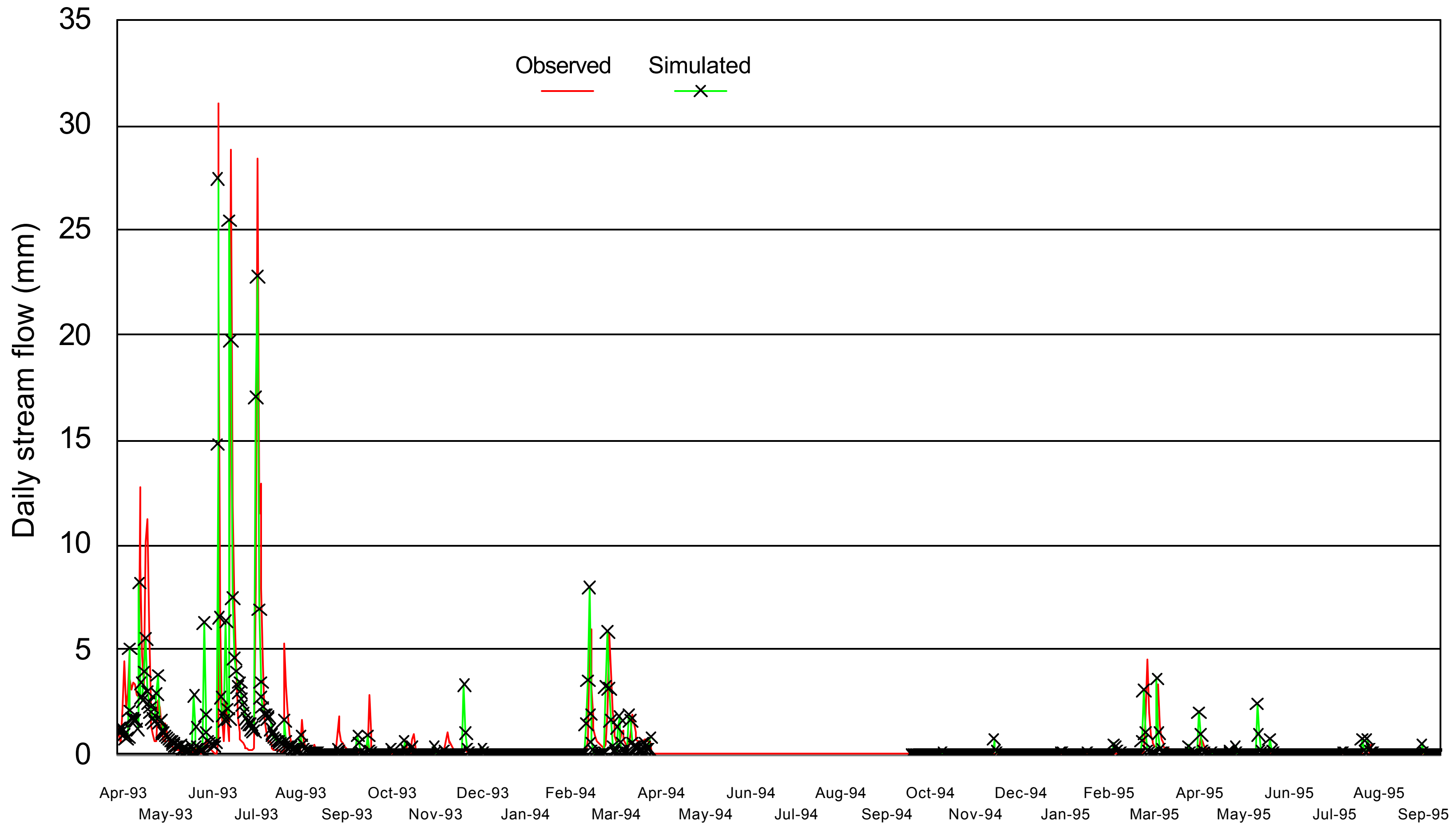


Figure 7-5. Observed and simulated daily stream flow - Upper East River at Midway Road. Model validation periods: April 1, 1993 to April 4, 1994 and Oct. 1, 1994 to Sept. 30, 1995.

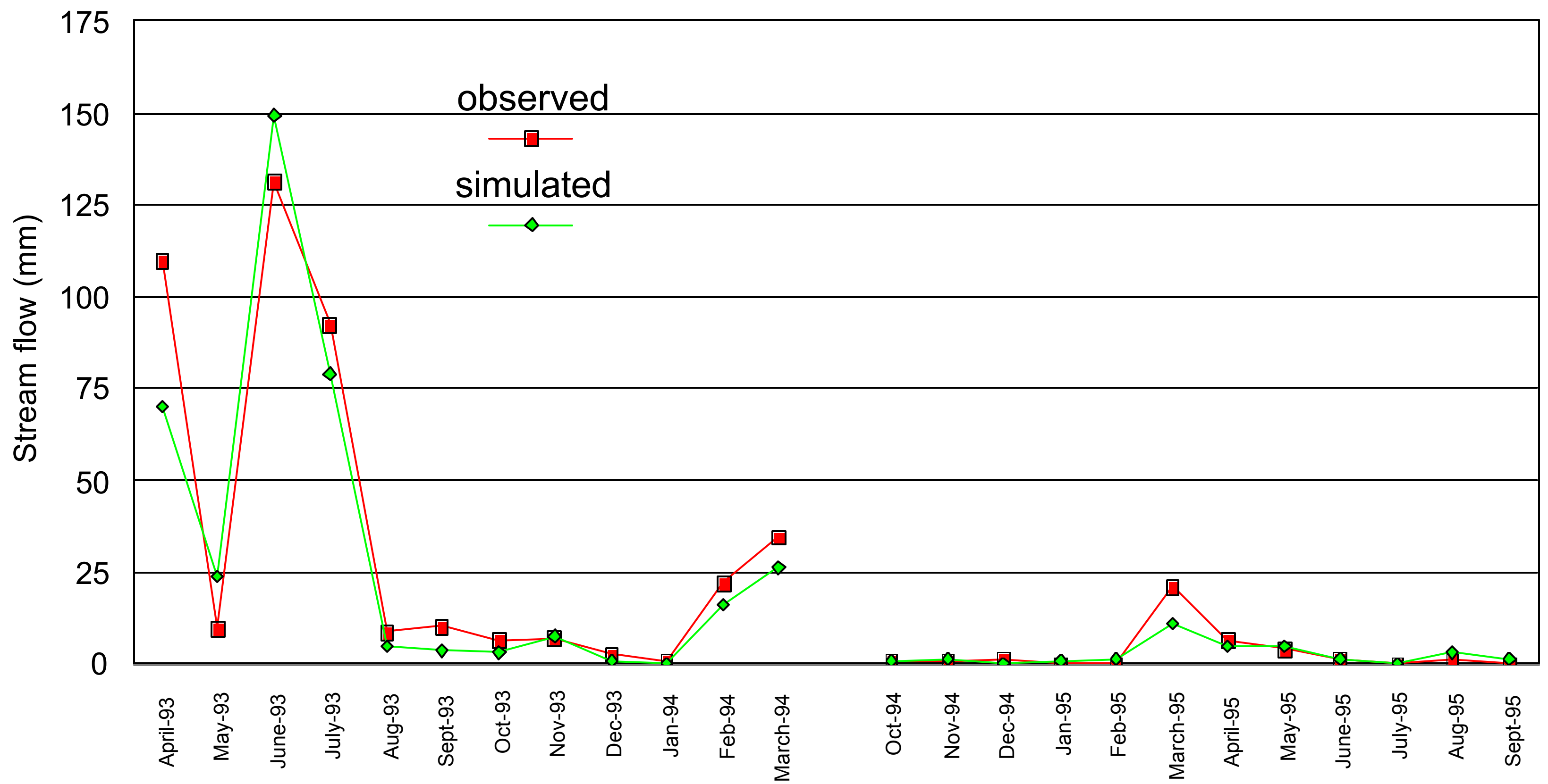


Figure 7-6. Observed and simulated monthly stream flow - Upper East River at Midway Road. Model validation periods: April 1993 to March 1994 and Oct. 1994 to Sept. 1995.

ASSESSMENT - SUSPENDED SEDIMENT AND PHOSPHORUS

The same parameter values selected during the model calibration phase were utilized during the validation/assessment phase.

Upper Bower Creek: Simulated and observed suspended sediment loads at the Bower Creek monitoring station site are compared in Figure 7-7. Only 12 of 17 USGS-measured events were unaffected by ice during the 1996-97 model validation period. The fit was acceptable, particularly since the largest events occurred in June, which is a transitional period during which crop growth and cover formation rapidly occur. The NSCE for 12 events was 0.89 (0.87 for all 17 events), and the coefficient of determination was 0.89 (0.87 for all 17 events). The total simulated suspended sediment load during the 12 measured events was 1,810 metric tons compared to the USGS-measured total of 1,730 metric tons (Walker et al. 2001). The total simulated suspended sediment load during all 17 events was 1,920 metric tons compared to the USGS-measured total of 1,820 metric tons. The simulated TSS load during the entire validation period was 2,400 metric tons compared to the USGS-estimated total of 2,300 metric tons (USGS 1996, USGS 1997). Although the data set was somewhat limited, these statistical measures and the relationship between simulated and observed loads (Figure 7-7) illustrate that the calibrated model was able to produce reasonable results.

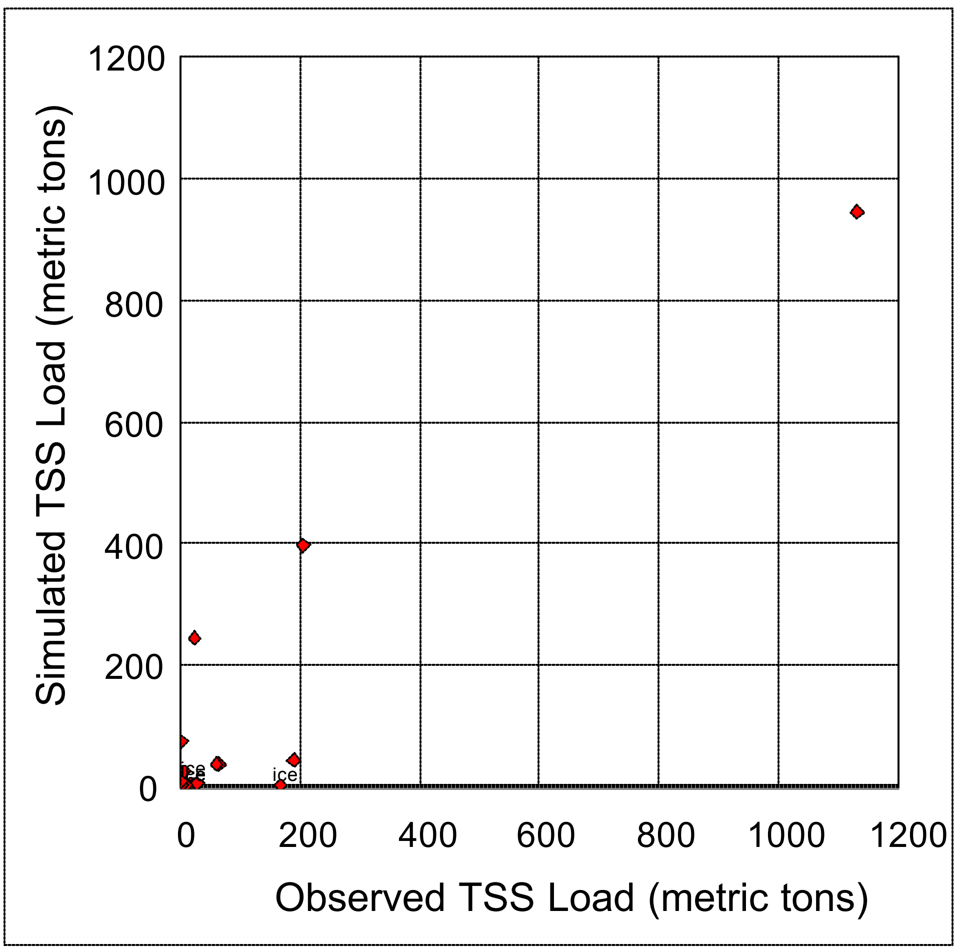


Figure 7-7. Observed and simulated suspended sediment load events at Upper Bower Creek, 1996-97 validation period.

Phosphorus: As shown in Figure 7-8, simulated phosphorus loads at Bower Creek correspond fairly well with the observed phosphorus loads measured by USGS during the validation period. The NSCE for 12 events was 0.88 (0.85 for all 17 events), and the coefficient of determination was 0.90 (0.87 for all 17 events).

The total simulated phosphorus load during the 12 measured events was 3,660 kg compared to the USGS-measured total of 4,220 kg (Walker et al. 2001). The simulated phosphorus load during the entire validation period was 6,790 kg compared to the USGS-estimated total of 7,470 kg (USGS 1996, USGS 1997). Although the data set was somewhat limited, these statistical measures, together with the relationship between simulated and observed loads (Figure 7-8) illustrate that the calibrated model was able to produce acceptable results.

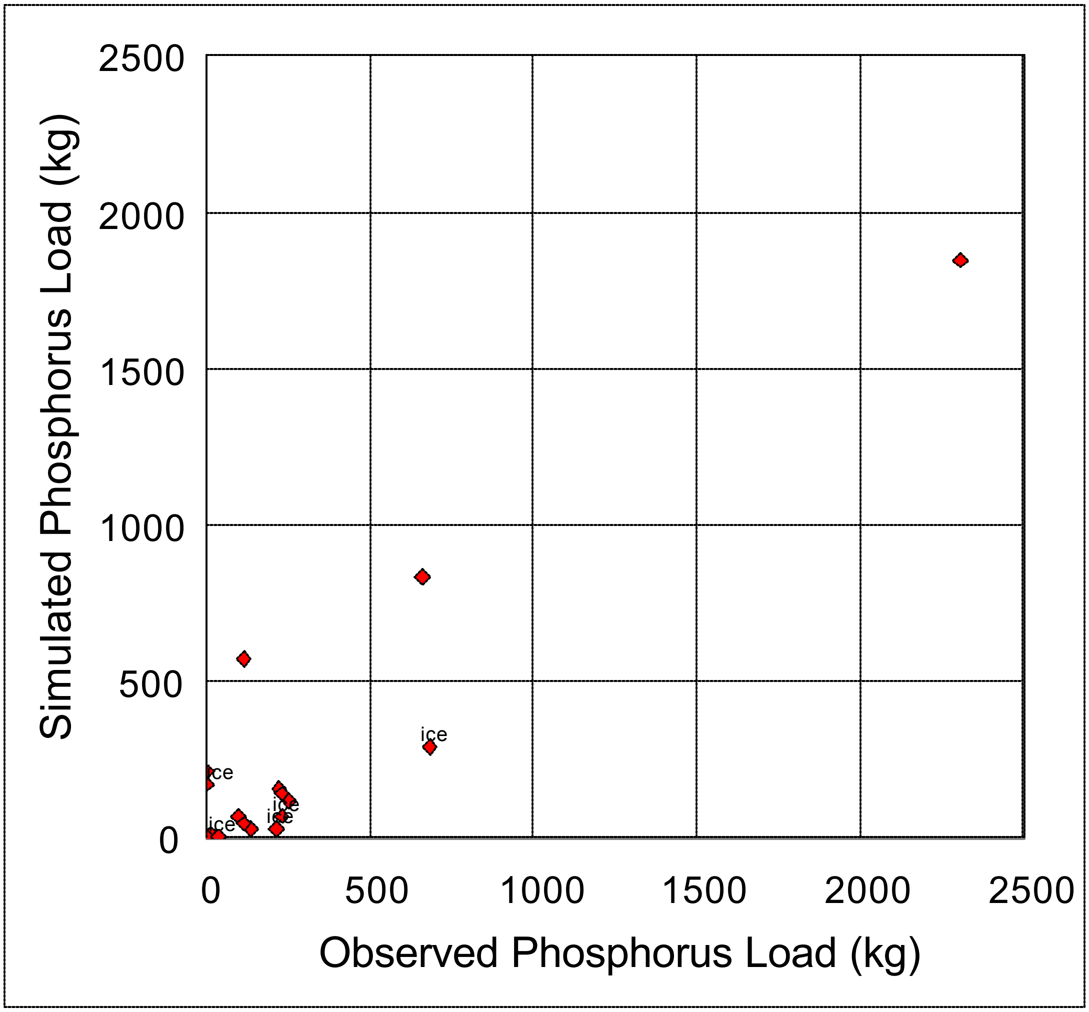


Figure 7-8. Observed and simulated phosphorus load events at Upper Bower Creek, 1996-97 validation period.

<u>Duck Creek at CTH FF</u>: A fairly large runoff event occurred between March 11 and 20, 1990. An estimate of the phosphorus load was calculated on the basis of USGS daily flows and 11 samples collected during this period by the USGS and analyzed for total phosphorus. Some of these samples were collected at the same time to compare automated pumped samples to manually collected equal-width-increment (EWI) samples. Phosphorus concentrations ranged from 0.17 to 0.78 mg/L during this period. The simulated phosphorus load of 4,900 kg from this event is relatively close to the estimated load of 7,500 kg.

A series of consecutive runoff events occurred between May 10 and 25, 1990. The phosphorus load was calculated on the basis of USGS daily flows and 23 samples collected near or during this period by the USGS and analyzed for total phosphorus. Phosphorus concentrations ranged from 0.17 to 1.2 mg/L during this period. Based on these data, phosphorus loads were estimated to be 670 kg (May 10-15), 620 (May 16-18) and 2,470 kg (May 19-25). These calculated loads compare to simulated loads of 1,060 kg (May 10-15), 310 (May 16-18) and 1,900 kg (May 19-25). The simulated load of 3,270 kg from all three events compares well to the estimated load of 3,760 kg.

A total phosphorus load was calculated for a period with two runoff events which occurred between June 13 and 21, 1990. The load was estimated on the basis of USGS daily flows and 11 samples collected near or during this period by the USGS. Phosphorus concentrations ranged from 0.41 to 1.1 mg/L during this period. Based on these data, the phosphorus load was estimated to be 3,200 kg compared to a simulated load of 4,000 kg. The highest recorded stream gage height of 21 feet occurred in the week following this event after 124.5 mm of rain fell at the Green Bay NWS station, but no samples were collected during this massive event.

