Des Moines Headwaters, Lower Des Moines, and East Fork Des Moines River Basins Watershed Model Development)

Prepared for

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June 27, 2016

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1 Introduction

This report transmits and describes the hydrologic and water quality calibration of a watershed model of Minnesota's Des Moines River Headwaters, Lower Des Moines, and East Fork Des Moines River watersheds (8-digit hydrologic unit code [HUC8]: 07100001, 07100002, and 07100003, collectively referred to as the Des Moines River Watershed in this report) developed using the Hydrologic Simulation Program - FORTRAN or HSPF model (Bicknell et al., 2014). The Minnesota Pollution Control Agency (MPCA) is developing HSPF models for most HUC8 watersheds in Minnesota. These models are intended to provide information that supports total maximum daily load studies (TMDLs), watershed restoration and protection strategies, and comprehensive watershed planning under Minnesota's Watershed Approach (Figure 1-1). In addition to simulating hydrology, these models are designed to support biological stressor identification and analysis of pollution-related impairments such as elevated turbidity and the effects of elevated nutrient concentrations. The models are also useful to support analysis needed to develop TMDLs for dissolved oxygen and temperature, as well as to provide a tool for evaluating appropriate point source effluent limits for permitted facilities and evaluating management scenarios.

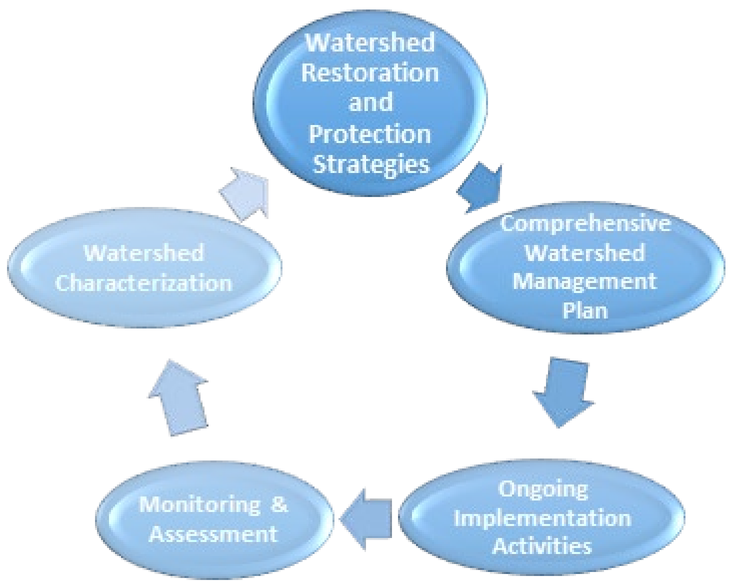

Figure 1-1. Minnesota's Watershed Approach

A watershed model is a tool to aid understanding of processes and consequences of human activities in a river basin, but is only one among a variety of tools. In particular, watershed models are not substitutes for the direct monitoring of physical and biological conditions. When properly calibrated to represent observations, the models can, however, provide a reasonable mechanism for the extrapolation of monitoring data in space (to unmonitored locations) and in time (to unmonitored or future time periods). The watershed model also enables experiments to investigate how changes (such as changes in land use, management practices, or climate) may affect conditions in the watershed and allow stakeholders to plan accordingly. To be useful for these purposes the credibility of the model (and its associated level of uncertainty) must be established through comparison to real world data and through stakeholder input. This report is the initial step in that process.

The Des Moines River - Headwaters watershed covers approximately 1,334 square miles of parts of Lyon, Pipestone, Murray, Cottonwood, Nobles, and Jackson counties in south-western Minnesota.

Beaver Creek, Lime Creek, Lake Shetek, Heron Lake (including Jack Creek and Okabena Creek), and West Fork Des Moines River are its sub-watersheds. The Des Moines – Headwaters basin flows into the Upper Des Moines watershed (hydrologic unit code 07100002), which spans the Minnesota – Iowa border and includes parts of Jackson and Martin counties. This modeling effort also covers parts of the East Fork Des Moines River (0710003), which originates in Minnesota and covers parts of Jackson and Martin counties but joins the Des Moines River in Iowa.

The extent of the Des Moines River HSPF model is shown in Figure 1-2. Although the project focus area lies in Minnesota, the model has been extended into Iowa to allow for the use of long-term flow monitoring data for model calibration and validation. In addition, one tributary flows from Iowa to the Minnesota portion of the watershed.