A moderate event occurred between March 18 and 30, 1991. A rough estimate of suspended sediment load was calculated on the basis of five samples collected by the USGS and analyzed for suspended sediment, and daily discharge measurements. Sediment concentrations ranged from 32 to 43 mg/L during this period. The simulated load of 380 tons understates the estimated load of 540 tons, but is reasonably close. A phosphorus load was estimated on the basis of three daily samples collected March 18, 23 and 24, ranging from 0.17 to 0.27 mg/L. The simulated phosphorus load of 2,110 kg compares reasonably well with an estimated load of 3,150 kg. The USGS noted that the daily discharge measurements were ice-affected until March 24, so the aforementioned calculated loads may not be accurate.

A moderate to large event occurred between April 7 and 15, 1992. An estimated phosphorus load was calculated on the basis of daily discharge measurements and nine samples collected by the USGS and analyzed for total phosphorus. Phosphorus concentrations ranged from 0.08 to 0.17 mg/L during this period. The simulated phosphorus load of 1,880 kg is substantially higher than the calculated estimate of 1,010 kg. If this period is extended to include the next event, then the simulated phosphorus load of 3,700 kg compares to an estimated load of 2,800 kg during the entire April 7 to 21, 1995 period. However, the load estimated for the latter portion of this period was based on assuming that the concentration measured on April 13 was applied for all remaining days.

A moderate-sized event was sampled by University of Wisconsin-Green Bay (UWGB) students and analyzed for total suspended solids (TSS) in March, 1995 (UWGB 1995a). TSS concentrations ranged from 81 to 101 mg/L. The flow-weighted mean TSS concentration of the six samples they collected was 89 mg/L. Five daily USGS-reported flow volumes from March 20-24, 1995 were estimated to constitute the majority of the runoff for the entire storm event. Multiplying the total estimated runoff by the flow-weighted mean TSS concentration of 89 mg/L gave a total estimated TSS event load of 270 tons. The SWAT-simulated suspended sediment load for this event was also 270 tons. The flow-weighted mean total phosphorus concentration of the six samples the UWGB students collected was 0.48 mg/L, and ranged from 0.44 to 0.58 mg/L. The resulting estimated phosphorus load of 1,400 kg is somewhat lower than the simulated load of 1,700 kg.

A moderate runoff event occurred between April 17 and 26, 1995. Sediment and phosphorus loads were estimated on the basis of USGS daily flows and 15 samples collected during this period by the USGS and analyzed for suspended sediment and total phosphorus. Some of these samples were collected at the same time to compare automated pumped samples to manually collected equal-width-increment samples. Suspended sediment concentrations ranged from 12 to 126 mg/L during this period. The simulated sediment load of 280 kg from this event overstates the calculated load of 120 kg. Phosphorus concentrations ranged from 0.04 to 0.29 mg/L during this period. The simulated phosphorus load of 2,050 kg from this event is also much larger than the calculated load of 430 kg. Simulated flows closely matched observed flows during this period, so overstated modeled loads were caused by simulated concentrations that were too high.

A large runoff event occurred between June 17 and 24, 1996. A rough estimate of suspended sediment load was calculated on the basis of 7 samples collected June 17-21 by the USGS, and analyzed for suspended sediment. Sediment concentrations ranged from 62 to 184 mg/L during this period. The flow-weighted mean suspended sediment concentration for each day was multiplied by the daily flow volume to estimate daily sediment loads. Where a daily sample was not collected, the flow-weighted mean suspended sediment concentration for the entire event (88 mg/L) was multiplied by the daily flow volume to estimate the sediment load for that day. This method produced an estimated 770 tons of suspended sediment during the entire event, so the simulated load of 820 tons is fairly close.

With some exceptions, the model was able to provide reasonable predictions of sediment and phosphorus loads for the limited data that was available for comparison. The total simulated phosphorus load from all of the aforementioned events was 19,900 kg, which is close to the estimated load of 20,500 kg. The NSCE for these selected 9 events was 0.68, and the R^2 coefficient of determination was 0.74. The total simulated sediment load from four events was 1,750 tons, which is close to the estimated load of 1,700 kg. However, there is insufficient observed data to fully assess whether the model can provide reliable load predictions at this site for these two constituents at this time, which is especially true for sediment because the database for moderate to large events is quite limited at this location.

East River at Midway Road: A limited number of events were sampled by the USGS between 1993 and 1995, so loads were not computed by the USGS. However, for two large events USGS daily flow volumes and limited suspended sediment concentrations were used to calculate approximate sediment loads of 5,900 tons for June 9-11, 1993, and 4,800 tons for July 5-10, 1993. Simulated sediment loads of 4,700 tons and 2,700 tons, respectively, were lower than the estimated loads. Phosphorus loads of 2,500 kg for June 9-11, 1993, and 2,400 kg for July 5-10, 1993 were estimated for the same two events with available phosphorus concentration data. Simulated phosphorus loads of 7,500 kg and 4,900 kg, respectively, were much higher than the estimated loads. The simulated event mean phosphorus concentrations were 1.3 and 0.88 mg/L, respectively (total simulated load/simulated flow), which were also much higher than the highest concentration measured from the four samples collected during these two events (0.45 mg/L). The observed dissolved fraction of phosphorus was notably high during these two events, ranging from 60% to 75%. It is unclear why the observed concentrations were relatively low or why the samples contained high proportions of dissolved phosphorus during such large runoff events. Overall, the model did not provide accurate phosphorus loads for these two events, but was better with sediment loads.

East River at Monroe Street (USGS #04851378): The USGS operated a monitoring station from March 1985 to October 1986 near the outlet of the East River (367 km^2), where it crosses Monroe Street in Green Bay (Figure 1-1). Daily suspended sediment and total phosphorus, as determined by the USGS (Hughes 1993), were compared to simulated values. However, the seiche effect from nearby Green Bay greatly affects this site (Quinlan 1989, Hughes 1993), which prevented greater use of this data set. Data summarized in Table 7-2 were judged sufficiently unaffected by seiche effects, so they were selected for comparison to simulated values.

Table 7-2. Major observed and simulated suspended sediment and phosphorus load events at East River, Monroe Street monitoring station (USGS Station #040851378): 1985-1986.

Event Period	Suspended Sediment Load (metric tons)		Phosphorus Load (kg)	
	observed	simulated	observed	simulated
6/22/85 - 6/24/85	230	390	530	1,300
9/05/85 - 9/08/85	190	120	710	540
9/21/85 - 9/25/85	330	130	1,600	770
10/04/85 - 10/06/85	720	1,200	7,400	2,700
10/12/85 - 10/14/85	160	150	640	480
11/01/85 - 11/05/85	5,400	6,600	12,400	10,800
11/16/85 - 11/21/85[17]	1,000	800	5,500	2,400
7/17/86 - 7/19/86	110	320	370	1,000
7/25/86 - 7/28/86	120	550	590	1,700
8/17/86 - 8/19/86	100	70	540	260
9/10/86 - 9/13/86	580	1,560	5,000	4,500
9/20/86 - 9/30/86	1,000	640	10,400	2,900
TOTAL	**10,000**	**12,500**	**45,600**	**29,000**

Events during snow melt, or mixed rain and snow events, such as occurred March 10-16, 1985, were not considered in this comparison. Data from March 1 to April 10, 1986 were also not included because reported flows were only estimated by USGS from the daily discharge record of the Kewaunee River since the acoustic velocity meter was not operating during this period (Hughes 1993). Events with flow reversals which caused a net negative daily flow or load on any day during the event were not considered in this comparison. Event dates reported in Table 7-2 are for the measured daily loads; whereas, the simulated dates

[17] Prior to this event (i.e., Nov. 11-15, 1985), the observed suspended sediment load was approximately 409 tons, compared to a simulated value of only 18 tons. The measured suspended sediment load during this period may have been caused in part, by snow melt from a 11/09/85 snowfall of about 19 cm as recorded at the Green Bay NWS station, and also from rainfall of 16 mm on the 11/14/1985 at Brillion (no meaningful rain occurred at the NWS station during this period).

may be slightly different due to phase differences between the modeled and measured results results. Potential loads from urbanizing sources were not included in the simulated loads listed in Table 7-2.

The Nash-Sutcliffe coefficient of efficiency for the suspended sediment loads listed in Table 7-2 was 0.87, while a R-squared values of 0.95 (un-transformed) was determined with linear regression analysis. The simulated total suspended sediment for all of these events was 12,500 tons compared to the observed load of 9,900 tons. With some exceptions, simulated suspended sediment loads were reasonably close to observed loads, particularly in light of the seiche-induced complexity of the system.

The NSCE for the total phosphorus loads listed in Table 7-2 was 0.54, while R-squared was 0.67 (un-transformed). The simulated total phosphorus load for all of these events was 29,000 kg compared to an estimated observed load of 45,600 tons. Predicted phosphorus loads understated observed loads by 36%. Three large events (10/4-5/85; 11/16-21/85; 9/20-30/86) accounted for most of this difference. These discrepancies could be due to a number of factors including: (1) inherent problems with the model or manner in which it was applied (e.g., barnyard loads may be understated for this time period); (2) temporal delays in phosphorus delivery to the monitoring station caused by settling and resuspension of sediment or algae, particularly in the lower reaches; (3) inherent uncertainties involved in measured load estimates; and (4) the fact that none of precipitation stations were located within the watershed during the assessment period. The largest discrepancy in phosphorus loads occurred during the prolonged runoff event of 9/201986 to 9/30/1986. From 9/19/1986 to 9/30/1986, 116.7 mm of precipitation was recorded at Brillion, compared to 79.1 mm at the Green Bay NWS. This disparity in precipitation inputs to the model may have influenced the failure of the model to accurately predict the phosphorus load during this event, as well as understate the sediment load. However, it should also be noted that the model greatly overstated the sediment load during the previous event, while the simulated phosphorus load was essentially the same as the measured load.

Overall, the model was able to estimate phosphorus loads at this site with only a fair degree of accuracy during this time period. Although better load estimates would've been preferred, the author believes the model can be expected to provide acceptable phosphorus load estimates for this watershed. Reducing the influence of the seiche on both simulated and measured loads, or improved accounting of this influence, would help in assessing the ability of the model to predict loads at this site.

An example of the sensitivity of the model to the spatial variability of precipitation occurred on May 10, 1985 (not shown in Table 7-2). Green Bay reported 24 mm of rainfall on this date; whereas, Brillion reported no precipitation on this date, and only 1.3 mm for the following day. The simulated suspended sediment load was 770 tons for this event, based mostly on the 14 subwatersheds which were assigned Green Bay precipitation data. However, the measured daily loads for that period were minor; thereby, suggesting that most of the East River had much less rainfall than was measured in Green Bay. Temporarily substituting Brillion precipitation in place of the Green Bay data set produced simulated results which parallel the observed values for this period. Therefore, it should be recognized that simulated loads may not match observed loads simply because the precipitation inputs did not adequately represent actual precipitation over the watershed. However, the lack of precise precipitation measurements is probably not the only reason for all of the discrepancies.

<u>Overall Model Assessment - Sediment and Phosphorus</u>: The model was able to predict suspended sediment loads reasonably well at all four sites, although the second event at the Upper East River station was under-estimated by over 40%. Simulated loads were reasonably close to observed loads given an intentional emphasis on large events and total loads. Direct comparisons between individual events, statistical measures and graphical relationships support the conclusion that the model as applied in this project can predict suspended sediment loads at the subwatershed and watershed scales with an acceptable degree of accuracy.

With some exceptions, the model was able to predict phosphorus loads reasonably well at the Upper Bower Creek, Duck Creek and East River-Monroe Street sites. The model greatly overstated the phosphorus load for two large events at the Upper East River station, but the observed concentrations of dissolved and total phosphorus seemed somewhat unusual given the size of these two events. Simulated phosphorus loads seemed to be biased somewhat low at the East River-Monroe Street site, which may indicate the model could be improved with better inputs or by applying it differently. Inherent uncertainty could also be at fault. However, seiche effects and an inadequate number of precipitation measurement stations may have contributed to the discrepancies at this location. Still, simulated loads were reasonably close to observed loads given an intended emphasis on large events and total loads. Direct comparisons between individual events, statistical measures and graphical relationships support the conclusion that the model can be applied to predict phosphorus loads at the subwatershed and watershed scale with an acceptable degree of accuracy.

It must be emphasized that the model should not be expected to accurately predict loads during all events. There are simply too many factors that might contribute to higher or lower than expected concentrations of sediment or phosphorus.

CHAPTER 8. MODEL ASSESSMENT - LOADS TO GREEN BAY AND WATERSHED CONTRIBUTIONS

This chapter describes how well the model was able to estimate the amount of suspended sediment and total phosphorus delivered to Green Bay and selected watershed outlets.

<u>Phosphorus Load to Green Bay</u>: The simulated average annual non-point source phosphorus load to lower Green Bay from all of the watersheds in the subbasin is 140,400 kg (0.90 kg/ha) for the 1989-2000 climatic period (1992 Baseline Scenario). These figures include the Duck Creek watershed, which drains directly to Green Bay. Point sources along the lower Fox contributed an additional 131,000 kg/year from 1990 to 2000 (routed to mouth), making the combined contribution to Green Bay from point and non-point sources 271,000 kg/yr, or 246,000 kg/year if Duck Creek is not included.

Dale Robertson of the USGS has estimated an average annual phosphorus load of about 302,500 kg/yr at the Winnebago outlet for a 1989-1999 period (unpublished data produced for the Oneida Tribe, 2004). This load is similar to a measured load estimate for 1990 of 360 MT (WDNR 1993a). The same technique that was used to route loads from point sources and urbanizing sources along the Fox River (see Chapter 2, page 39) was applied to the 302,500 kg/yr phosphorus load at the Lake Winnebago outlet to derive an estimated contribution to Green Bay of 288,000 kg/yr for the 1989 to 2000 period.

With the addition of point sources and the estimated contribution from Lake Winnebago, the simulated 1989-2000 (1992 Baseline Scenario) average phosphorus load at the Fox River outlet to Green Bay is 534,000 kg/yr (LF05, Duck Creek is not included). If Duck Creek is included, the total load is 559,000 kg/yr. The former value falls within the 395,000 kg/yr to 719,000 kg/yr range of loads summarized by Klump et al. (1997) and within the 500,000 kg/yr to 605,000 kg/yr range estimated by Robertson and Saad (1996) for a 1980-90 period using a constituent transport model and available discharge and phosphorus data. Using this same methodology with updated data, Robertson has estimated an average annual phosphorus load of about 563,000 kg/yr at the Fox River outlet for the same 1989-2000 period (unpublished data produced for the Oneida Tribe, 2004). The simulated phosphorus load at the mouth is about 5% lower than this latest load estimate, whereas the simulated contribution below Lake Winnebago is about 11% lower than Robertson's estimate. On this basis, the modeled phosphorus loads at the river mouth appear to be consistent with loads estimated more directly by the USGS and others with observed Fox River discharge and phosphorus data.

<u>Sediment Load to Green Bay</u>: The simulated average annual non-point source suspended sediment load to lower Green Bay from all of the watersheds in the subbasin is 57,700 tons (0.37 t/ha) for the 1989-2000 climatic period (1992 Baseline Scenario). These figures include the Duck Creek watershed, which drains directly to Green Bay. The total load to Green Bay without Duck Creek is 49,400 t/yr. Point sources along the lower Fox contributed an estimated additional average of 3,000 t/yr from 1989 to 1995 (WDNR 2001). Based on this estimate and the simulated loads, the combined contribution to Green Bay from point and non-point sources for the 1989 to 2000 period was 60,700 t/yr.

Pierre-Gustin (1995) estimated a load at the Lake Winnebago outlet of 68,000 metric ton of TSS per year for a 1986-1990 period. Robertson of the USGS has estimated an average suspended sediment load of about 54,400 t/yr at Neenah, or Lake Winnebago outlet, for a 1989-1999 period (unpublished data produced for the Oneida Tribe, 2004). The same technique that was used to route loads from point sources and urbanizing sources along the Fox River (Chapter 2) was applied to the 54,400 t/yr sediment load at the Lake Winnebago outlet to derive an estimated contribution to Green Bay of 46,500 t/yr for the 1989 to 2000 period (trapping efficiencies were reduced slightly to account for greater proportion of fine particles compared to sediment from watershed outlets). An additional component was required to account for river growth of biotic solids, so an average river growth contribution of 20,000 t/yr TSS was added to the Fox River between Lake Winnebago and the outlet to Green Bay based on 1989 to 1995 data summarized by the WDNR (2001).

With the addition of point sources, the estimated contribution from Lake Winnebago, and the estimated biotic solids contribution, the simulated 1989-2000 (1992 Baseline Scenario) average suspended sediment load at the Fox River outlet to Green Bay is 119,000 t/yr (LF05, Duck Creek is not included). If Duck Creek is included, the total load is 127,000 t/yr. The former value is 17% lower than the 143,000 t/yr average suspended sediment load which was estimated by Robertson of the USGS for the load at the Fox River outlet (unpublished data produced for the Oneida Tribe, 2004). Robertson's estimate was for the 1989-2000 period and was based on including both the USGS and the Green Bay Metropolitan Sewerage District (GBMSD) data in the analysis and calibrating his constituent transport model with data from 1988 to 2003. The simulated load is also lower than the 151,000 t/yr suspended sediment load estimated by Robertson and Saad (1996) for a 1980-90 period using a similar method. However, the simulated load to Green Bay falls only slightly below the 123,000 t/yr load at the Fox River outlet which was estimated by Robertson of the USGS for a 1989-1999 period using only data collected by the GBMSD. This latter estimate illustrates that load estimates derived directly from river discharge and TSS or suspended sediment concentrations at the river mouth are not exact. Therefore, the simulated sediment load may not be that different than the actual load.

It is also possible that the biotic component may contribute a larger amount of solids to the mouth of the Fox River, or the East River contributes a significant amount of biotic solids that were not accounted for, or that the estimated load from Lake Winnebago is understated. Therefore, it is difficult to assess whether the simulated sediment loads are truly accurate or not. Although modeled suspended sediment loads at the river mouth were lower than those estimated more directly by the USGS with observed Fox River discharge and sediment data, the simulated loads to Green Bay were close enough to show that the model was able to provide reasonable estimates of suspended sediment contributions from the watersheds in the subbasin.

<u>Watershed Loads</u>: Simulated 1989-2000 average annual suspended sediment yields ranged from 0.22 t/ha in the Duck Creek watershed (LF05), up to 0.59 t/ha in the Plum, Kankapot Watershed (LF03). These yields are all greater than the median value of 0.11 t/ha (32.4 English tons/mi^2) that was reported by Corsi et al. (1997) for rural monitored areas of the Southeastern Wisconsin Till Plains Ecoregion. However, the simulated yields are well within the 0.015 t/ha (4.4 English tons/mi^2) to 6.0 t/ha (1,710 English tons/mi^2) range cited by Corsi et al. (1997).