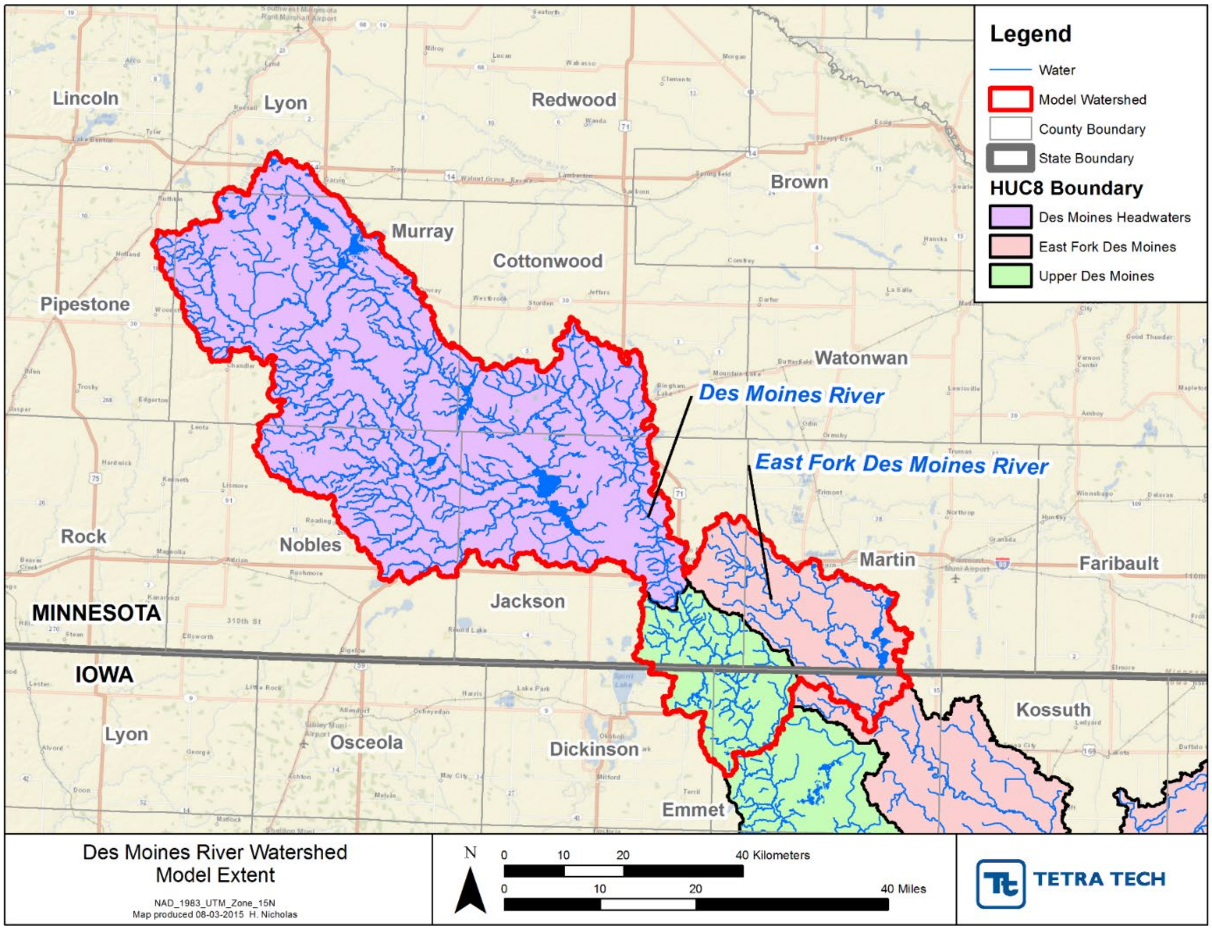

Figure 1-2. Des Moines Headwaters, Upper Des Moines, and East Fork Des Moines River Watershed Model Area

Two meetings with stakeholders were held as part of the model development process: September 30, 2015 and April 27, 2016. Meeting attendees represented the following organizations:

- Cottonwood Soil and Water Conservation District
- Heron Lake Watershed District
- Jackson County
- Jackson Soil and Water Conservation District
- Martin Soil and Water Conservation District

- Minnesota Board of Water and Soil Resources
- Minnesota Department of Agriculture
- Minnesota Department of Natural Resources
- MPCA (Willmar and Mankato offices)
- Murray County
- Nobles County

At the first meeting, an overview of the project was provided, including the model structure, types of input and output data, and potential uses of the model. A data inventory was presented and stakeholders were asked to provide information on additional data that could be incorporated into the model development or calibration. At the second meeting, the model structure was summarized, including the data used to develop and calibrate the model. Preliminary hydrology and sediment calibration graphics and the sediment source assessment were presented. Potential approaches to model scenarios were discussed.

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2 Watershed Model Development

2.1 Upland Representation

The HSPF model for the Des Moines River watershed was set up using a Hydrologic Response Unit (HRU) approach. In general, the HRU approach holds that landscapes possess an identifiable spatial structure, and that the corresponding patterns of runoff and stream chemistry are strongly influenced by climate, geology, and land use. An HRU is defined as a unit of land with relatively homogenous hydrologic properties determined by its underlying characteristics of soils, slopes, and land cover.

2.1.1 Geology, Soils, and Slopes

Like most of Minnesota, the surface features of the Des Moines River watershed were shaped by glaciation. Prior to agriculture, the native vegetation was dominantly tall grass prairie.

The Des Moines River watershed is characterized by various till deposits from the Des Moines lobe of the Wisconsin Glaciation (Figure 2-1). These till deposits are generally loam to clay-loam in texture and can range in thickness from less than 100 to more than 500 feet. A buried Quaternary aquifer is present in much of the watershed. The underlying bedrock geology (Figure 2-2) is primarily sandstone or shale, with substantial areas of conglomerate and amphibolite in the Des Moines Headwaters.

The entire watershed is relatively flat with higher elevations along the western edges of the watershed (Figure 2-3). HRUs can be distinguished by slope classes where slope varies significantly within a land use or soil type. Given the mild slopes prevalent in the Des Moines watershed this was not deemed necessary for this model.