Simulated 1989-2000 average annual phosphorus yields ranged from 0.66 kg/ha in the Duck Creek watershed (LF05), up to 1.32 kg/ha in the Plum, Kankapot Watershed (LF03). These yields are all greater than the median value of 0.50 kg/ha (283 lbs/mi^2) that was reported by Corsi et al. (1997) for rural monitored areas of the Southeastern Wisconsin Till Plains Ecoregion. However, the simulated yields are well within the 0.07 kg/ha (40.7 lbs/mi^2) to 3.1 kg/ha (1,800 lbs/mi^2) range cited by Corsi et al. (1997).

Dale Robertson of the USGS estimated an average phosphorus load of about 13,300 kg/yr at the USGS Duck Creek monitoring station at CTH FF for a 1989-2000 period by applying a constituent transport model to available discharge and concentration data (unpublished data produced for the Oneida Tribe, 2004). The simulated average annual load for this same period and site was 21,300 kg/yr, which is 60% higher than the load estimated by Robertson (R^2 of 0.61). The simulated average suspended sediment load of 4,500 t/yr is much higher than Robertson's estimated load of 1,900 t/yr (R^2 of 0.19). The precise cause of these discrepancies is not known, especially for sediment. As discussed in Chapter 7, the model seemed to be able to provide acceptable predictions of phosphorus and sediment loads for a number of events at this site, although the observed sediment data was fairly sparse for moderate and large events. The model may be overstating erosion for a number of reasons. Or it could be understating the effect of limited transport capacity in the main stem, tributaries and wetland complexes of the Duck Creek watershed. It is also possible that the sampling frequency, especially for sediment, has been too infrequent to adequately capture the distribution of loads from this stream, thereby affecting load estimations, particularly during highly erosive events. More accurate measured loads will be available for USGS water years 2004 to 2006 because the monitoring station at CTH FF was upgraded through the Lower Fox River Watershed Monitoring Program in October 2003. Loads from this station and four other sites will serve to better assess the accuracy of modeled results presented in this report.

Simulated Comparisons: The SWAT-modeled average annual suspended sediment load from Duck, Apple, Ashwaubenon and Dutchman Creeks combined, and delivered to Green Bay (LF05 and LF02), is 18,600 tons, or 0.27 t/ha at the watershed outlets (1989-2000; 1992 Baseline Scenario). This figure is much lower than the sediment load estimated by the Duck, Apple and Ashwaubenon Creeks Priority Watershed Project where the total load "delivered to streams" from all sources in these watersheds was 100,700 tons, or 1.46 t/ha per year (WDNR 1997; 110,016 English t/yr in Table 3-11). The latter estimate was based in part, on WINHUSLE (Baun 1995) modeling results for upland rural loads and SLAMM modeling for urban sources. However, it is uncertain whether these estimated loads were routed to the watershed outlets or just to the subwatershed outlets. The SWAT-simulated average annual total phosphorus load from Duck, Apple, Ashwaubenon and Dutchman Creeks combined and delivered to Green Bay, is 54,400 kg, or 0.80 kg/ha (1989-2000, 1992 Baseline Scenario). This figure is lower than the load estimated by the Duck, Apple and Ashwaubenon Creeks Priority Watershed Project where the total phosphorus load "delivered to streams" from these watersheds was 103,300 kg/yr (WDNR 1997; 227,805 English lbs/yr in Table 3-10). These comparisons show a marked discrepancy between loads estimated for this project and those estimated by the Priority Watershed Project. The precise reason for these differences is unknown; however, it is possible that the latter load estimates were not routed to watershed outlets which would account for some of the difference. Also, these comparisons are based on comparing loads from different simulation models, rather than to loads derived directly from observations.

Overall Model Assessment: Simulated phosphorus loads to Green Bay and to the Duck Creek monitoring station at CTH FF were reasonably close to loads estimated by Robertson of the USGS and Klump et al. (1997), and by Robertson of the USGS, respectively. The model was less able to match sediment loads to Green Bay, but still provided acceptable predictions. However, simulated loads to the Duck Creek station were two times greater than directly estimated loads. With this exception, simulated loads were reasonably close to observed loads. Overall, there is sufficient evidence to support the conclusion that the model can be applied to predict sediment and phosphorus loads to Green Bay from the subbasin with an acceptable degree of accuracy.

CHAPTER 9. SENSITIVITY ANALYSIS

Methods: A sensitivity analysis was conducted to assess the influence of selected key inputs on simulated total suspended sediment or solids (TSS), phosphorus, stream flow and stream recharge. To perform this analysis, the NRCS curve number (CN), manure depth fraction, soil available water capacity (AWC) and soil labile phosphorus concentrations were adjusted in the calibration subwatershed LF01-15 to determine the sensitivity of the SWAT model to changes in each of these parameters.

In addition, the SPCON and RS-5 parameters were varied in the East River Watershed (LF01) to determine how sensitive modeled output was to changes in the maximum concentration of entrained sediment and the organic phosphorus settling rate (QUAL2E submodel), respectively. The SPCON parameter affects the amount sediment that may settle in a stream reach that is used to route water through a watershed or subbasin/basin. Within SWAT, the RS-5 parameter is utilized in the QUAL2E in-stream water quality submodel, and it affects the settling rate of organic phosphorus in a stream reach, thereby affecting the transport of phosphorus through the system
.

As previously described in Chapter 2, the SWAT model was modified slightly to permit the input of a coefficient that was used to multiply the curve numbers in the SWAT management and subwatershed input files. A similar procedure was applied to available water capacity. Therefore, changes in model outputs are compared as the CN and AWC coefficients are varied, rather than directly adjusting the actual CN and AWC values used to calibrate the model.

Results: Model outputs derived with the calibration parameter set are compared to those derived with the adjusted values in Table 9-1 on a relative basis. Stream recharge was most sensitive to changes in the curve number. Sediment and phosphorus loads were also sensitive to the curve number, but much less so to available water capacity. Relatively large changes in AWC did not greatly affect any of the tested outputs. Qiu (1993) found that sediment yield was very sensitive to the curve number in her application of the SWRRBWQ model to the Upper Bower Creek subwatershed. A sensitivity analysis conducted by Baumgart (1998) on Duck Creek (LF05) found that SWAT-simulated TSS yields were roughly 2 to 3 times as sensitive to curve number and AWC than the values shown in Table 9-1.

Although altering the NRCS curve number had a substantial impact on stream recharge, TSS load, and phosphorus load, it did not greatly affect total stream flow. The rationale for this observation is that when the curve number is increased, less percolation of incoming precipitation is permitted which decreases groundwater inputs to stream flow. The reverse is also true. Therefore, altering the curve number in an attempt to calibrate SWAT to measured stream flows may result in an unwanted shift in the water budget to the stream. For example, increasing the curve number to obtain greater stream flow will come at the expense of decreasing groundwater inputs to the stream, which may or may not be warranted. In addition, altering the curve number has a much greater effect on stream recharge (also surface runoff) and TSS load than stream flow, and these effects may not always be desirable.

Saturated conductivity, also called permeability, was not tested because previous work by Baumgart (1998) had shown little effect on TSS or stream flow, although there was a large impact on lateral flow.

Table 9-1. Sensitivity of suspended sediment, phosphorus and hydrologic outputs to selected inputs of the SWAT model.

Input Parameter and Default Value	Input Value	Input Percent Change	Sediment Load	Total Phosphorus Load	Sol-P Load	Stream Flow	Stream Recharge
			Deviation from standard output (input = default)				
Curve Number (0.985)	***1.034***	**5%**	20.2%	17.7%	15.0%	4.6%	-63.9%
LF01-15 Bower Creek	***1.005***	**2%**	8.1%	7.0%	5.8%	1.3%	-27.6%
	0.965	**-2%**	-8.1%	-6.6%	-5.3%	-0.9%	28.3%
	0.936	**-5%**	-19.7%	-16.7%	-13.7%	-1.9%	68.9%
Available Water Capacity (0.97)	***1.455***	**50%**	-10.4%	-9.1%	-7.9%	-6.7%	-0.5%
LF01-15 Bower Creek	***1.164***	**20%**	-4.5%	-3.6%	-3.0%	-3.3%	-2.5%
	1.067	**10%**	-2.5%	-1.8%	-1.3%	-1.8%	-1.7%
	0.873	**-10%**	2.6%	2.2%	2.0%	2.2%	3.6%
	0.776	**-20%**	5.4%	4.7%	4.2%	5.0%	9.7%
	0.485	**-50%**	14.8%	13.6%	12.6%	18.2%	52.0%
Manure Depth Fraction (0.5)	***0.75***	**50%**	0.0%	4.3%	6.6%	0.0%	0.0%
LF01-15 Bower Creek	***0.6***	**20%**	0.0%	1.7%	2.6%	0.0%	0.0%
	0.4	**-20%**	0.0%	-1.7%	-2.6%	0.0%	0.0%
	0.3	**-40%**	0.0%	-3.5%	-5.3%	0.0%	0.0%
	0.1	**-80%**	0.0%	-7.0%	-10.6%	0.0%	0.0%
Soil Phosphorus (40 ppm)	***80***	**100.0%**	0.0%	45.2%	38.2%	0.0%	0.0%
LF01-15 Bower Creek	***60***	**50.0%**	0.0%	22.6%	19.1%	0.0%	0.0%
	48	**20.0%**	0.0%	9.0%	7.6%	0.0%	0.0%
	36	**-10.0%**	0.0%	-4.5%	-3.8%	0.0%	0.0%
	30	**-25.0%**	0.0%	-11.3%	-9.5%	0.0%	0.0%
	20	**-50.0%**	0.0%	-22.6%	-19.0%	0.0%	0.0%
SPCON (800 mg/L)	***1200***	**50%**	7.6%	0.0%	0.0%	0.0%	0.0%
LF01 East River	***1000***	**25%**	4.3%	0.0%	0.0%	0.0%	0.0%
	600	**-25%**	-6.1%	0.0%	0.0%	0.0%	0.0%
	400	**-50%**	-15.3%	0.0%	0.0%	0.0%	0.0%
	300	**-63%**	-22.1%	0.0%	0.0%	0.0%	0.0%
QUAL2E, RS-5 (0.15)	***0.45***	**200%**	0.0%	-24.9%	-1.3%	0.0%	0.0%
LF01 East River	***0.3***	**100%**	0.0%	-14.9%	-0.7%	0.0%	0.0%
	0.25	**40%**	0.0%	-10.6%	-0.5%	0.0%	0.0%
	0.1	**-33%**	0.0%	6.4%	0.3%	0.0%	0.0%
	0.05	**-67%**	0.0%	13.6%	0.6%	0.0%	0.0%
	0.01	**-93%**	0.0%	20.1%	0.8%	0.0%	0.0%

Varying the fraction of manure that is applied to the surface from 0.75 to 0.1 (50 to -80%) affected soluble phosphorus (6.6 to -10.6%) more than total phosphorus (4.3 to -7.0%). The total range in soluble phosphorus was 17.2%. Although the simulated relative impact was not overly large, the differential impact of either incorporating manure or applying it directly to the surface can be significant, particularly when reduced tillage practices are employed. This point will be made more clear in the discussion of alternative scenarios in Chapter 11.

Relative changes in soil labile phosphorus concentrations in the top 203 mm (8") produced a response in phosphorus loads that was about half as large as the change in soil phosphorus. For example, increasing the soil phosphorus concentration by 50% raised total phosphorus by 23% and soluble phosphorus by 19%; conversely, decreasing the soil phosphorus concentration by 50% had the same but opposite effect.

With regards to the sediment transport, the model was not particularly sensitive to SPCON. Reducing SPCON from 800 to 300 mg/L only decreased the sediment load by 22%, while increasing it to 1200 mg/L only raised it by 7.6%. Therefore, it does not appear that the model as applied in this project is particularly sensitive to this parameter. This finding also suggests that it was not unreasonable to lower SPCON from 800 mg/L to 300 mg/L in LF05 to account for the greater extent of wetland complexes in the Duck Creek watershed and expected relative reduction in sediment transport capacity.

The model was not particularly sensitive to RS-5 in the QUAL2E submodel, which is a parameter in the in-stream water quality model that affects phosphorus settling and related transport. Reducing RS-5 from 0.15 to 0.01 (-93%) only raised the total phosphorus load by 20%, while increasing it to 0.45 (+200%) only decreased it by 25%. Therefore, it does not appear that the model as applied in this project is overly sensitive to this parameter. This finding also suggests that it was not unreasonable to raise RS-5 from 0.15 to 0.25 for LF05 to account for reduced phosphorus transport related to greater wetland complexes in the Duck Creek watershed.

This analysis was not meant to be exhaustive. Sensitivity analyses of SWAT reported by Baumgart (1998), Lenhart et al. (2002), Tinureh (2004) and many others include other parameters and a more extensive analysis than described in this chapter.

Since the SWAT model requires numerous site-specific inputs which affect loads and hydrology outputs, results from this sensitivity analysis may not necessarily be the same if a different data set was modeled. Thus, as the same parameters are altered in a different data set, hydrology outputs and constituent yields may vary more or less than observed here.

CHAPTER 10. MODEL RESULTS - LOWER FOX RIVER SUBBASIN SUSPENDED SEDIMENT AND PHOSPHORUS LOADS

WATERSHED AND SUBWATERSHED CONTRIBUTIONS

Watershed Non-point Source Contributions: Simulated average annual suspended sediment and phosphorus non-point source loads from watersheds in the Lower Fox River Subbasin are summarized in Table 10-1 for 1992 and 2000 Baseline conditions. Data are summarized for 1977-2000 and 1989-2000 climatic periods. The latter period is included so that load estimates generated in this project can be better compared to loads estimated by Dr. Dale Robertson of the USGS with a constituent transport model, as well as other sources. Loads generated under the Baseline 1992 Scenario are probably the most appropriate for this purpose. Loads are routed to the watershed outlet (i.e., Fox River or Green Bay), and to Lower Green Bay. Point source loads are not included in these estimates, but for suspended sediment, they are assumed to be relatively small compared to non-point sources. In Table 10-1 and elsewhere, "Sed-P" represents both sediment-attached and organic phosphorus, "Sol-P" represents soluble phosphorus, "Total P" represents total phosphorus, and "TSS" represents suspended sediment, or total suspended solids (used interchangeably in this report).

The simulated average annual non-point source contribution to Green Bay from the subbasin was 57,700 tons (0.37 t/ha)[18] of suspended sediment and 140,000 kg (0.90 kg/ha) of phosphorus for a 1989-2000 climatic period (1992 Baseline Conditions). These figures include the Duck Creek watershed, which drains directly to Green Bay. Point source contributions were excluded. As shown in Table 10-1, substantial differences in suspended sediment and phosphorus yields among the watersheds in the subbasin were simulated by the model. For example, sediment yields from LF03 are about twice that from LF05, and also much higher than LF02, LF04 and LF06. Phosphorus yields from LF03 are two times greater than from LF05, and substantially higher than LF04. Water quality data collected by government agencies from some of these watersheds seems to support these simulated differences. Mean and median concentrations of TSS from samples collected by the WDNR (Northeast Region, unpublished watershed monitoring data 1997 to 2002) from LF03 watersheds such as Plum, Kankapot and Garners Creeks were all at least twice as high as those from Mud Creek, which is a subwatershed of LF04 (Mud Cr. TSS: mean = 31 mg/L, median = 18 mg/L, n=23). In addition, phosphorus concentrations from LF03 watersheds Plum and Kankapot Creeks were all at least four times higher than those from Mud Creek (Mud Cr. phosphorus: mean = 0.17 mg/L, median = 0.11 mg/L, n=23).

USGS water quality data for samples collected from Duck Creek at CTH FF (upper 3/4 of LF05) were compiled to calculate suspended sediment mean and median concentrations of 40 mg/L and 25 mg/L (n=175, up to July 2001) and total phosphorus mean and median concentrations of 0.26 mg/L and 0.21 mg/L, respectively. These Duck Creek concentrations are similar to those measured by the WDNR from Mud Creek, so the observed data strongly suggests that yields from LF05 are also much lower than LF03. Simulated results are consistent with this finding, thereby lending greater credibility to the relative rankings among the watersheds shown in Table 10-1. However, caution should still be used when interpreting these results, for relative differences may not be statistically significant.

[18] Subbasin area of 1,554 km^2 doesn't include all surface waters. Area is 1,581 km^2 with all surface water.

Table 10-1. Simulated average annual suspended sediment and phosphorus non-point source loads from watersheds in the Lower Fox River Subbasin.