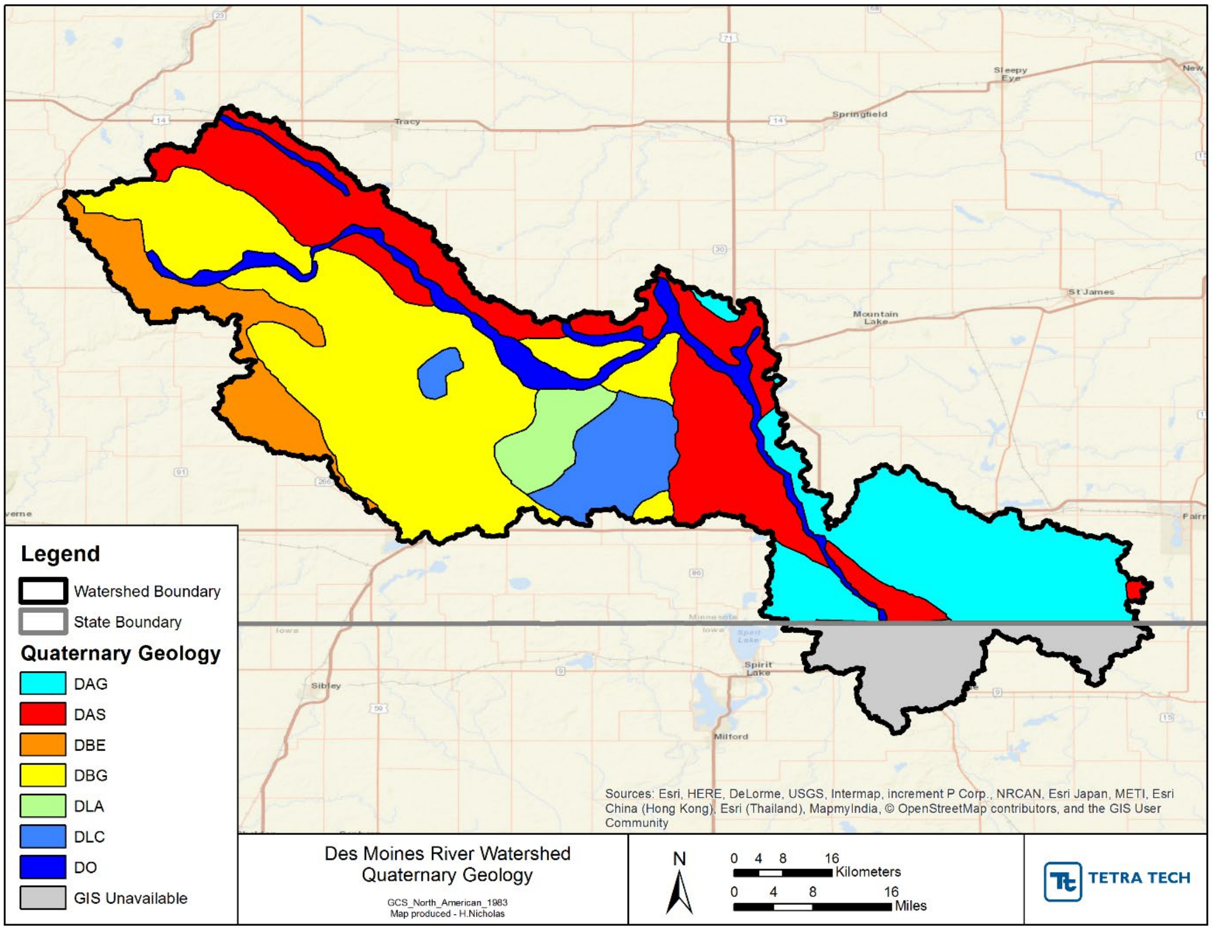

Figure 2-1. Quaternary Geology of the Des Moines River Watershed (Minnesota Portion)

Key:

DAG	Ground Moraine (Des Moines Lobe – Bemis Moraine)
DAS	Stagnation Moraine (Des Moines Lobe - Altamont Moraine)
DBE	End Moraine (Des Moines Lobe – Bemis Moraine)
DBG	Ground Moraine (Des Moines Lobe – Bemis Moraine)
DLA	Sand and Gravel (Glacial Lake Sediment – Undivided as to Moraine)
DLC	Clay and Clayey Silt (Glacial Lake Sediment – Undivided as to Moraine)
DO	Outwash – Undivided as to Moraine Association

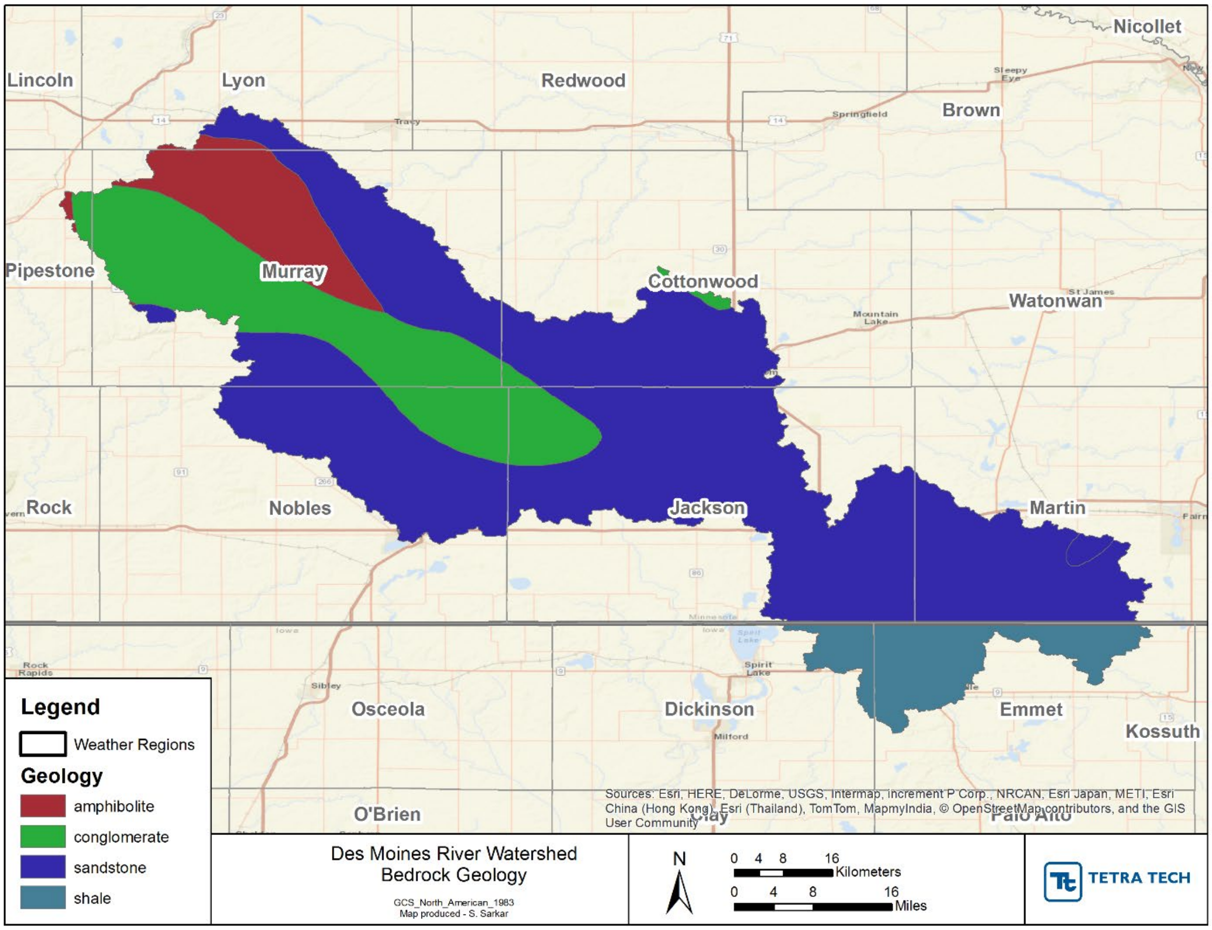

Figure 2-2. Bedrock Geology of the Des Moines River Watershed

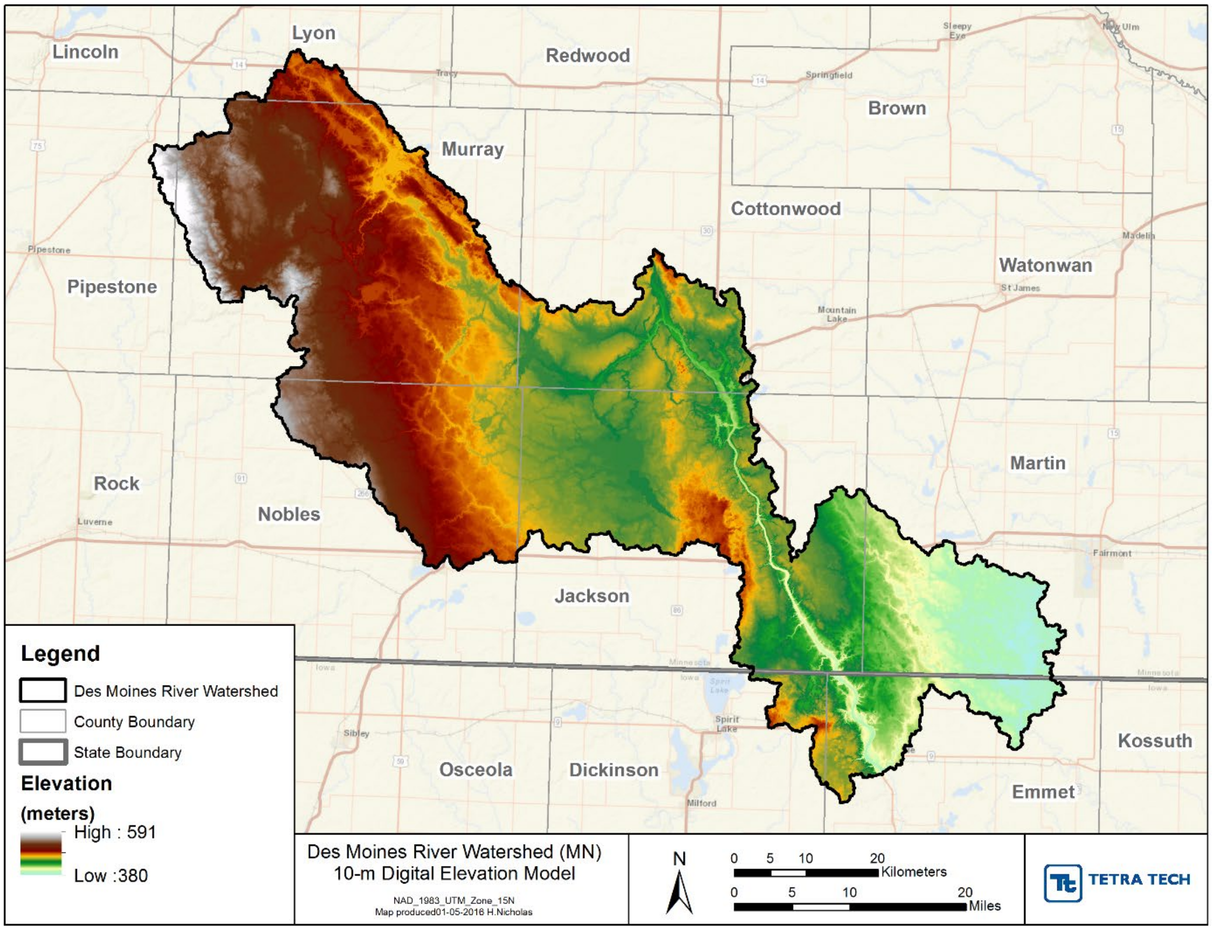

Figure 2-3. Digital Elevation Model of the Des Moines River Watershed

For the purposes of hydrologic modeling, soils in the watershed were distinguished primarily by hydrologic soil group (HSG), which classifies soils according to infiltration potential, from HSG A (excessively drained) to D (poorly drained). The distribution of HSGs is shown in Figure 2-4, determined using 10-meter resolution gridded Soil Survey Geographic database (gSSURGO). The watershed contains a large proportion of coarse-grained, well-drained soils associated with glacial outwash channels. For soils with a dual designation (*e.g. "B/D"*), the two designators represent performance under drained and undrained conditions. Soils with dual designations are bested suited for agriculture with the use of artificial drainage through the installation of drain tiles and ditches. Many of these areas were originally seasonal wetlands, but have been converted to productive corn/soy agriculture through ubiquitous use of tile drains. The land use processing uses the first (drained) designator for cropland and the second (undrained) designator for all other land uses. The wetlands/water land cover class is not subdivided by HSG, and HSG is not relevant to impervious or developed land uses due to the disturbance of native soils and altered soil characteristics in built-up environments.