	Area	Routed to Watershed Outlet				Routed to Lower Green Bay			
		TSS	Sed-P	Sol-P	Total P	TSS	Sed-P	Sol-P	Total P
	(sq. km)	(ton) (t/ha)	(kg) (kg/ha)	(kg) (kg/ha)	(kg) (kg/ha)	(ton) (t/ha)	(kg) (kg/ha)	(kg) (kg/ha)	(kg) (kg/ha)
1977-2000 Annual Average - Baseline 2000 Scenario									
LF01	**372.9**	14,500	14,600	20,200	34,900	14,500	14,600	20,200	34,900
East River		(0.39)	(0.39)	(0.54)	(0.94)	(0.39)	(0.39)	(0.54)	(0.94)
LF02	**291.0**	9,700	12,300	15,500	27,900	9,000	11,400	15,500	26,900
Apple, Dutchman, Ash.		(0.33)	(0.42)	(0.53)	(0.96)	(0.31)	(0.39)	(0.53)	(0.92)
LF03	**213.5**	12,000	14,500	14,100	28,600	10,900	13,100	14,100	27,200
Plum, Kankapot, Garners		(0.56)	(0.68)	(0.66)	(1.34)	(0.51)	(0.61)	(0.66)	(1.27)
LF04	**98.0**	3,800	3,800	3,600	7,400	3,500	3,400	3,600	7,000
Fox River, Mud Cr.		(0.39)	(0.39)	(0.37)	(0.76)	(0.36)	(0.35)	(0.37)	(0.71)
LF05	**389.2**	7,800	7,600	18,300	26,000	7,800	7,600	18,300	26,000
Duck Creek		(0.20)	(0.20)	(0.47)	(0.67)	(0.20)	(0.20)	(0.47)	(0.67)
LF06	**106.6**	4,000	3,900	4,600	8,600	3,600	3,500	4,600	8,100
LLBDM, Neenah Slough		(0.38)	(0.37)	(0.43)	(0.81)	(0.34)	(0.33)	(0.43)	(0.76)
LFM	**83.4**	3,600	3,300	3,200	6,500	3,400	3,100	3,200	6,300
L. Fox Main Channel		(0.43)	(0.40)	(0.38)	(0.78)	(0.41)	(0.37)	(0.38)	(0.76)
Lower Fox	**1554.6**	55,400	60,000	79,500	139,900	52,700	56,700	79,500	136,400
Subbasin		(0.36)	(0.39)	(0.51)	(0.90)	(0.34)	(0.36)	(0.51)	(0.88)
1989-2000 Annual Average - Baseline 2000 Scenario									
LF01	**372.9**	16,500	15,800	19,900	35,700	16,500	15,800	19,900	35,700
East River		(0.44)	(0.42)	(0.53)	(0.96)	(0.44)	(0.42)	(0.53)	(0.96)
LF02	**291.0**	10,400	12,800	14,600	27,400	9,600	11,800	14,600	26,400
Apple, Dutchman, Ash.		(0.36)	(0.44)	(0.50)	(0.94)	(0.33)	(0.41)	(0.50)	(0.91)
LF03	**213.5**	12,500	14,600	13,500	28,100	11,300	13,200	13,500	26,700
Plum, Kankapot, Garners		(0.59)	(0.68)	(0.63)	(1.32)	(0.53)	(0.62)	(0.63)	(1.25)
LF04	**98.0**	3,800	3,900	2,900	6,800	3,500	3,500	2,900	6,500
Fox River, Mud Cr.		(0.39)	(0.40)	(0.30)	(0.69)	(0.36)	(0.36)	(0.30)	(0.66)
LF05	**389.2**	8,600	8,300	17,900	26,200	8,600	8,300	17,900	26,200
Duck Creek		(0.22)	(0.21)	(0.46)	(0.67)	(0.22)	(0.21)	(0.46)	(0.67)
LF06	**106.6**	4,000	3,900	4,000	7,900	3,600	3,500	4,000	7,400
LLBDM, Neenah Slough		(0.38)	(0.37)	(0.38)	(0.74)	(0.34)	(0.33)	(0.38)	(0.69)
LFM	**83.4**	3,800	3,700	2,600	6,200	3,600	3,500	2,600	6,000
L. Fox Main Channel		(0.46)	(0.44)	(0.31)	(0.74)	(0.43)	(0.42)	(0.31)	(0.72)
Lower Fox	**1554.6**	59,600	63,000	75,400	138,300	56,700	59,600	75,400	134,900
Subbasin		(0.38)	(0.41)	(0.49)	(0.89)	(0.36)	(0.38)	(0.49)	(0.87)
1989-2000 Annual Average - Baseline 1992 Scenario									
LF01	**372.9**	16,600	15,800	21,300	37,100	16,600	15,800	21,300	37,100
East River		(0.45)	(0.42)	(0.57)	(0.99)	(0.45)	(0.42)	(0.57)	(0.99)
LF02	**291.0**	11,100	14,000	15,700	29,700	10,200	12,900	15,700	28,600
Apple, Dutchman, Ash.		(0.38)	(0.48)	(0.54)	(1.02)	(0.35)	(0.44)	(0.54)	(0.98)
LF03	**213.5**	12,700	14,500	13,700	28,200	11,500	13,100	13,700	26,800
Plum, Kankapot, Garners		(0.59)	(0.68)	(0.64)	(1.32)	(0.54)	(0.61)	(0.64)	(1.26)
LF04	**98.0**	4,000	4,200	3,400	7,600	3,600	3,800	3,400	7,200
Fox River, Mud Cr.		(0.41)	(0.43)	(0.35)	(0.78)	(0.37)	(0.39)	(0.35)	(0.73)
LF05	**389.2**	8,400	8,100	17,700	25,800	8,400	8,100	17,700	25,800
Duck Creek		(0.22)	(0.21)	(0.45)	(0.66)	(0.22)	(0.21)	(0.45)	(0.66)
LF06	**106.6**	4,100	4,200	4,500	8,700	3,600	3,700	4,500	8,200
LLBDM, Neenah Slough		(0.38)	(0.39)	(0.42)	(0.82)	(0.34)	(0.35)	(0.42)	(0.77)
LFM	**83.4**	4,000	4,000	2,900	6,900	3,800	3,700	2,900	6,700
L. Fox Main Channel		(0.48)	(0.48)	(0.35)	(0.83)	(0.46)	(0.44)	(0.35)	(0.80)
Lower Fox	**1554.6**	60,900	64,800	79,300	144,100	57,700	61,200	79,300	140,400
Subbasin		(0.39)	(0.42)	(0.51)	(0.93)	(0.37)	(0.39)	(0.51)	(0.90)

<u>Non-point Source Subwatershed Loads</u>: Average annual suspended sediment and phosphorus <u>non-point source loads, as routed to the subwatershed outlet</u>, are summarized by watershed in Table 10-2 for each major landuse type (Baseline 2000 Scenario,1977-2000 climate). The percent load and percent area associated with each major landuse are also listed, so load contributions can be compared on a relative basis. Areas listed in this table do not include the surface waters within the subbasin, which are composed almost entirely of Fox River waters on an areal basis.

Within the subbasin, agricultural sources contribute 74% of the simulated phosphorus load, and 65% of the sediment load at the subwatershed outlet level (<u>non-point sources only</u>). Urban and urbanizing sources contribute 20.8% of the simulated phosphorus load, and 31.7% of the sediment load. Loads from other simulated sources are relatively insignificant. On a percent area basis, urbanizing areas contributed disproportionately high sediment and phosphorus loads relative to other sources. For example, urbanizing areas comprised about 1.2% of the land area in the LF02 watershed, but contributed 13.1% of the simulated suspended sediment load, and 5.3% of the phosphorus load from all of the subwatersheds in this watershed. The estimated barnyard phosphorus load dropped from 13,400 kg/yr (8.4%) in 1992 to 8,400 kg/yr (5.4%) in 2000 (1992 source loads not shown). Simulated barnyard sediment loads are very low because the barnyard component of the model was calibrated for phosphorus, rather than both constituents because it was difficult to match BARNY-derived loads of sediment and phosphorus at the same time. Average annual sediment and phosphorus loads are also listed for each subwatershed in Tables 10-3 and 10-4, respectively, by major landuse type (2000 Baseline Scenario).

Average annual suspended sediment and phosphorus yields, <u>as routed to the subwatershed outlet</u>, are illustrated in Figures 10-1 and 10-2, respectively. These yields represent Baseline 2000 conditions. The relative contribution to lower Green Bay from some subwatersheds in the headwaters of large watersheds such as the East River and Duck Creek, are likely to be noticeably lower than indicated in Figures 10-1 and 10-2 and Tables 10-3 and 10-4 because of disproportionately greater settling of suspended sediment and phosphorus along longer transport paths. Therefore, these data should not be directly applied to ranking loads or yields to Green Bay because they represent loads and yields at the subwatershed outlet.

As shown in Figure 10-1, suspended sediment yields <u>at subwatershed outlets</u> are expected to be greatest in the Plum Creek Watershed (LF03), followed by much of the East River Watershed (LF01) and Apple Creek (LF02-1,2,4). The lowest yields are from portions of Duck Creek and subwatersheds LF01-12 and LF01-5, both of which have a substantial proportion of wetlands (LF05-15 is essentially all wetland).

The patterns shown in Figure 10-2 are similar to those in Figure 10-1. Phosphorus yields from subwatersheds are again expected to be greatest in the Plum Creek Watershed (LF03), followed by much of the East River Watershed (LF01), the upper portion of Duck Creek (LF05-13,14,16) and Apple Creek (LF02-1,2,3,4). The lowest yields are from the lower portions of Duck Creek and a number of urban subwatersheds.

Table 10-2. Lower Fox River Subbasin simulated sediment and phosphorus subwatershed loads. Summarized by landuse type and watershed. Baseline 2000 Scenario - routed to subwatershed outlet.

Suspended Sediment Average Annual Load in Metric Ton

Watershed	Ag	Barnyard	Urban	Grassland	Forest	Wetland	Barren	Golf	Urbanize	TOTAL
LF01 Load	12,268	32	2,343	11	199	10	537	20	1,303	16,700
% load	73.5%	0.2%	14.0%	0.1%	1.2%	0.1%	3.2%	0.1%	7.8%	
area	57.4%	0.9%	19.3%	2.2%	13.5%	3.9%	1.0%	0.8%	0.9%	372.9
LF02 Load	6,398	32	2,198	7	48	4	132	9	1,340	10,200
% load	62.7%	0.3%	21.6%	0.1%	0.5%	0.0%	1.3%	0.1%	13.1%	
area	59.2%	1.0%	26.3%	2.2%	7.8%	1.3%	0.4%	0.7%	1.2%	291.0
LF03 Load	9,871	28	1,869	4	83	5	283	13	536	12,700
% load	77.7%	0.2%	14.7%	0.0%	0.7%	0.0%	2.2%	0.1%	4.2%	
area	64.6%	1.1%	20.7%	1.3%	7.6%	2.1%	1.2%	0.7%	0.6%	213.5
LF04 Load	899	4	2,307	2	13	1	123	6	470	3,800
% load	23.7%	0.1%	60.7%	0.1%	0.3%	0.0%	3.2%	0.2%	12.4%	
area	19.4%	0.3%	68.8%	1.6%	5.7%	0.8%	1.3%	0.9%	1.2%	98.0
LF05 Load	9,675	34	1,379	8	81	30	332	27	1,380	12,900
% load	75.0%	0.3%	10.7%	0.1%	0.6%	0.2%	2.6%	0.2%	10.7%	
area	54.8%	0.9%	18.0%	1.9%	13.1%	8.6%	0.7%	1.1%	0.9%	389.2
LF06 Load	1,312	7	2,069	2	12	4	121	3	529	4,100
% load	32.0%	0.2%	50.5%	0.0%	0.3%	0.1%	2.9%	0.1%	12.9%	
area	30.9%	0.5%	54.4%	1.2%	6.4%	3.2%	1.5%	0.8%	1.2%	106.6
LFM Load	1,015	2	2,133	2	21	2	5	0	466	3,600
% load	28.2%	0.1%	59.3%	0.0%	0.6%	0.1%	0.1%	0.0%	13.0%	
area	17.7%	0.3%	66.7%	1.5%	9.7%	2.5%	0.1%	0.0%	1.4%	83.4
TOTAL	**41,400**	**140**	**14,300**	**40**	**460**	**60**	**1,500**	**80**	**6,000**	**64,100**
% TOTAL	**64.7%**	**0.2%**	**22.3%**	**0.1%**	**0.7%**	**0.1%**	**2.4%**	**0.1%**	**9.4%**	

Phosphorus Average Annual Load in Kilogram

Watershed	Ag	Barnyard	Urban	Grassland	Forest	Wetland	Barren	Golf	Urbanize	TOTAL
LF01 Load	29,316	2,711	4,327	447	456	162	881	307	1,466	40,100
% load	73.1%	6.8%	10.8%	1.1%	1.1%	0.4%	2.2%	0.8%	3.7%	
area	57.4%	0.9%	19.3%	2.2%	13.5%	3.9%	1.0%	0.8%	0.9%	372.9
LF02 Load	20,203	1,649	4,258	325	217	48	249	202	1,507	28,700
% load	70.4%	5.7%	14.8%	1.1%	0.8%	0.2%	0.9%	0.7%	5.3%	
area	59.2%	1.0%	26.3%	2.2%	7.8%	1.3%	0.4%	0.7%	1.2%	291.0
LF03 Load	23,337	1,642	2,967	180	210	66	552	189	603	29,700
% load	78.6%	5.5%	10.0%	0.6%	0.7%	0.2%	1.9%	0.6%	2.0%	
area	64.6%	1.1%	20.7%	1.3%	7.6%	2.1%	1.2%	0.7%	0.6%	213.5
LF04 Load	2,497	204	3,671	79	54	11	242	92	529	7,400
% load	33.7%	2.8%	49.6%	1.1%	0.7%	0.1%	3.3%	1.2%	7.1%	
area	19.4%	0.3%	68.8%	1.6%	5.7%	0.8%	1.3%	0.9%	1.2%	98.0
LF05 Load	24,784	1,561	3,972	376	425	406	545	377	1,552	34,000
% load	72.9%	4.6%	11.7%	1.1%	1.3%	1.2%	1.6%	1.1%	4.6%	
area	54.8%	0.9%	18.0%	1.9%	13.1%	8.6%	0.7%	1.1%	0.9%	389.2
LF06 Load	4,173	390	2,940	69	71	50	285	82	595	8,700
% load	48.0%	4.5%	33.8%	0.8%	0.8%	0.6%	3.3%	0.9%	6.8%	
area	30.9%	0.5%	54.4%	1.2%	6.4%	3.2%	1.5%	0.8%	1.2%	106.6
LFM Load	2,313	212	3,270	72	75	25	13	0	525	6,500
% load	35.6%	3.3%	50.3%	1.1%	1.2%	0.4%	0.2%	0.0%	8.1%	
area	17.7%	0.3%	66.7%	1.5%	9.7%	2.5%	0.1%	0.0%	1.4%	83.4
TOTAL	**106,600**	**8,400**	**25,400**	**1,550**	**1,510**	**770**	**2,800**	**1,250**	**6,800**	**155,000**
% TOTAL	**68.8%**	**5.4%**	**16.4%**	**1.0%**	**1.0%**	**0.5%**	**1.8%**	**0.8%**	**4.4%**	

Table 10-3. Lower Fox River Subbasin simulated sediment loads by subwatershed for each landuse type (metric ton/year). Baseline 2000 Scenario - as routed to subwatershed outlet.

Subwatershed	Ag	Barnyard	Urban	Grassland	Forest	Wetland	Barren	Golf	Urbanize	TOTAL	Yield (t/ha)
LF01-1	493	1	44	1	7	0	3	1	33	583	0.47
LF01-2	750	1	40	1	20	0	2	6	25	845	0.69
LF01-3	1,397	4	45	1	25	0	9	0	41	1,521	0.48
LF01-4	924	2	40	1	12	0	86	0	30	1,097	0.61
LF01-5	133	0	3	0	1	1	3	0	2	145	0.18
LF01-6	1,262	3	33	1	20	0	12	0	21	1,353	0.55
LF01-7	583	2	10	1	4	2	20	0	6	629	0.37
LF01-8	12	0	373	0	2	0	0	0	57	444	0.43
LF01-9	31	0	392	0	5	1	0	0	108	537	0.31
LF01-10	0	0	45	0	0	0	0	0	0	45	0.36
LF01-11	487	1	83	0	21	0	20	3	54	671	0.46
LF01-12	699	3	58	1	4	3	6	0	76	850	0.20
LF01-13	1,514	4	448	1	35	1	197	9	356	2,565	0.46
LF01-14	852	2	18	0	2	0	9	0	13	897	0.51
LF01-15	1,753	5	52	1	6	1	28	0	40	1,885	0.52
LF01-16	1,193	3	304	1	32	1	135	0	261	1,930	0.52
LF01-17	186	1	354	0	4	0	4	0	179	728	0.45
LF02-1	735	3	122	1	6	0	6	4	71	949	0.43
LF02-2	1,025	4	262	0	7	1	5	0	178	1,482	0.43
LF02-3	745	5	79	1	3	0	7	2	69	910	0.32
LF02-4	1,386	6	526	1	9	1	43	0	370	2,343	0.44
LF02-5	484	2	315	2	6	0	2	0	204	1,016	0.29
LF02-6	946	5	110	1	6	0	6	0	108	1,182	0.29
LF02-7	48	0	525	1	4	1	2	0	151	732	0.31
LF02-8	430	2	239	1	4	0	6	2	169	853	0.31
LF02-9	598	4	20	0	3	0	56	0	20	700	0.27
LF03-1	813	1	39	0	28	0	11	0	40	933	0.77
LF03-2	2,493	6	53	1	10	0	49	4	36	2,653	0.79
LF03-3	1,139	3	38	0	5	0	39	0	25	1,250	0.59
LF03-4	1,152	4	7	1	7	0	32	0	6	1,208	0.48
LF03-5	13	0	244	0	4	0	0	0	38	298	0.50
LF03-6	2,213	8	63	1	7	1	73	0	41	2,406	0.54
LF03-7	873	2	0	0	1	3	29	0	0	909	0.56
LF03-8	766	2	535	0	5	1	37	0	251	1,597	0.56
LF03-9	0	0	821	0	2	0	8	4	62	897	0.45
LF03-10	410	1	68	0	15	0	5	5	37	540	0.83
LF04-1	259	1	805	1	3	0	8	6	229	1,312	0.34
LF04-2	511	2	405	1	5	1	67	0	195	1,186	0.43
LF04-3	87	0	283	0	4	0	35	0	11	419	0.49
LF04-4	42	0	814	0	2	0	14	0	36	909	0.39
LF05-1	73	0	164	0	2	2	3	0	160	406	0.31
LF05-2	3	0	357	0	2	1	1	0	73	437	0.23
LF05-3	3	0	98	0	0	1	9	0	38	149	0.33
LF05-4	361	2	186	0	6	1	10	4	251	822	0.29
LF05-5	41	0	137	0	3	0	0	10	128	320	0.29
LF05-6	629	3	81	0	10	4	16	5	195	944	0.24
LF05-7	373	1	26	1	10	1	19	4	69	504	0.38
LF05-8	533	2	66	1	5	1	43	0	84	734	0.38
LF05-9	335	1	16	0	7	2	1	0	26	389	0.25
LF05-10	743	4	31	1	5	2	11	0	30	828	0.23
LF05-11	247	1	14	0	1	0	18	0	15	297	0.29
LF05-12	1,473	6	30	1	15	2	27	0	73	1,627	0.31
LF05-13	2,491	7	100	2	8	2	165	3	151	2,929	0.53
LF05-14	1,730	6	57	1	4	4	7	0	74	1,882	0.38
LF05-15	2	0	0	0	0	6	0	0	0	9	0.01
LF05-16	637	2	15	1	2	0	1	0	13	670	0.42
LF06-1	286	1	578	1	3	1	17	2	165	1,054	0.39
LF06-2	17	0	524	0	1	0	0	1	80	623	0.36
LF06-3	1,009	5	240	1	7	2	84	0	178	1,528	0.37
LF06-4	0	0	726	0	1	1	19	0	105	853	0.41
LFM1-1	467	1	138	0	2	0	2	0	62	672	0.72
LFM1-2	458	1	187	1	6	0	3	0	111	767	0.50
LFM1-3	73	0	341	0	2	0	0	0	63	479	0.47
LFM1-4	12	0	771	0	2	0	0	0	38	824	0.37
LFM1-5	6	0	697	0	8	2	0	0	192	904	0.34
TOTAL	**41,438**	**139**	**14,298**	**36**	**456**	**56**	**1,533**	**79**	**6,024**	**64,060**	**0.41**

Table 10-4. Lower Fox River Subbasin simulated phosphorus loads by subwatershed for each landuse type (kg/year). Baseline 2000 Scenario - as routed to subwatershed outlet.