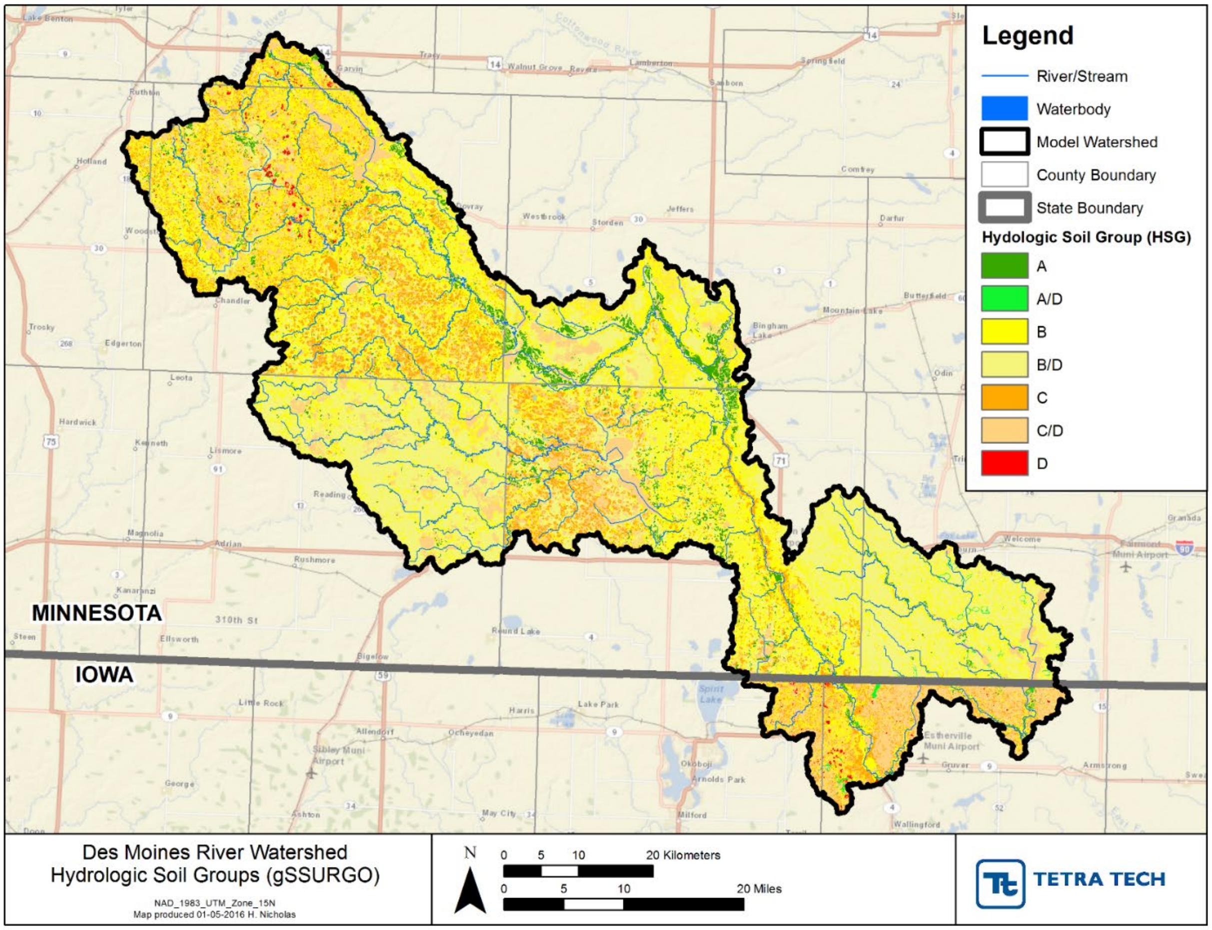

Figure 2-4. Hydrologic Soil Groups in the Des Moines River Watershed

2.1.2 Land Use and Land Cover

Land use in the model is primarily based on National Land Cover Database (NLCD) 2011 (MRLC, 2011). According to NLCD 2011, cultivated land is the major land use covering approximately 81% of the watershed area.

The land use in the watershed has been stable over recent time. The USDA Cropland Data Layer (CDL; http://www.nass.usda.gov/Research_and_Science/Cropland/SARS1a.php) coverage from 2010 to 2014 were reviewed and show that corn and soybeans occupy the majority of the watershed, and that these and other land uses have similar acreages from year to year. NLCD data for 2006 was also reviewed to investigate land use change over time in the watershed (Table 2-1). Comparison of the two datasets does not suggest a need for land use change representation in the modeling. NLCD classes were aggregated for modeling purposes based on MPCA guidelines as summarized in Table 2-2.

Table 2-1. NLCD Land Use for 2006 and 2011 in the Des Moines River Watershed

NLCD Land Use	2006 (acres)	2011 (acres)
Open Water	29,050	29,624
Developed, Open Space	54,234	52,763
Developed, Low Intensity	7,358	8,552
Developed, Medium Intensity	2,674	3,126
Developed, High Intensity	662	851
Barren Land	421	418
Mixed Forest	4,146	4,058
Deciduous Forest	51	48
Evergreen Forest	8,520	8,471
Shrub	961	1,030
Herbaceous Grassland	45,969	45,949
Pasture Hay	16,465	16,114
Cultivated Crops	871,604	871,171
Woody Wetlands	1,684	1,674
Emergent Herbaceous Wetlands	29,133	29,083

Table 2-2. Land Use Aggregation for the Des Moines River Watershed HSPF Model

NLCD Land Use	Percent of Watershed	Model Land Use
Deciduous Forest	0.38%	Forest
Evergreen Forest	0.00%	
Mixed Forest	0.79%	
Pasture/Hay	1.50%	Pasture
Shrub/Scrub	0.10%	Grassland
Barren	0.04%	
Herbaceous	4.28%	
Cultivated Crops	81.20%	Cropland
Developed Open Space	4.92%	Dev Open
Developed Low Intensity	0.80%	Dev Low
Developed Medium Intensity	0.29%	Dev Med/High
Developed High Intensity	0.08%	
Woody Wetlands	0.16%	Water/Wetlands
Emergent Wetlands	2.71%	
Water	2.76%	

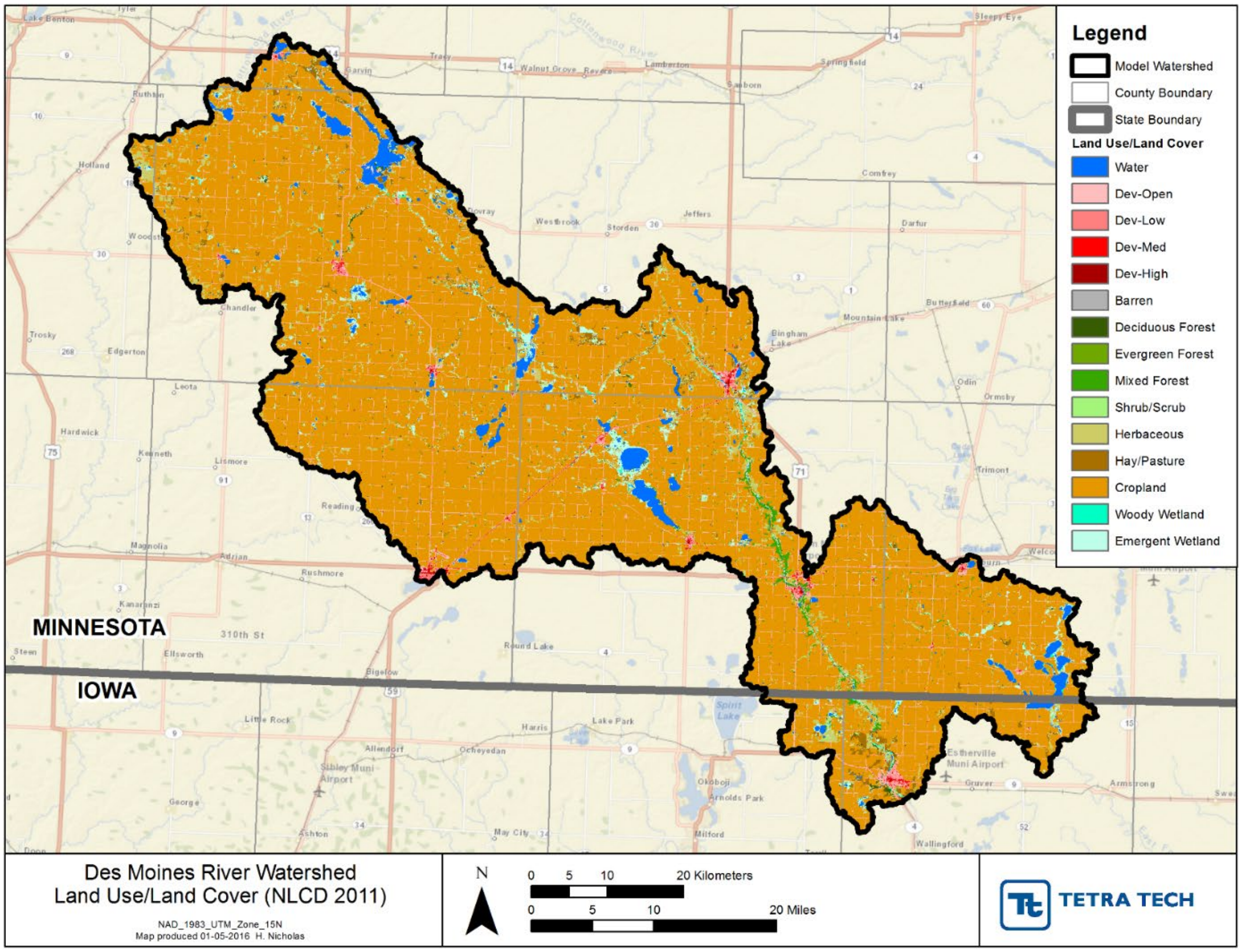

Figure 2-5. Land use in the Des Moines River Watershed from NLCD 2011

2.1.3 Development of HRUs

HRUs were developed consistent with the methods outlined in Modeling Guidance for BASINS/HSPF Applications under the MPCA One Water Program (AQUA TERRA, 2012). The HRU distribution is shown in Table 2-3 and Figure 2-6. Each land segment is assigned a three-digit numeric code of the form *abc* and represents the land use-HSG combination associated with each HRU. Different weather regions are assigned to HRUs by adding a multiple of *50* to the numeric code for each weather station.

The HRU numbering scheme summarized in Table 2-3 is applied directly to pervious land segments (PERLNDs). The numbers assigned to impervious land segments (IMPLNDs) are the same as those associated with their respective pervious land segment, although the HSG designation is not relevant to impervious land. The imperviousness raster associated with NLCD land cover was used to determine impervious fractions with developed land use classes. Effective imperviousness for each developed land use class, calculated from total imperviousness using Sutherland equation (AQUA TERRA, 2012), are shown in Table 2-3. Impervious HRUs in the model are 108, 109, and 110 for developed open, low density developed, and medium/high density developed land, respectively.