Subwatershed	Ag	Barnyard	Urban	Grassland	Forest	Wetland	Barren	Golf	Urbanize	TOTAL	Yield (kg/ha)
LF01-1	1,102	104	89	44	15	5	7	16	37	1,418	1.15
LF01-2	1,402	96	81	48	35	1	4	75	28	1,770	1.44
LF01-3	3,451	294	97	38	41	3	18	0	46	3,988	1.25
LF01-4	1,858	152	88	18	22	5	111	0	34	2,290	1.28
LF01-5	471	37	8	7	7	24	7	0	3	562	0.68
LF01-6	2,675	212	72	37	55	2	22	0	24	3,098	1.26
LF01-7	1,505	151	18	27	17	31	42	0	7	1,798	1.07
LF01-8	21	0	588	1	6	0	0	0	64	680	0.67
LF01-9	122	15	701	3	14	8	1	13	121	998	0.57
LF01-10	0	0	55	0	0	0	0	0	0	55	0.44
LF01-11	998	59	158	8	42	2	28	52	61	1,409	0.97
LF01-12	3,029	289	154	71	31	54	20	11	86	3,747	0.87
LF01-13	3,368	305	891	26	58	9	311	140	400	5,508	1.00
LF01-14	2,131	165	33	22	10	3	20	0	14	2,399	1.36
LF01-15	4,207	569	104	49	26	7	59	0	45	5,067	1.41
LF01-16	2,486	219	583	42	63	8	221	0	294	3,917	1.05
LF01-17	489	43	608	5	15	1	9	0	201	1,371	0.85
LF02-1	2,001	199	172	24	16	3	12	82	80	2,589	1.17
LF02-2	2,834	261	518	12	31	9	10	0	200	3,875	1.13
LF02-3	2,625	258	136	26	12	4	18	66	77	3,222	1.14
LF02-4	3,942	220	953	33	35	10	93	0	416	5,701	1.07
LF02-5	1,598	105	732	79	32	5	3	0	230	2,784	0.81
LF02-6	3,371	235	224	36	31	2	14	1	121	4,035	0.98
LF02-7	171	17	1,024	35	22	8	4	0	170	1,452	0.62
LF02-8	1,506	160	461	59	21	4	12	53	190	2,466	0.89
LF02-9	2,155	193	39	20	18	4	83	0	22	2,534	0.96
LF03-1	1,380	73	78	9	59	0	14	0	45	1,658	1.36
LF03-2	5,283	338	108	25	30	1	87	63	41	5,977	1.78
LF03-3	2,801	208	62	14	15	2	84	0	28	3,214	1.51
LF03-4	3,083	262	13	37	24	3	74	0	6	3,502	1.39
LF03-5	19	0	397	0	7	0	0	0	43	467	0.79
LF03-6	6,029	471	93	47	24	9	141	0	47	6,861	1.55
LF03-7	2,164	154	0	28	6	41	51	0	0	2,444	1.52
LF03-8	1,900	100	951	17	15	6	77	0	283	3,349	1.18
LF03-9	0	0	1,138	0	4	0	16	78	70	1,307	0.65
LF03-10	677	35	127	2	25	3	7	48	41	965	1.48
LF04-1	827	80	1,443	24	20	3	18	92	257	2,763	0.71
LF04-2	1,407	120	631	46	23	8	157	0	219	2,611	0.95
LF04-3	162	2	391	6	5	0	41	0	12	619	0.72
LF04-4	100	1	1,206	5	6	0	26	0	41	1,386	0.60
LF05-1	218	4	331	1	12	31	10	0	180	787	0.61
LF05-2	6	0	850	3	14	11	1	1	83	969	0.52
LF05-3	6	0	190	0	4	10	19	0	43	272	0.59
LF05-4	987	42	589	3	37	13	20	74	282	2,048	0.72
LF05-5	78	1	381	10	12	3	0	136	144	766	0.69
LF05-6	1,575	105	395	17	48	54	27	65	220	2,507	0.64
LF05-7	530	30	130	21	38	11	20	43	78	900	0.68
LF05-8	1,306	82	192	23	26	9	64	0	94	1,797	0.93
LF05-9	900	48	50	15	34	25	2	0	29	1,103	0.72
LF05-10	2,598	158	64	46	39	30	26	0	34	2,995	0.85
LF05-11	854	58	31	7	7	4	26	0	16	1,002	0.96
LF05-12	3,517	221	155	43	79	32	46	0	82	4,176	0.79
LF05-13	5,670	371	407	87	39	25	269	57	170	7,095	1.29
LF05-14	4,774	316	175	70	24	53	15	0	83	5,509	1.12
LF05-15	7	0	0	0	2	87	0	0	0	97	0.13
LF05-16	1,758	124	33	30	9	6	1	0	14	1,976	1.24
LF06-1	896	97	829	24	17	10	38	58	186	2,156	0.79
LF06-2	58	1	754	1	4	1	1	24	90	935	0.55
LF06-3	3,218	292	362	44	44	32	195	0	200	4,386	1.06
LF06-4	1	0	995	0	7	7	50	0	119	1,178	0.57
LFM1-1	968	165	256	5	7	0	4	0	70	1,477	1.57
LFM1-2	1,127	38	311	45	20	2	7	0	125	1,676	1.10
LFM1-3	179	8	501	13	5	0	1	0	70	776	0.76
LFM1-4	23	0	1,176	9	10	0	1	0	43	1,261	0.57
LFM1-5	16	0	1,026	0	32	23	0	0	216	1,314	0.50
TOTAL	**106,621**	**8,368**	**25,404**	**1,547**	**1,509**	**768**	**2,768**	**1,249**	**6,777**	**155,012**	**1.00**

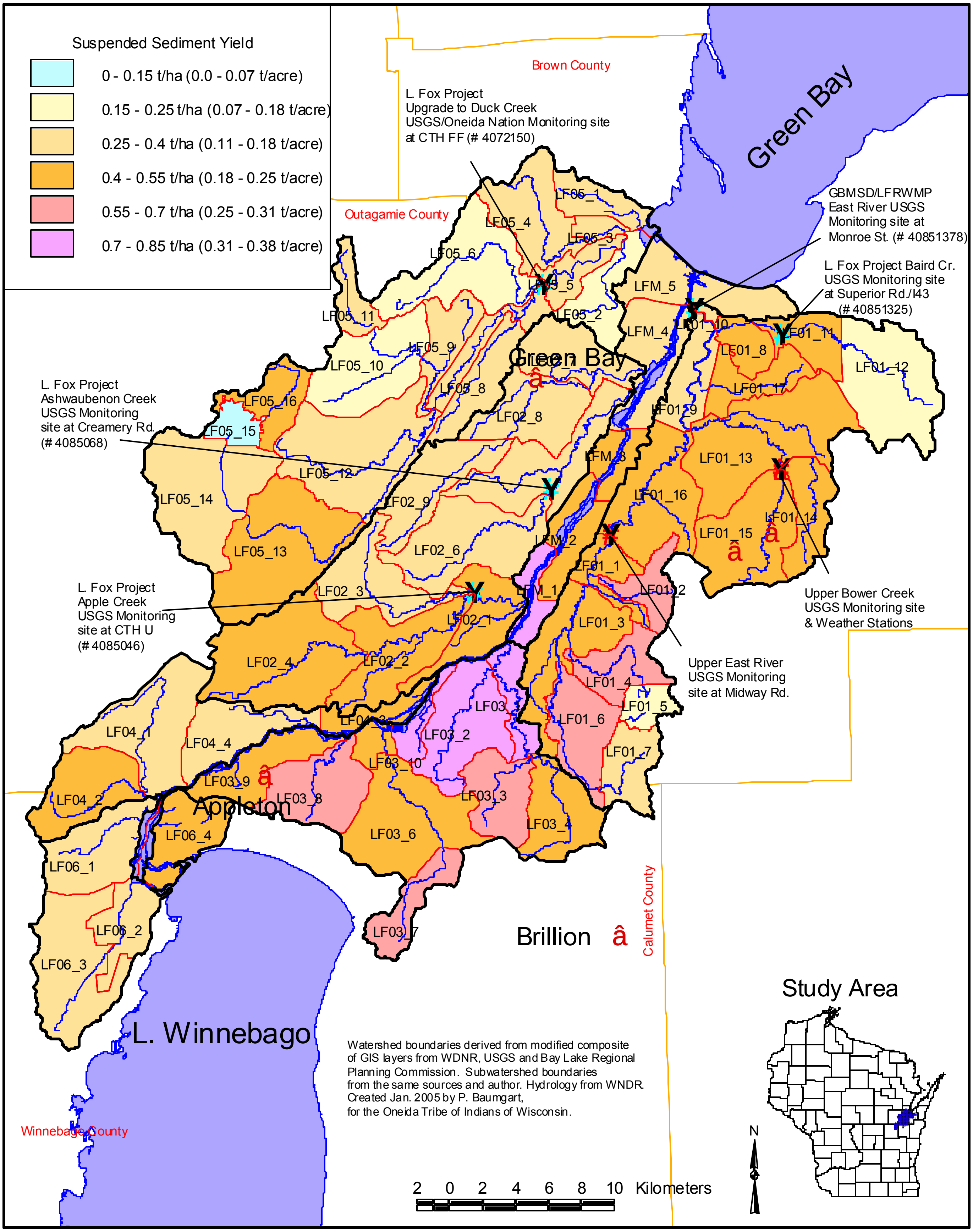


Figure 10-1. Simulated average annual non-point source sediment yields from subwatershed outlets in the Lower Fox River Subbasin (2000 Baseline Scenario; 1977-2000 climatic period).

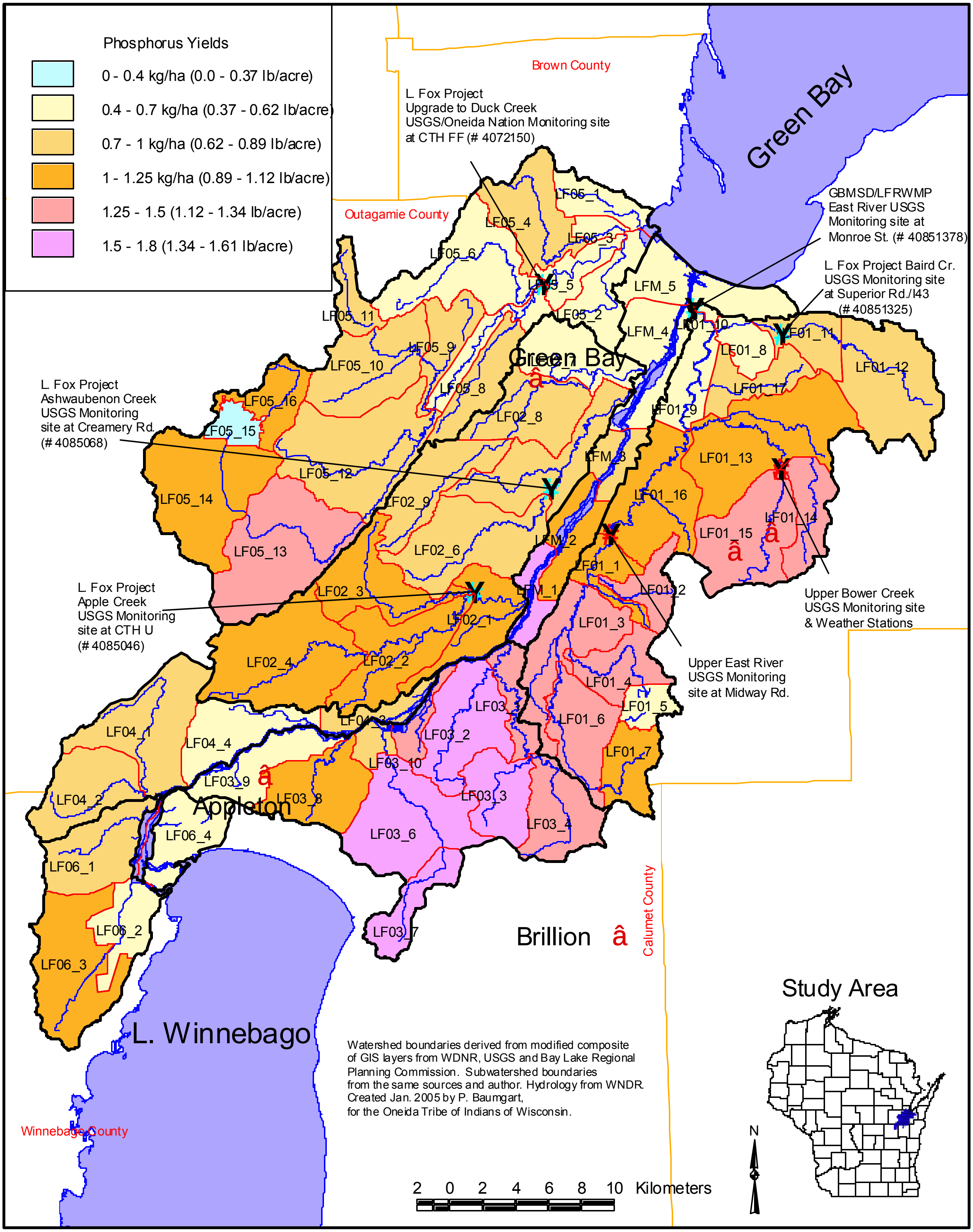


Figure 10-2. Simulated average annual non-point source phosphorus yields from subwatershed outlets in the Lower Fox River Subbasin (2000 Baseline Scenario; 1977-2000 climatic period).

LOAD ALLOCATIONS TO GREEN BAY

Load allocations to lower Green Bay of suspended sediment and phosphorus from the Lower Fox River Subbasin and Lake Winnebago are illustrated in Figures 10-3 and 10-4, respectively. These allocations are based on model simulations conducted in this project and 1989-2000 Lake Winnebago loads provided by Robertson of the USGS. Baseline 1992 and 2000 scenarios are both shown in these figures. The load that the Duck Creek Watershed contributes to lower Green Bay is included, although it doesn't flow into the Fox River. Each of these scenarios represent conditions estimated to be present near that time period. Estimated loads from Lake Winnebago are included in the top two pie charts to better illustrate the relative contributions to lower Green Bay from the lake and the Lower Fox River Subbasin. The bottom two pie charts illustrate the relative subbasin loads to Green Bay, without loads from Lake Winnebago.

As shown in Figure 10-3, Lake Winnebago was the largest single source of suspended sediment to lower Green Bay (46,500 t/yr), followed by agriculture (39,700 t/yr)[19], estimated river growth of biotic solids (20,000 t/yr) and urban and urbanizing sources (15,100 t/yr). Point sources and other sources of suspended sediment contributed 2.4% and 2.3%, respectively. These rankings are for the 1992 Baseline Scenario. The combined urban load is estimated to have increased to 18,300 t/yr by the year 2000, in response to a large increase in urban area. For this same reason, the agricultural load is estimated to have dropped to 36,300 t/yr by the year 2000 as agricultural areas were displaced by urban areas.

As shown in Figure 10-4, Lake Winnebago was the largest single source of phosphorus to lower Green Bay (288,000 kg/yr), followed by point sources (131,000 kg/yr), agriculture with barnyards (110,000 kg/yr) and urban and urbanizing sources (22,700 kg/yr). These rankings are for the 1992 Baseline Scenario. By the year 2000, point source contributions of phosphorus had decreased to 83,000 kg/yr, so agriculture was a larger relative contributor by that time. In contrast, the urban and urbanizing load is estimated to have increased to 28,500 kg/yr by 2000, in response to a substantial increase in urban area. The combined phosphorus load from agricultural sources dropped between 1992 and 2000 primarily because the amount of land under agricultural production decreased due to urbanization.

[19] The simulated barnyard sediment load was very low so it was lumped with "other sources". The load was low because the barnyard component of the model was calibrated for phosphorus, rather than both constituents because it was difficult to match BARNY-derived loads of sediment and phosphorus at the same time. Plus, phosphorus loads from barnyard sources were assumed to be the primary concern.

Figure 10-3. Estimated suspended sediment load allocation (t/yr) to Green Bay from Lake Winnebago and Lower Fox Subbasin. Baseline 1992 and 2000 conditions (1989-2000 climate).

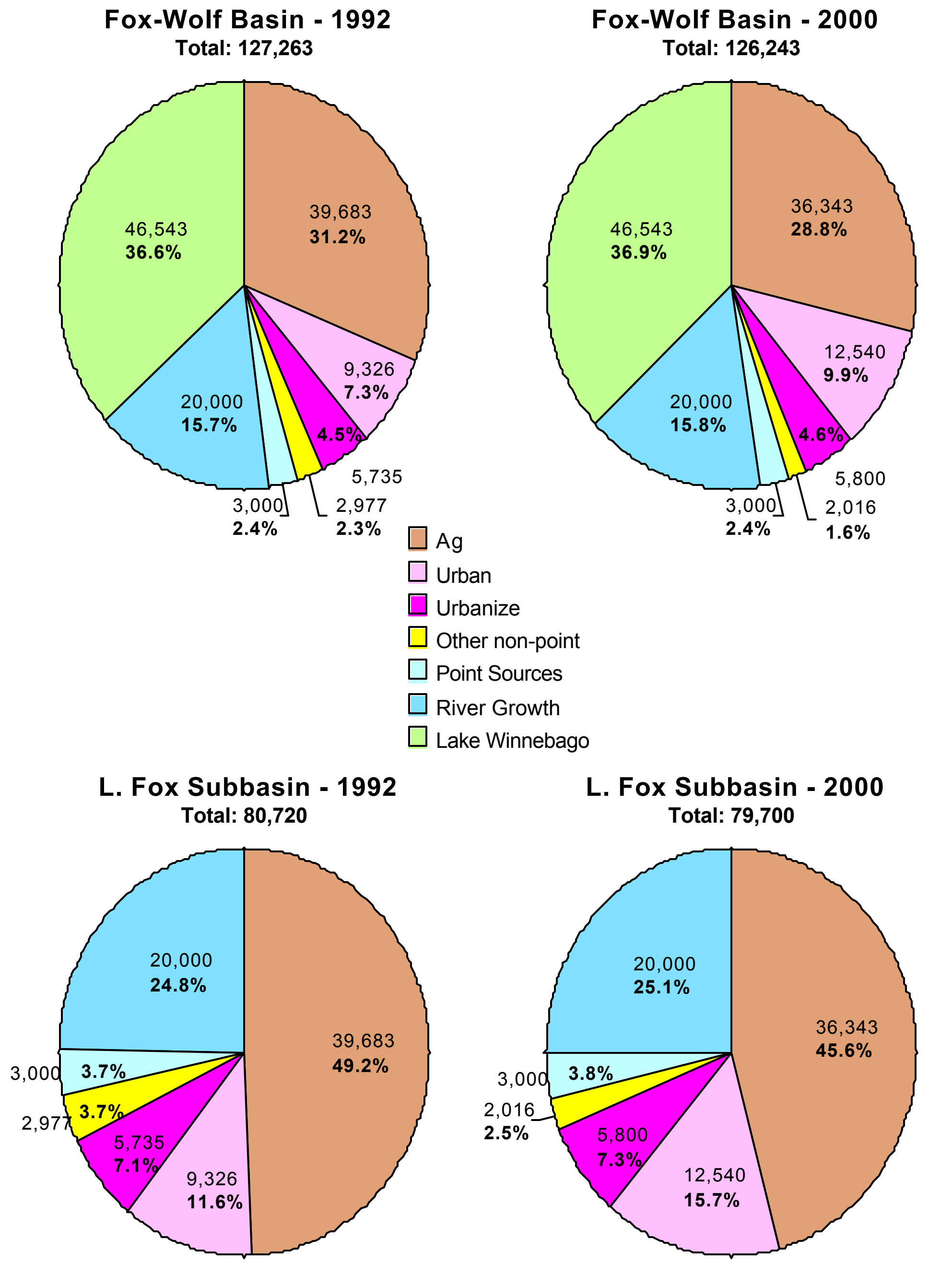
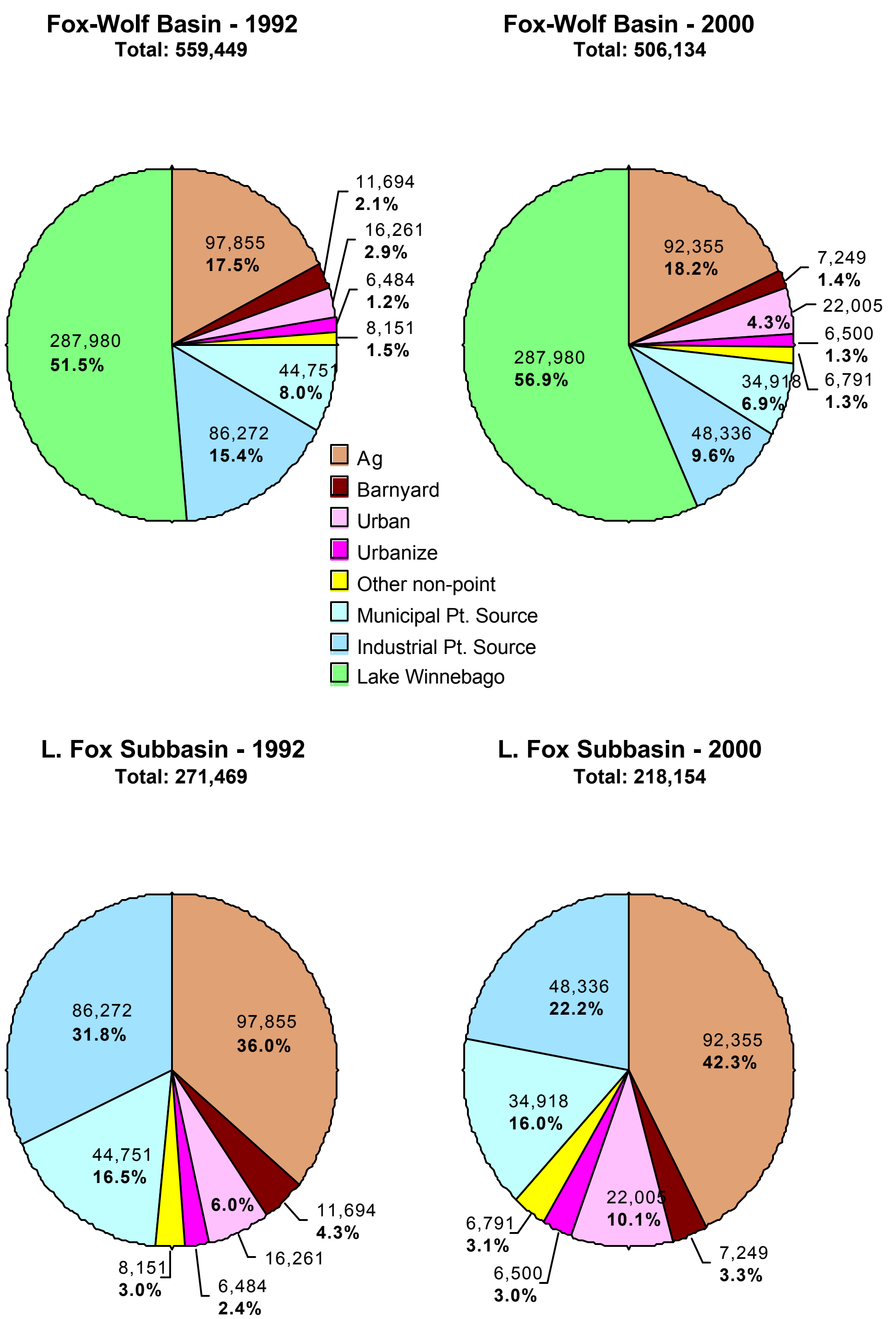


Figure 10-4. Estimated phosphorus load allocation (kg/yr) to Green Bay from Lake Winnebago and Lower Fox Subbasin. Baseline 1992 and 2000 conditions (1989-2000 climate).



CHAPTER 11. EVALUATION OF ALTERNATIVE SCENARIOS

This chapter describes how the SWAT model was applied to simulate the impact of implementing alternative policy changes, or scenarios, as well as the outcome of these scenarios compared to current practices. Ten major alternative scenarios were developed and simulated. Different options within each scenario were also explored. An additional scenario was developed to evaluate the effect of alternative VBS simulation methods. To determine the simulated impact of implementing alternative scenarios, the calibrated and validated model was applied to the entire subbasin for a 24 year climatic period (1977-2000). All scenarios were compared to the Baseline 2000 Scenario (current conditions) to evaluate the impact. In addition, results from the 1989 to 2000 latter period of these model simulations were also evaluated to determine if there were any differences.

The following section describes each alternative scenario and how the model was applied to simulate the impact on Lower Fox River Subbasin sediment and phosphorus load contributions to Green Bay.

DEVELOPMENT OF ALTERNATIVE SCENARIOS

The following alternative scenarios were developed, simulated, and then compared to modeled output from the Baseline 2000 Scenario (i.e., current conditions).

1. Entire subbasin forested.
Scenario 1: What if the entire subbasin was forested?

In this scenario, all landuses other than wetland, were changed to forest. All model inputs for the expanded forested areas were the same as the forest HRU, although spatially sensitive model inputs remained the same. This scenario is essentially a type of hindcast, for nearly the entire subbasin was forested prior to European settlement.

2. Nutrient Management - Soil-test phosphorus averages 25 ppm. 40 ppm. 50 ppm (Bray P1).
Scenario 2: What will be the estimated effect of a comprehensive nutrient management plan which requires that phosphorus inputs be limited to crop agronomic needs, assuming that soil-test phosphorus levels <u>remain at the following levels</u>:

(2a) Current 40 ppm average (Bray-P1)
(2b) Levels present in mid-1970's of about 25 ppm
(2c) Maximum level of 50 ppm permitted in NR151 and Aug. 2002 Wisc. Conservation Planning Technical Note WI-1 for NRCS 590 which doesn't require additional management (all fields at 50 ppm; e.g., farmers optimize operations and all fields average 50 ppm)

In this scenario, a number of management changes were instituted to ensure that soil phosphorus (P) levels did not increase over time due to net gains from fertilizer and manure applications. Therefore, no supplemental phosphorus in the form of starter fertilizer was added to the soybean crop. No supplemental phosphorus in the form of commercial fertilizer was added to the alfalfa crop. As was done for the 2000 Baseline Scenario, commercial fertilizer was applied at crop agronomic needs (i.e., harvest removal rates) for corn under the cash crop rotation. Under the dairy crop rotation, only the minimal recommended starter rate of 87 lbs/acre of 9-23-30 (Kelling et al. 1998) was initially applied to the corn crop. Even when soil nutrients are sufficient to meet crop needs, starter fertilizer still boosts yields slightly under the right conditions. Phosphorus levels in the dairy feed ration were reduced by 25% to further decrease the potential for increasing soil phosphorus levels. This reduction in dietary phosphorus is the same as that used in the low dietary phosphorus dairy feed ration scenario.

However, even these steps were not enough to stabilize simulated soil phosphorus levels, which continued to increase, although at a much lower rate than before. Therefore, manure phosphorus additions were further reduced another 25% (total reduction of 43.8% compared to the 2000 Baseline Scenario). Phosphorus levels in the soil still increased slightly, but it was found that applying 80% of the recommended starter to corn under the dairy rotation was just enough to stabilize soil levels for zero net total phosphorus gain. All of these management changes were therefore utilized to stabilize soil phosphorus levels under this series of scenarios (2a,b,c).

Justification of the soil test phosphorus concentration of 25 ppm (Bray P1) that is assumed under Scenario 2b is provided in Chapter 2 under the soil phosphorus section. As previously described, this level was present in the recent past, so it should be technically achievable with changes in management and enough time to reduce the excess phosphorus currently present in the soil.

3. Vegetative Buffer strip (VBS) implementation:
Scenario 3: What if buffers strips are comprehensively implemented throughout the subbasin?

In this scenario, it was assumed that VBS's were installed at the following levels:

(3a) 50% of streams as delineated in WDNR 1:24k hydro layer
(3b) 100% of streams as delineated in WDNR 1:24k hydro layer
(3c) 100% of all above streams plus road ditches (road ditch network scenario could also serve as a close approximation for including non-delineated streams/ditches that could be buffered).

Three additional options were modeled in this project to evaluate the effect of alternative VBS simulation methods on potential reductions of phosphorus and sediment. These options are included as Alternative Scenario 11 to distinguish the results from the default method that was applied for all other scenarios, because different methods were utilized to compute reductions from installing VBS's compared to the default method. All VBS simulation methods are described in Chapter 2 under the riparian buffer strip section.

4. Conservation tillage and manure incorporation - current levels versus reduced tillage intensity and greater levels of manure incorporation.
Scenario 4: What if conservation tillage use increased dramatically? What if all manure applied to fields is incorporated immediately?

Under this series of scenarios, the area of land dedicated to conservation tillage (i.e., reduced tillage practices) increases to the proportions listed below, compared to the Baseline 2000 Scenario. In addition, the effect of incorporating all manure that is applied to fields is included as an option under the baseline scenario and each conservation tillage scenario. This option was added because reduced tillage can increase the concentration of phosphorus in the uppermost soil layer compared to conventional tillage, particularly when manure is applied to the surface. Simulating manure incorporation was therefore included to determine whether potential adverse effects related to manure applications in reduced tillage systems might be reduced by immediate incorporation of the manure.

(4a) Current level of no-till (NT), remainder 100% mulch till (MT)
(4b) Current level of no-till, remainder 100% mulch till, all manure incorporated (MI)
(4c) 100% no-till
(4d) 100% no-till, with all manure incorporated (MI)
(4e) 60% MT, 30% NT, 10% CT (conventional tillage)
(4f) 60% MT, 30% NT, 10% CT, with all manure incorporated (MI)
(4g) Current tillage practices, except all manure incorporated (MI)

5. Number of cows increase by 15%.
Scenario 5: What happens if the cow numbers increase by 15%?

This scenario was modeled by simply increasing the amount of manure generated by 15%. Alternatively, the amount of land available for manure application could have been decreased to simulate greater cow density, but model results would be complicated by the associated increase in urban landuse and decrease of agricultural land and the underlying assumptions used to model these landuses. The selected method was chosen because it simply looks at increased cow numbers without other compounding factors that might influence the modeled outcome. Otherwise, the outcome of this scenario may have been influenced more by the change in landuse than the increase in cow density.

6. Expanded row crops and decreased alfalfa acreage (alfalfa acreage decreases by 33.3%).
Scenario 6: What if alfalfa acreage decreases by 33.3% as row crop acreage increases (corn, soybean)?

In this scenario, the baseline dairy rotation was altered by replacing one of the alfalfa years with corn silage, resulting in a 33.3% decrease in the amount of alfalfa acreage. The total amount of applied manure remained the same; however, the total amount of applied commercial fertilizer increased somewhat because starter fertilizer was applied to the additional corn silage crop.

7 All cows fed low phosphorus feed: reduce phosphorus in dairy cow feed ration by 25%.
Scenario 7: What if the amount of phosphorus in the dairy cow feed ration is reduced by 25%?

Phosphorus in the dairy cow feed ration was reduced by 25%, compared to 2000 levels. At levels estimated for the year 2000, this reduction translates to roughly a 25% reduction in manure phosphorus concentrations. The fertilizer/manure input file was adjusted accordingly to simulate this scenario. A detailed justification of these assumptions is discussed in Chapter 2.

8. Innovative manure management options (composting facility, 20% of manure composted).
Scenario 8: What if 20% of farm manure and packing plant waste is composted?

In this scenario, it was assumed that 20% of all the manure that is generated in the subbasin is composted at a central facility instead of applying it directly to fields. Based on a preliminary estimate by Brad Holtz, agronomist with the Brown County LCD, roughly 50% of the nutrients would remain in the liquid fraction after being separated from the solid fraction which is processed at the composting facility (Holtz 2004). The liquid fraction is then applied to the farm fields, so approximately 50% of the remaining nutrients are displaced through composting and are either shipped out of the watershed or used to replace commercial fertilizer. For this scenario, it was assumed that this 50% figure could be applied to phosphorus. Therefore, about 10% of the total amount of phosphorus applied as manure is displaced under the assumptions used in this scenario. Packing plant wastes that are applied to fields within the subbasin were not explicitly accounted for in the model. Therefore, as long as the relative fractions of farm manure and packing plant wastes displaced through composting are about the same, the simulated impact should be approximately correct. If the actual proportion of phosphorus in the solid fraction is higher; for example 66.7%, then only 15% of the manure would need to be composted to reduce the total applied phosphorus by 10%.

Another scenario explored the impact of installing manure digestion facilities which would produce energy and utilize 20% of the manure generated by farms in the subbasin. However, this scenario was not modeled because most operations are run such that the majority of the phosphorus stays on site.[20]

9. Intensive rotational grazing (IRG) adopted by 20%, 40%, or 100% of all dairy farmers.
Scenario 9: What if intensive rotational grazing was adopted by a significant number of dairy operations?

Four options were simulated under this scenario. Under Scenarios 9a, 9b, and 9c, it was assumed that 20%, 40% and 100% of all dairy farms adopted intensive rotational grazing in place of conventional dairy farm management practices. Only dairy operations switched management in these first three options, leaving approximately 20% of the remaining agricultural area unaffected. In Scenario 9d, it was assumed that 100% of all farm operations adopted IRG. For all IRG scenarios, it was assumed that: (1) only pasture was grown on these crop land acres, (2) grazing paddocks were rotated on a regular basis throughout the growing season, (3) a portion of the spring hay crop was harvested for feeding during the non-growing season, (4) cows were pastured as much as possible throughout most of the year, (5) the dairy ration phosphorus level was reduced by 25% compared to the 2000 Baseline Scenario and (6) manure was applied to dairy crop land at the same rate as simulated conventional dairy operations, only most of it was applied via grazing cows, with 20% applied in mid-summer.

10. Urban area doubles (year 2025 to 2030). Current BMPS and WDNR non-point regulation requirements for urban sources).
Scenario 10: What will happen when the amount of urban area in the subbasin doubles?

Three alternative options were simulated under this scenario. In each scenario, it was assumed that the fraction of urban area within the subbasin doubled. Figure 2-3 in Chapter 2 shows the GIS layer that was created to simulate the approximate expanded areal coverage of urbanized areas within the subbasin under this scenario. This GIS layer was used to provide the required inputs to the SWAT model in the same manner that the 2000 landuse layer was utilized for the 2000 Baseline Scenario (i.e., current condition).

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[20] This scenario was originally included in the project. It involved applying anaerobic digestion technology to about 20% of the manure generated in the subbasin. This technology was expected to reduce the quantity of manure to be spread, generate electricity from methane production, and reduce the net amount of phosphorus applied to land in the subbasin. With this method, the liquid fraction is applied to farm fields. The dry fraction might be transported out of the subbasin, but it generally contains a relatively small portion of the total phosphorus in the manure. In addition, most if not all of the farms now implementing this technology currently use the dry fraction as bedding, so the only significant reduction of phosphorus that might be achieved is the net reduction that could accrue from replacing bedding material that is imported and contains a significant amount of phosphorus (Dr. John Katers, University of Wisconsin - Green Bay, personal comm. Nov. 2004).

Urbanized areas shown in Figure 2-3 are only rough approximations of where they might be when the urban area doubles compared to the current 2000 scenario. While the precise location of future urbanized areas is likely to be different than that depicted in Figure 2-3, the effect of spatial errors on the modeled output is expected to be relatively small compared to potential errors caused by faulty assumptions, such as urban loading rates. If long-term rates of urbanization continue, it is estimated that it will take until the year 2025 to 2030 to roughly double the urban area within the subbasin, at which time urban areas would comprise about 55-58% of the subbasin. Because it was assumed that the urban area doubled, the annual amount of land undergoing urbanization was also assumed to double in this scenario, compared to the 1992 and 2000 Baseline scenarios. Consequently, the total load from urbanization would essentially double if efforts were not taken to reduce construction site erosion.

In the first scenario (10a), it was assumed that sediment and phosphorus yields from urban non-point sources would not change. So no reductions were assumed to occur from either new or older urban areas.

In the third scenario (10c Optimistic), it was assumed that areas that had become urbanized since 2000 would have lower sediment and phosphorus yields compared to areas that were already defined as urban under the Baseline 2000 Scenario. Urban management files were created so that resulting simulated loads generally reflected reductions specified in NR151 for new and older urban areas. These files were set up such that sediment reductions were approximately 80% from new urban areas and 40% from older urban areas under the Optimistic Scenario. Phosphorus reductions from urban non-point sources are not specified in NR151, so model inputs were set such that somewhat lower reductions of about 60% were obtained from new urban areas and 30% from older urban areas under the Optimistic Scenario. Loads from urban construction sites were also reduced according to NR151 requirements (80% for TSS; 60% for phosphorus), but as previously stated in Chapter 2 under the urbanizing areas section, these loads were not simulated within the SWAT model.