The HRUs associated with cultivated land (111-Cropland AB, 112-Cropland CD, 113-Cropland Drained) were further categorized by tillage practices and manure application. For row crops, an important distinction is whether conventional or conservation tillage is used (where conservation tillage is defined as maintaining at least 30 percent residue cover or using no till practices). Information on conservation tillage is available only at the county scale due to producer privacy concerns, and approximate rates of conservation tillage must be estimated by area-weighting the county level data to model subbasins. Information regarding county-level cropland tillage practices obtained from the 2007 Tillage Transect Survey (Minnesota Board of Water and Soil Resources, 2008) was used to differentiate cropland in the model according to conservation and conventional tillage practices. Based on the survey, about 57% of cropland in the Minnesota portions of these watersheds uses conventional tillage, and about 43% uses conservation tillage. Tillage transects were not obtained for the Iowa portion of the study area and all cropland in this part of the basin is currently assumed to use conventional tillage in the current iteration of the model.

Manure application can have important implications for nutrient accumulation and affect soil structure and hydrologic response. Manure application amounts are also available only at the county level, and the county level results are used to estimate the fraction of land receiving manure application in a given year. The acreage of cropland that receives manure application for counties in Minnesota and South Dakota was retrieved from the Agricultural Census (USDA, 2012) and area-weighted to model subbasins. Manure application is estimated to occur on approximately 8% of fields within the entire model area in a given year based on 2012 data. Manure application is assumed to occur primarily on conventionally tilled Cropland AB and Cropland Drained HRUs. These sub-divisions are accomplished as a post-processing exercise using county-level data and therefore are not spatially explicit.

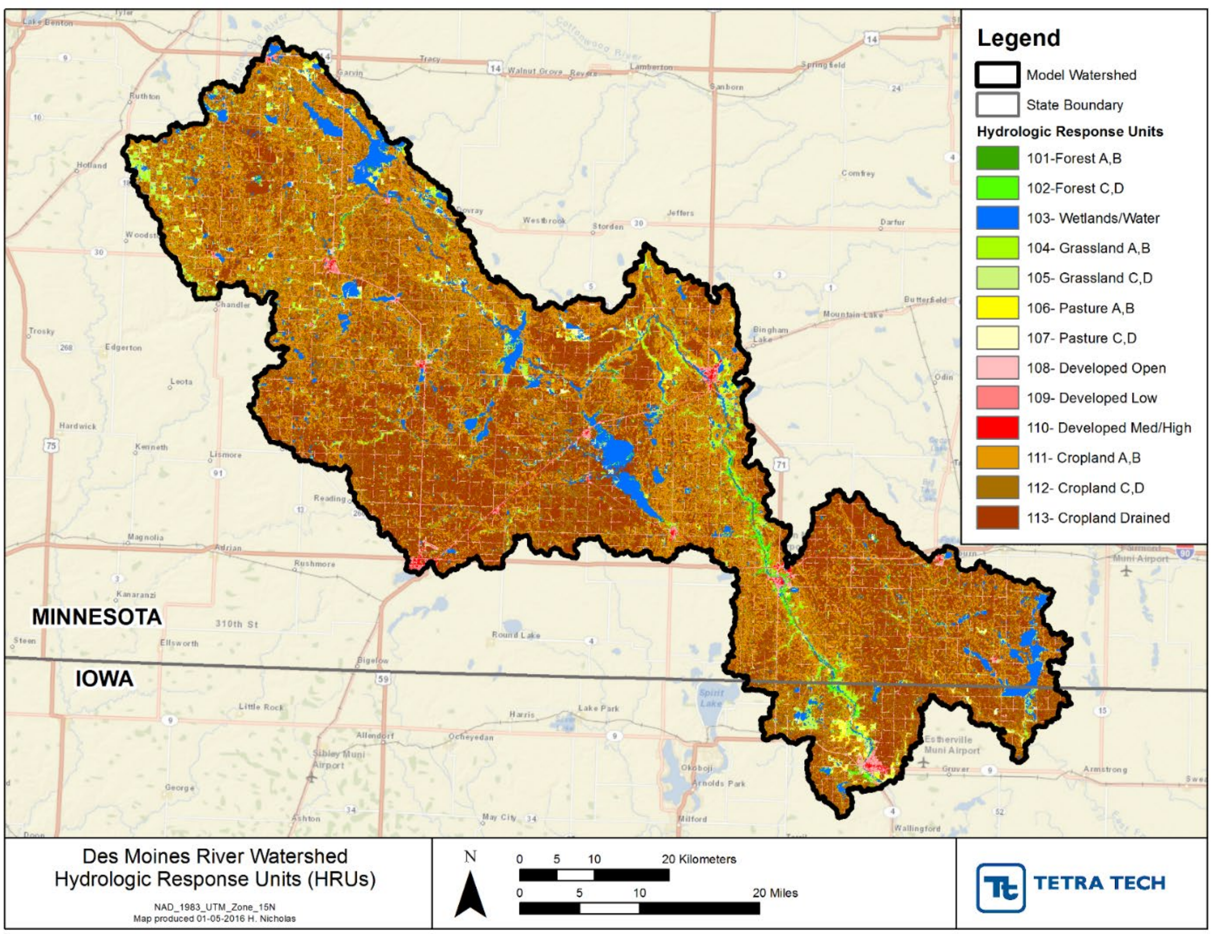

Figure 2-6. Hydrologic Response Units for the Des Moines River Watershed Model

Note: Cropland totals at the sub-basin level are further subdivided into areas in conventional vs. conservation tillage and areas receiving manure application during post-processing based on county-level statistics.