The second scenario (10b Conservative) assumed that actual reductions from urban areas would be less than those assumed under Scenario 10c. It is possible that not all urban stormwater objectives will be achieved everywhere, or that some specified reductions might not be achieved from a stream monitoring perspective where coarse particles associated with bedload sediment are not typically measured as suspended sediment. Therefore, this more conservative scenario was developed. For Conservative Scenario 10b, model inputs were set so that sediment reductions of approximately 50% were obtained from new urban areas and 25% from older urban areas. Model inputs were also set such that phosphorus reductions of about 38% were obtained from new urban areas and 16% from older urban areas. Loads from urban construction sites were also reduced according to these more conservative levels (50% for suspended sediment and 38% for phosphorus).

11. Buffer strip implementation, alternative VBS simulation methods:
Scenario 11: What if buffers strips are comprehensively implemented throughout the subbasin and alternative VBS simulation methods are applied to predict the impact?

The three options modeled under this scenario were developed to evaluate the effect of alternative VBS simulation methods on potential reductions of phosphorus and sediment. These options all utilize different methods and assumptions to estimate the impact of installing VBS's than the default method which was applied for all other scenarios. In these alternative VBS scenarios, it was assumed that VBS's were installed on 100% of the streams as delineated in WDNR 1:24k hydrology layer.

In the first option (11a), the default method was applied except it was assumed that the trapping efficiency for TSS with hydrologic group B soils was 65%, compared to the default assumption of 45%.

Under the second (11b) and third (11c) options, it was assumed that the suspended sediment and sediment-attached phosphorus trapping efficiencies for B soils were both 80%. The soluble phosphorus trapping efficiency was set at 48%. The impact zone concept was also substantially altered. That is, it was assumed that all source areas could be effectively treated by installing VBS's along only the 1:24k hydrology streams, compared to the 90 m impact zone assumed for all other scenarios, including the 1992 and 2000 Baseline scenarios. With this method, if buffers were installed on 100% of the 1:24k streams, then there should be an 80% reduction in suspended sediment from areas with B soils (i.e., 80% trapping efficiency effecting 100% of the source). The road ditch network and extended stream/ditch network were effectively ignored in these two scenarios. The second option differed from the third in that the effect of forested and wetland areas adjacent to streams was incorporated into the calculations as existing buffer strips that reduced loads from adjacent upland areas, which was also done in the default method. The third option assumed no influence from these areas. All else being equal, there should be a greater impact with the third alternative VBS method because it was assumed that VBS's could be added to the entire delineated stream network, even where adjacent forested or wetland areas already exist to potentially act as buffer strips.

Additional information on the application of these alternative VBS simulation method can be found in the riparian buffer strip section in Chapter 2. Again, the model was not recalibrated when these alternatives were simulated. Instead, an estimation method that is described in Chapter 2 was employed to account for the different techniques.

When NR151 is fully implemented, will it be enough to reach the Green Bay Remedial Action Plan, Science and Technical Advisory Committee's objectives for the Bay?: Another alternative scenario was evaluated to determine the feasibility of using the model framework to estimate the impact of implementing the NR151 requirements. This scenario is not fully developed yet. However, Scenario 2 covers part of the 590 Nutrient Management Standard (Criterion II - Soil Test P Levels and Management Considerations). It does not appear that some site-specific practices or prohibitions that are listed in NR151 can be easily modeled at the scale utilized in this project (e.g., clean water diversions, manure management prohibitions, etc.). One of the key limiting factors in modeling these types of BMP's is that there often is a lack of accurate data on current conditions, particularly with regard to spatially and temporally sensitive practices such as where, when or how manure is applied or stored.

ALTERNATIVE SCENARIOS - RESULTS

The simulated impacts of alternative scenarios on average annual sediment and phosphorus loads to Green Bay from non-point sources relative to the Baseline 2000 Scenario are summarized in Tables 11-1 and 11-2 (1977-2000 climatic period, 24 year model simulations). The simulated impacts of alternative scenarios on loads from the largely agricultural Plum Creek Watershed (LF03) are provided in Table 11-3 so that the effects of landuses other than agriculture are reduced compared to the results shown in Tables 11-1 and 11-2. Therefore, simulated reductions due to increased implementation of conservation tillage, nutrient management, VBS, IRG and other agricultural practices are greater in LF03 than the subbasin as a whole. Some of the simulated reductions are influenced by confounding factors such as the level of BMP's already present under the Baseline 2000 Scenario, or differential effects of routing sediment, sediment-attached phosphorus (includes organic phosphorus) and soluble phosphorus through streams, wetlands and the river.[21]

Excluding Scenarios 1 and 11, the greatest reductions were simulated when Intensive Rotational Grazing (IRG) was widely implemented (Scenario 9). When 100% of all farms switched to IRG, there was a 50% reduction of TSS, and a 48% reduction of total phosphorus delivered to Green Bay. However, TSS was reduced by 81% and total phosphorus by 65% when 100% IRG was simulated in the LF01-15 calibration watershed, which is nearly all rural (data not shown). These reductions are similar to the 64% decrease in total phosphorus simulated by Gassman et al. (2002) when they applied the APEX and SWAT models to the 1,279 km^2 Lake Fork Reservoir Watershed in northeast Texas. Landuse in this Texas watershed was nearly all rural in 1996, and 71% of the watershed land area was used for pasture-based dairy or beef production. It should be noted that the scenarios involving 100% adoption by dairy farmers or all farmers are very unlikely in the subbasin, which is quite different than the watershed modeled in Texas. High levels of BMP adoption are primarily presented here to illustrate the maximum possible simulated effect that might be expected from a particular policy change. Even obtaining the 20% adoption rate of Scenario 9a, and the associated 7% sediment reduction and 7.6% phosphorus reduction from the entire subbasin may require a fairly significant reversal of the current trend toward concentrated large dairy operations. Still, some farmers may see significant financial benefits or other rewards from switching to IRG, particularly if high energy prices favor IRG over large scale concentrated dairy operations.

As shown in Table 11-3, limiting the amount of phosphorus applied to crop agronomic needs and stabilizing soil phosphorus at current levels resulted in a simulated phosphorus decrease of 14.4% from LF03 under Scenario 2a in this project compared to decreases ranging from 21% to 38% in the Texas modeling effort under a somewhat similar scenario (Gassman et al. 2002).

[21] For example, reductions of sediment-attached phosphorus are often slightly greater than those for suspended sediment when VBS's are installed under Scenario 3, and reductions of total phosphorus are nearly the same as for suspended sediment. But the sediment trapping efficiency of the VBS was assumed to be the same as for sediment-attached phosphorus, and higher than for soluble phosphorus. This conflict can be explained by the fact that soluble phosphorus is relatively unaffected as it is transported through the system; whereas, a greater fraction of suspended sediment is trapped, and a somewhat smaller fraction of sediment-attached phosphorus is trapped compared to sediment. Similar effects were obtained when conservation tillage was simulated, and differential transport of the constituents serves to explain these seemingly contradictory results.

Table 11-1. Simulated impacts of Alternative Scenarios on suspended sediment and phosphorus non-point source loads to Green Bay from Lower Fox River Subbasin. Based on 1977-2000 climatic period.

Scenarios	TSS (ton)	Sed-P (kg)	Sol-P (kg)	Total P (kg)
Baseline 2000 Conditions	52,600	56,800	79,600	136,300
1. Entire Subbasin Forested	3,800	2,200	12,300	14,500
2a. Nutrient Management: Soil P stable at 40 ppm (Ag soils only)	52,600	52,400	66,500	118,900
2b. Nutrient Management: Soil P stable at 25 ppm (Ag soils only)	52,600	41,300	52,400	93,700
2c. Nutrient Management: Soil P stable at 50 ppm (Ag soils only)	52,600	59,800	75,800	135,600
3a. VBS - installed on 50% of 1:24k hydrology streams	51,700	55,500	78,300	133,800
3b. VBS - installed on 100% of 1:24k hydrology streams	50,700	54,000	76,700	130,700
3c. VBS - installed on 100% of 1:24k streams & all road ditches	48,600	51,200	73,900	125,100
4a. Conservation Tillage - current NT, rest MT	43,000	46,300	79,000	125,300
4b. Conservation Tillage - current NT, rest MT, all manure incorporated	42,700	44,700	69,100	113,800
4c. Conservation Tillage - 100% NT	32,100	34,100	73,600	107,700
4d. Conservation Tillage - 100% NT, all manure incorporated	31,000	31,500	59,700	91,300
4e. Conservation Tillage - CT10%, MT60%, NT30%	41,400	45,400	77,500	122,900
4f. Conservation Tillage - CT10%, MT60%, NT30% (all manure incorporated)	41,000	43,500	66,700	110,200
4g. Baseline Conditions, but all manure incorporated	52,500	55,400	71,800	127,200
5.Cow #'s Increase by 15%	52,600	57,800	83,000	140,800
6. Decrease alfalfa acreage by 33%, and increase row crops	58,300	64,300	86,000	150,300
7. Dairy P feed ration reduced by 25%	52,600	55,000	73,900	128,900
8. Composting Facility: 20% of manure displaced	52,600	56,100	77,300	133,400
9a. Intensive Rotational Grazing, 20% of dairy farms adopt	48,900	51,900	74,000	125,900
9b. Intensive Rotational Grazing, 40% of dairy farms adopt	45,200	47,100	68,300	115,400
9c. Intensive Rotational Grazing, 100% of dairy farms adopt	32,900	30,600	51,300	82,000
9d. Intensive Rotational Grazing, 100% of ALL farms adopt	26,200	23,800	47,000	70,900
10a. Urban area doubles (about year 2025-30), current BMP practices	53,800	51,400	62,400	113,900
10b. Urban area doubles (about year 2025-30), BMP Level 1 Conservative	39,800	39,700	58,900	98,600
10c. Urban area doubles (about year 2025-30), BMP Level 2 Optimistic	31,400	33,400	55,000	88,400
11a. VBS - Default Method, but 65% TE, All 1:24k streams (estimated)	49,800	52,800	75,500	128,300
11b. VBS - Alt. Method #2a - All 1:24k streams (estimated, with natural VBS's)	34,400	35,100	59,000	94,100
11c. VBS - Alt. Method #2b - All 1:24k streams (estimated, without natural VBS's)	28,900	28,300	50,000	78,300

Table 11-2. Simulated impacts of Alternative Scenarios on suspended sediment and phosphorus non-point source loads to Green Bay from Lower Fox Subbasin, as percent change from Baseline 2000 Scenario.

Scenarios	TSS	Sed-P	Sol-P	Total P
1. Entire Subbasin Forested	-92.8%	-96.1%	-84.6%	-89.4%
2a. Nutrient Management: Soil P stable at 40 ppm (Ag soils only)	-0.0%	-7.7%	-16.4%	-12.8%
2b. Nutrient Management: Soil P stable at 25 ppm (Ag soils only)	-0.0%	-27.3%	-34.1%	-31.3%
2c. Nutrient Management: Soil P stable at 50 ppm (Ag soils only)	0.0%	5.3%	-4.7%	-0.5%
3a. VBS - installed on 50% of 1:24k hydrology streams	-1.6%	-2.2%	-1.6%	-1.9%
3b. VBS - installed on 100% of 1:24k hydrology streams	-3.5%	-4.8%	-3.6%	-4.1%
3c. VBS - installed on 100% of 1:24k streams & all road ditches	-7.5%	-9.8%	-7.2%	-8.3%
4a. Conservation Tillage - current NT, rest MT	-18.1%	-18.4%	-0.7%	-8.1%
4b. Conservation Tillage - current NT, rest MT, all manure incorporated	-18.7%	-21.2%	-13.2%	-16.5%
4c. Conservation Tillage - 100% NT	-39.0%	-39.9%	-7.6%	-21.0%
4d. Conservation Tillage - 100% NT, all manure incorporated	-41.0%	-44.4%	-24.9%	-33.1%
4e. Conservation Tillage - CT10%, MT60%, NT30%	-21.2%	-20.0%	-2.6%	-9.9%
4f. Conservation Tillage - CT10%, MT60%, NT30% (all manure incorporated)	-22.0%	-23.3%	-16.2%	-19.2%
4g. Baseline Conditions, but all manure incorporated	-0.1%	-2.5%	-9.8%	-6.7%
5.Cow #'s Increase by 15%	-0.0%	1.9%	4.3%	3.3%
6. Decrease alfalfa acreage by 33%, and increase row crops	10.9%	13.2%	8.1%	10.2%
7. Dairy P feed ration reduced by 25%	-0.0%	-3.1%	-7.1%	-5.5%
8. Composting Facility: 20% of manure displaced	0.0%	-1.2%	-2.8%	-2.2%
9a. Intensive Rotational Grazing, 20% of dairy farms adopt	-7.0%	-8.5%	-7.0%	-7.6%
9b. Intensive Rotational Grazing, 40% of dairy farms adopt	-14.1%	-17.1%	-14.1%	-15.3%
9c. Intensive Rotational Grazing, 100% of dairy farms adopt	-37.5%	-46.0%	-35.5%	-39.9%
9d. Intensive Rotational Grazing, 100% of ALL farms adopt	-50.2%	-58.0%	-40.9%	-48.0%
10a. Urban area doubles (about year 2025-30), current BMP practices	2.3%	-9.4%	-21.5%	-16.5%
10b. Urban area doubles (about year 2025-30), BMP Level 1 Conservative	-24.3%	-30.1%	-26.0%	-27.7%
10c. Urban area doubles (about year 2025-30), BMP Level 2 Optimistic	-40.2%	-41.2%	-30.9%	-35.2%
11a. VBS - Default Method, but 65% TE, All 1:24k streams (estimated)	-5.2%	-7.0%	-5.1%	-5.9%
11b. VBS - Alt. Method #2a - All 1:24k streams (estimated, with natural VBS's)	-34.6%	-38.2%	-25.9%	-31.0%
11c. VBS - Alt. Method #2b - All 1:24k streams (estimated, without natural VBS's)	-45.0%	-50.2%	-37.2%	-42.6%

Table 11-3. Simulated impacts of Alternative Scenarios on suspended sediment and phosphorus non-point source loads from the Plum Creek Watershed (LF03), as percent change from Baseline 2000 Scenario.

Scenarios	TSS	Sed-P	Sol-P	Total P
1. Entire Subbasin Forested	-92.5%	-95.0%	-86.6%	-90.7%
2a. Nutrient Management: Soil P stable at 40 ppm (Ag soils only)	-0.0%	-9.4%	-19.0%	-14.4%
2b. Nutrient Management: Soil P stable at 25 ppm (Ag soils only)	-0.0%	-33.5%	-39.2%	-36.5%
2c. Nutrient Management: Soil P stable at 50 ppm (Ag soils only)	0.0%	6.6%	-5.5%	0.3%
3a. VBS - installed on 50% of 1:24k hydrology streams	-2.9%	-3.5%	-2.5%	-3.0%
3b. VBS - installed on 100% of 1:24k hydrology streams	-6.1%	-7.2%	-5.2%	-6.1%
3c. VBS - installed on 100% of 1:24k streams & all road ditches	-11.5%	-13.3%	-9.5%	-11.3%
4a. Conservation Tillage - current NT, rest MT	-27.4%	-23.0%	-1.8%	-12.0%
4b. Conservation Tillage - current NT, rest MT, all manure incorporated	-28.2%	-26.5%	-16.5%	-21.3%
4c. Conservation Tillage - 100% NT	-53.6%	-47.2%	-9.3%	-27.6%
4d. Conservation Tillage - 100% NT, all manure incorporated	-56.1%	-52.8%	-29.9%	-40.9%
4e. Conservation Tillage - CT10%, MT60%, NT30%	-31.4%	-25.6%	-3.8%	-14.3%
4f. Conservation Tillage - CT10%, MT60%, NT30% (all manure incorporated)	-32.6%	-29.7%	-19.9%	-24.6%
4g. Baseline Conditions, but all manure incorporated	-0.2%	-3.0%	-11.4%	-7.3%
5.Cow #'s Increase by 15%	0.0%	2.3%	5.0%	3.7%
6. Decrease alfalfa acreage by 33%, and increase row crops	15.0%	15.9%	9.6%	12.7%
7. Dairy P feed ration reduced by 25%	0.0%	-3.8%	-8.3%	-6.1%
8. Composting Facility: 20% of manure displaced	0.0%	-1.5%	-3.3%	-2.4%
9a. Intensive Rotational Grazing, 20% of dairy farms adopt	-9.7%	-10.4%	-8.2%	-9.2%
9b. Intensive Rotational Grazing, 40% of dairy farms adopt	-19.6%	-21.0%	-16.4%	-18.6%
9c. Intensive Rotational Grazing, 100% of dairy farms adopt	-50.9%	-55.0%	-41.2%	-47.8%
9d. Intensive Rotational Grazing, 100% of ALL farms adopt	-66.2%	-68.6%	-46.5%	-57.1%
10a. Urban area doubles (about year 2025-30), current BMP practices	-8.9%	-13.1%	-19.1%	-16.2%
10b. Urban area doubles (about year 2025-30), BMP Level 1 Conservative	-24.2%	-25.0%	-21.5%	-23.2%
10c. Urban area doubles (about year 2025-30), BMP Level 2 Optimistic	-33.4%	-31.4%	-23.9%	-27.5%
11a. VBS - Default Method, but 65% TE, All 1:24k streams (estimated)	-8.9%	-10.3%	-7.4%	-8.8%
11b. VBS - Alt. Method #2a - All 1:24k streams (estimated, with natural VBS's)	-51.5%	-52.6%	-34.9%	-43.4%
11c. VBS - Alt. Method #2b - All 1:24k streams (estimated, without natural VBS's)	-61.0%	-62.4%	-44.1%	-52.9%

However, the simulated reduction in LF03 increases to 36.5% when soil phosphorus (Bray-P1) is decreased from 40 ppm to 25 ppm under Scenario 2b. So, it is possible that reductions may have been somewhat greater if soil phosphorus levels were not as elevated as they currently are in the subbasin. Phosphorus reductions of 12.8% and 31.3% are predicted for the entire subbasin under Scenarios 2a and 2b, respectively. Under Scenario 2a, the reduction of total phosphorus in the mostly rural subwatershed LF01-15 was 16% (data not shown), which is slightly higher than in LF03. An additional decrease of 2 to 3% is predicted under Scenario 2a and 2b when only the last 12 years of the model simulation are utilized for comparison purposes. Therefore, the long-term benefits from lowering soil phosphorus levels are actually predicted to be greater than the figures listed in the tables show.

Clearly, steps need to be taken to reduce soil phosphorus levels if the simulated impacts under Scenario 2b are correct. Results from Scenario 2b show that elevated levels of phosphorus in soils likely play a key role in the relatively high phosphorus export to streams observed in the subbasin. Results from this scenario imply that adverse impacts related to accumulations of excess soil phosphorus over the past 30 years may have countered many of the positive benefits gained from improved farm practices over this same period. Results from Scenario 2c suggest that allowing average soil phosphorus levels to rise to 50 ppm (Bray-P1) before requiring them to stabilize will not produce a positive improvement.

Under Scenario 3, reductions of phosphorus to Green Bay ranged from 1.9% when 50% of the streams that were delineated in the 1:24k stream network had a VBS installed (3a) , to 4.1% when a VBS was installed on all of these streams (3b), up to 8.3% when a VBS was installed on all of these streams plus all road ditches

(3c). Sediment reductions were about the same. Instead of installing a VBS on all of the road ditches under Scenario 3c, a VBS could be installed on the extended stream network (not delineated in the 1:24k stream network), and the results may be similar to 3c given that the extended stream network may be nearly as large as the 1:24k network. The 11.5% reduction in sediment from LF03 to lower Green Bay that is listed in Table 11-3 under Scenario 3c provides an example of the simulated impact that might be expected in a watershed where it was assumed that a negligible number of VBS's were installed by the year 2000. It should be noted that "natural" VBS's already present along streams reduced the potential for improvements under Scenario 3 with the default VBS simulation method.

Scenario 11 was provided to show the potential effect of utilizing alternative VBS simulation methods. Under Scenario 11a, VBS simulated reductions increase by about 44% over those listed in under Scenario 3b, as a result of raising the default method's trapping efficiency to 65%. Results from Scenarios 11b and 11c show that there is a dramatic difference between the default method and these alternative methods. For example, suspended sediment is predicted to be reduced by 61%, and phosphorus by 53% in LF03 under Scenario 11c (Table 11-3).

Under Scenario 4, conservation tillage options decreased simulated subbasin sediment loads to Green from 18% to 39%, depending on the level of reduced tillage. Similarly, total phosphorus loads decreased from 8% to 21%, although much lower reductions were predicted for soluble phosphorus. Total phosphorus loads decreased from 16.5% to 33% when it was assumed that all manure was incorporated immediately. Greater improvements were simulated under this set of options because phosphorus concentrations near the soil surface are expected to be significantly elevated under no-till. In the mostly rural Plum Creek Watershed (LF03), larger reductions of up to 56% of sediment, and 41% of phosphorus exported to lower Green Bay were simulated (Table 11-3).

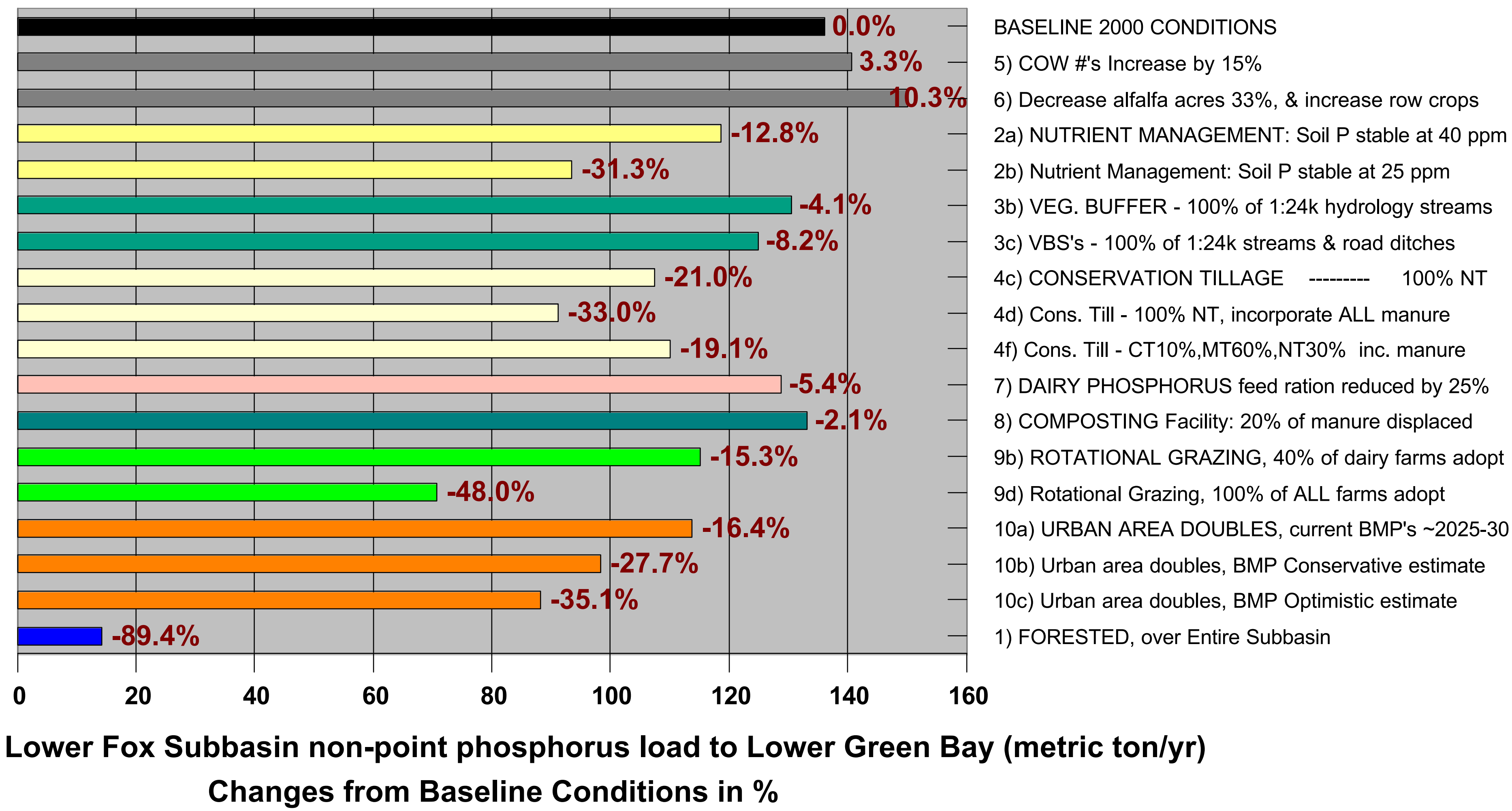
Under Scenario 5, a 3.3% increase in total phosphorus is predicted if the number of cows in the subbasin are increased by 15% and nothing is done to compensate for the increased input of phosphorus. Results from Scenario 6 indicate that if the current trend continues, and the relative acreage of alfalfa is decreased by 33% and replaced with a more erosive crop such as corn silage, then subbasin non-point source loads of sediment to Green Bay are predicted to increase by 11%, and phosphorus by 10%, if nothing else changes.

Under Scenario 7, reducing the level of phosphorus present in the dairy feed ration by 25% caused a 5.5% decrease in the subbasin phosphorus load to lower Green Bay, and a 6.1% decrease in the load from LF03 to Green Bay. Under Scenario 8, displacing 20% of generated manure through implementation of a manure composting program caused a 2.2% decrease in the subbasin phosphorus load to lower Green Bay, and a 2.4% decrease in the load from LF03 to Green Bay. A greater simulated reduction would be expected if more than the 50% of the phosphorus is retained in the dry fraction sent to be composted.

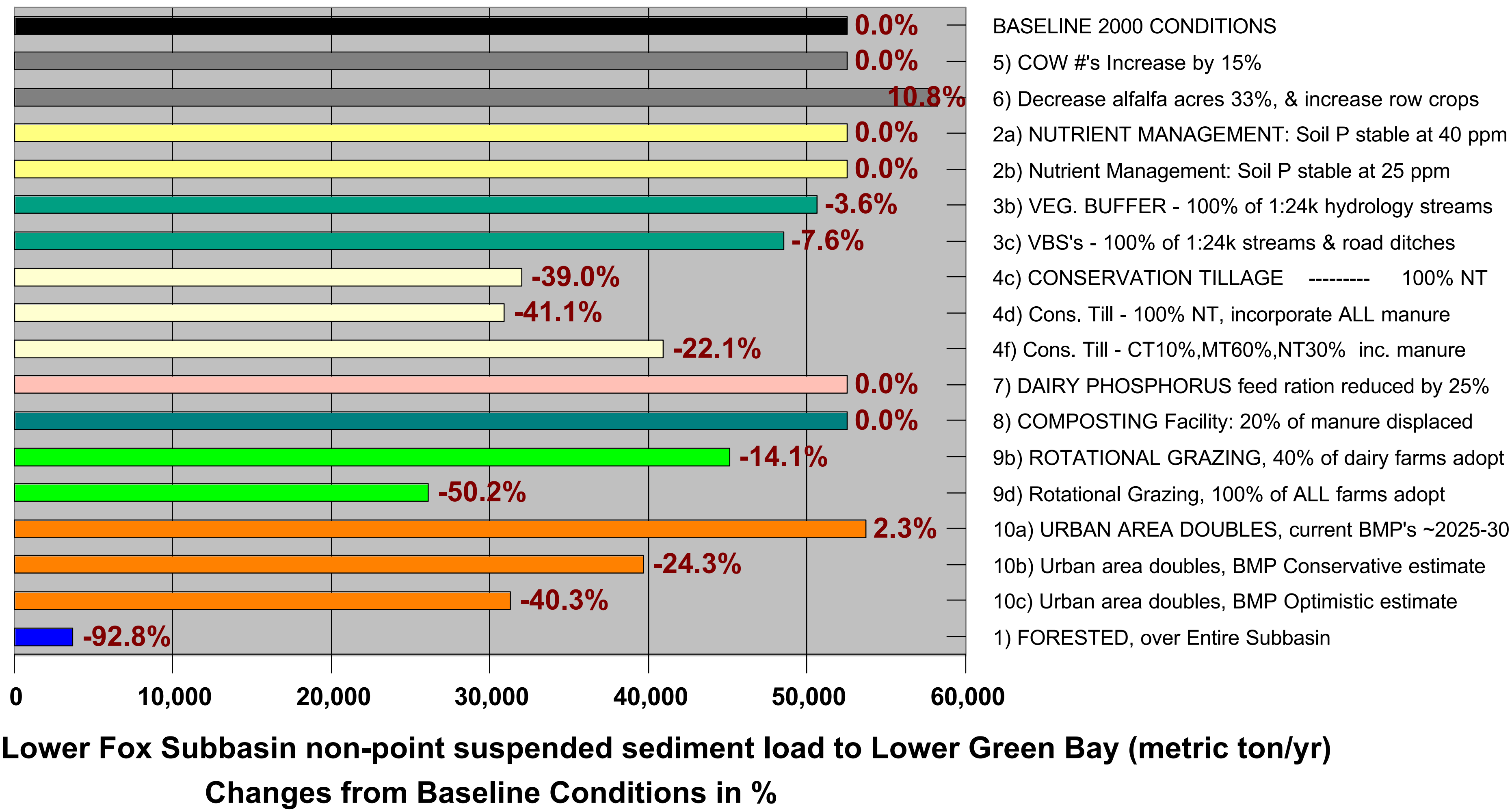
If the urban area doubles in the subbasin, simulations under Scenario 10 imply that there should be a substantial decrease in sediment and phosphorus non-point source loads to Green Bay from the subbasin, assuming urban stormwater requirements under NR151 are fully implemented. Predicted sediment reductions range from 24% to 40% for the conservative and optimistic scenarios, respectively. Predicted phosphorus reductions range from 28% to 35% for the conservative and optimistic scenarios, respectively. These reduction would be expected to occur even if no management changes are made from the other sources. <u>However, the simulated changes are heavily dependant on the assumed yields of sediment and phosphorus from urban sources relative to agricultural sources</u>. If the actual yields from urban and agriculture were approximately the same, then instead of getting a 16.5% reduction in phosphorus from the subbasin, the predicted impact under Scenario 10a ought to be negligible because this scenario assumed that urban management practices did not change appreciably. The predicted impact on sediment loads under Scenario 10a is a slight increase of 2.3%, which implies that simulated sediment yields from urban areas are approximately the same as those from rural areas where the predominant landuse is agriculture.

While the database used to calibrate the urban component of the model was fairly robust and primarily composed of sites from Wisconsin; it was not very site-specific relative to the database that was utilized to calibrate the model to predict stream flow and loads from mostly agricultural areas. My confidence in the accuracy of the modeled results would've been much less had the calibration and validation data sets been composed of monitoring data from rural areas that were not within the subbasin. Therefore, monitoring of an urban area within the subbasin is highly recommended, for it will increase confidence in model predictions by providing the data needed to calibrate and validate the model in an urban setting. This need for additional monitoring is particularly important given that urban landuse is expected to become the dominant landuse within the subbasin in the next 25 to 30 years.

Addendum - Fig. 11-1. Simulated impact of Alternative Scenarios on phosphorus non-point source loads.



Addendum - Fig. 11-2. Simulated impact of Alternative Scenarios on sediment non-point source loads.



CHAPTER 12. SUMMARY AND CONCLUSIONS

The following points summarize the major findings and conclusions of this project:

1) Modifications to the SWAT 2000 model (version 4/18/2001) were necessary to produce simulated results that adequately tracked observed stream flow and loads in the subbasin. The most important changes were the addition of a coefficient to reduce evapotranspiration by a factor of 0.806 and modification of the sediment sub-routine to allow adjustments of the coefficient and exponents in the sediment equation to facilitate calibration.

2) Overall, the modified and calibrated SWAT model performed well during the calibration and validation periods that were examined in this project. With some exceptions, the simulated daily and monthly hydrographs preserved the peaks and recessions of the observed hydrographs. Simulated monthly water yields were generally in good agreement with observed yields. Simulated total water yields were in close agreement with observed yields.

3) The model was able to predict suspended sediment and phosphorus event loads reasonably well at the Upper Bower Creek, Duck Creek and East River-Monroe Street monitoring locations. Direct comparisons between individual events, statistical measures and graphical relationships support the conclusion that the model can be applied to predict suspended sediment and phosphorus loads at the subwatershed and watershed scale with an acceptable degree of accuracy.

4) Simulated phosphorus export to Green Bay and to the Duck Creek monitoring station were generally close to loads estimated by Robertson of the USGS (personal comm. 2004) and Klump et al. (1997), and by Robertson of the USGS, respectively. The model was less able to match suspended sediment export to Green Bay, but still provided acceptable predictions. The simulated sediment load at the Duck Creek station was greater than the load directly estimated by Robertson. With this exception, simulated sediment loads were reasonably close to observed loads. Overall, there is sufficient evidence to support the conclusion that the model can be applied to predict sediment and phosphorus loads to Green Bay from watersheds in the subbasin with an acceptable degree of accuracy.

5) Substantial differences in suspended sediment and phosphorus yields among the watersheds in the subbasin were simulated by the model. Water quality data collected by government agencies from some of these watersheds are consistent with simulated differences, thereby lending credibility to the relative rankings among the watersheds.

6) Additional continuous monitoring to be conducted by the Lower Fox River Watershed Monitoring Program (www.uwgb.edu/watershed) and the GBMSD at five major stations from 2004 to 2006 will help to better assess the ability of the model to estimate loads throughout the subbasin.

7) Under the Baseline 1992 Scenario, Lake Winnebago was the largest single source of suspended sediment to lower Green Bay, followed by agriculture, river growth of biotic solids, and urban and urbanizing sources. Point sources and other sources of suspended sediment contributed 2.4% and 2.3%, respectively.

8) Under the Baseline 1992 Scenario, Lake Winnebago was the largest single source of phosphorus to lower Green Bay, followed by point sources, agriculture, and urban and urbanizing sources. Point source contributions had decreased by the year 2000, so agriculture became a larger relative source.

9) Simulations of alternative scenarios showed a large range in sediment and phosphorus reductions. The largest reductions related to BMP's were obtained through wide-scale implementation of intensive rotational grazing, followed by conservation tillage and nutrient management. However, some of the higher BMP adoption rates that were simulated are not likely to be achieved in the near future.

10) If urban area doubles in the subbasin, simulations under Alternative Scenario 10 imply that there should be a substantial decrease in sediment and phosphorus loads to Green Bay from non-point sources in the subbasin. Predicted sediment reductions range from 24% to 40% for the conservative and optimistic scenarios, respectively. Predicted phosphorus reductions range from 28% to 35% for the conservative and optimistic scenarios, respectively.

11) The accuracy of the simulated changes under Scenario 10 (urban area doubles) depends greatly on the assumed yields of sediment and phosphorus from urban sources relative to agricultural sources. A major shift to urban landuse as the dominant landuse in the subbasin is expected to occur in the next 25 to 30 years. Therefore, monitoring of an urban area within the subbasin is highly recommended, for it will increase confidence in model predictions by providing the data needed to calibrate and validate the model in an urban setting.

12) When comparing the simulated impact of implementing BMPs, caution should be used so as to not place too much emphasis on small differences. Small differences may be dwarfed by known and unknown sources of error.

Conclusion: Direct comparisons between individual events, statistical measures and graphical relationships support the conclusion that the model can be applied to predict sediment and phosphorus loads at the subwatershed and watershed scale with an acceptable degree of accuracy. Furthermore, there is sufficient evidence to support the conclusion that the model can be applied to predict sediment and phosphorus loads to Green Bay from watersheds in the subbasin with an acceptable degree of accuracy. Therefore, I conclude that the SWAT model, as applied in this project, can be reliably used as a tool to make improved management decisions.

Models of all types should primarily be judged by a criterion that is based on whether we can make better decisions with the model, or without it. In my judgement, the SWAT model as applied in this project meets that criterion. However, the most important tools are the skills and decision making abilities of the resource manager(s) who must be aware of the limitations of each model they rely on. Limitations that potentially affect SWAT-simulated results include potential input errors, inappropriate assumptions and an inability to mimic all of the complex processes and interactions which affect sediment and phosphorus delivery to streams, and nutrient and sediment transport to lower Green Bay. The numerous interactions between climatological factors, plants, soil, nutrients, soil organisms, and management practices are difficult to comprehend, let alone accurately predict. Models are limited by our understanding of the system and our ability to provide accurate, representative inputs at the appropriate scale.

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