Table 2-3. Hydrologic Response Units for the Des Moines River Watershed Model

HRU Code	Description	HSG	Area (acres)	Percent Impervious	Data Source(s)
101	Forest - A, B	A, B	4,471	0%	NLCD Deciduous Forest, Evergreen Forest, Mixed Forest + gSSURGO HSG Overlay
102	Forest - C, D	C, D	8,085	0%	NLCD Deciduous Forest, Evergreen Forest, Mixed Forest + gSSURGO HSG Overlay
103	Wetlands/Water	-	60,353	0%	NLCD Emergent Wetlands, Woody Wetlands
104	Grassland - A, B	A, B	24,403	0%	NLCD Shrub/Scrub, Herbaceous, Barren, + gSSURGO HSG Overlay
105	Grassland - C, D	C, D	22,828	0%	NLCD Shrub/Scrub, Herbaceous, Barren, + gSSURGO HSG Overlay
106	Pasture - A, B	A, B	6,955	0%	NLCD Pasture + gSSURGO HSG Overlay
107	Pasture - C, D	C, D	9,099	0%	NLCD Pasture + gSSURGO HSG Overlay
108	Developed Open Space	-	52,451	1.6%	NLCD Developed Open Space
109	Developed Low	-	8,490	17.6%	NLCD Developed Low Density
110	Developed Med/High	-	3,936	56.0%	NLCD Developed Medium Density, High Density
111*	Cropland - A, B	A, B	280,567	0%	NLCD Cropland + gSSURGO HSG Overlay
112*	Cropland - C, D	C, D	298,011	0%	NLCD Cropland + gSSURGO HSG Overlay (D with slope >1%)
113*	Cropland - Drained	D	289,433	0%	NLCD Cropland + gSSURGO HSG Overlay + Slope Overlay (Cross listed soils, and D soils with slope <1%)

*Note that cropland HRUs are sub-divided based on tillage and manure application practices.

Source Key:

NLCD: Land use data developed by The Multi-Resolution Land Characteristics (MRLC) consortium from decadal Landsat satellite imagery and other supplementary datasets. < http://www.mrlc.gov/ >

SSURGO: Digital soils data produced and distributed by the Natural Resources Conservation Service (NRCS) - National Cartography and Geospatial Center (NCGC). < http://www.nrcs.usda.gov/wps/portal/nrcs/detail/soils/survey/?cid=nrcs142p2_053627 >

2.2 METEOROLOGY

Watershed responses are largely determined by meteorological inputs. Meteorological data required for an HSPF model consist of hourly precipitation (PREC), air temperature (ATEM), cloud cover (CLOU), dew point temperature (DEWP), solar radiation (SOLR), wind speed (WIND), and evapotranspiration (PEVT). MPCA has historically primarily relied on data available from the EPA-BASINS meteorological data set (USEPA, 2008) combined with local observed precipitation. However, the current version of the BASINS data extends only through 2009 necessitating analysis of newer data from the National Climatic Data Center (NCDC) and Minnesota State Climatologist for more recent periods, while significant QA work including patching missing observations is required for the local observer data. In addition, point-in-space monitoring records are often not representative of precipitation over a surrounding model area, especially during summer convective storms.

In recent years, several gridded meteorological products have been made available which have shown promise for water resources applications. Two such products were used for the development of the Des Moines River watershed model. North American Land Data Assimilation System (NLDAS; Xia et al., 2012) provides continuous and gridded hourly data from 1979 to present and consists of all the meteorological forcing parameters required for an HSPF application. NLDAS was generally used for the development of meteorological time-series for the watershed models except for precipitation. The spatial resolution of NLDAS is, however, relatively large (cell size approximately 12 km by 12 km) and may not represent spatial variation in precipitation over a small region well. As a result, another gridded dataset called PRISM (http://prism.oregonstate.edu) was used for the development of precipitation time-series. PRISM provides continuous daily precipitation data from 1980 to present at a spatial resolution of 4 km by 4 km. Daily PRISM precipitation data were disaggregated to an hourly time-step using NLDAS hourly precipitation data as template.

Based on discussions with the MPCA Project Manager after analysis of the available data sets, with the exception of precipitation the Des Moines River model has been constructed using NLDAS time-series. PRISM was used for the generation of precipitation time-series. Development of meteorological time-series is discussed in detail below.

2.2.1 Data Processing

A total of 53 NLDAS grid cells intersect the Des Moines River watershed. Hourly files for the continental US (CONUS) were downloaded and a Python script was developed to extract data for the grids intersecting the watershed.

NLDAS precipitation data were compared to rain-gauge records from BASINS meteorology stations in the watershed to ensure that they were in general agreement with each other. Comparisons were carried out for the following stations:

- MN214534 - Lake Wilson
- MN219033 - Windom
- MN214453 - Lakefield 2 NE
- MN217602 - Sherburn 3 WSW
- MN219170 - Worthington 2 NNE
- IA132724 - Estherville 2 N

An exact match is not expected, as totals at point gages can be affected by local convective storms and orographic effects. Monthly rainfall reported by NLDAS was generally in agreement with BASINS (Figure 2-7). The total rainfall reported by NLDAS from 1993 to 2009 was within 5% of that reported by BASINS stations. The difference was more than 10% for MN214534 and IA132724. Given these differences another gridded product PRISM (PRISM Climate Group) was evaluated for use in the watershed model.

The PRISM Climate Group produces daily precipitation data at a spatial resolution of approximately 4 km by 4 km from 1980 to the present for the CONUS. The PRISM dataset is primarily an interpolated product on point measurements with orographic correction. Figure 2-8 shows that the PRISM data generally correlates better with BASINS data. It was also found that the PRISM dataset generally represents the summer convective storm magnitudes better than the NLDAS dataset. PRISM precipitation is thus used in the watershed model. The differences in rainfall reported by PRISM and NLDAS are likely attributable to the differing interpolation techniques and spatial resolution used by the two products.

Other NLDAS meteorological parameters were also compared with the BASINS meteorological data and were generally found to be in agreement.

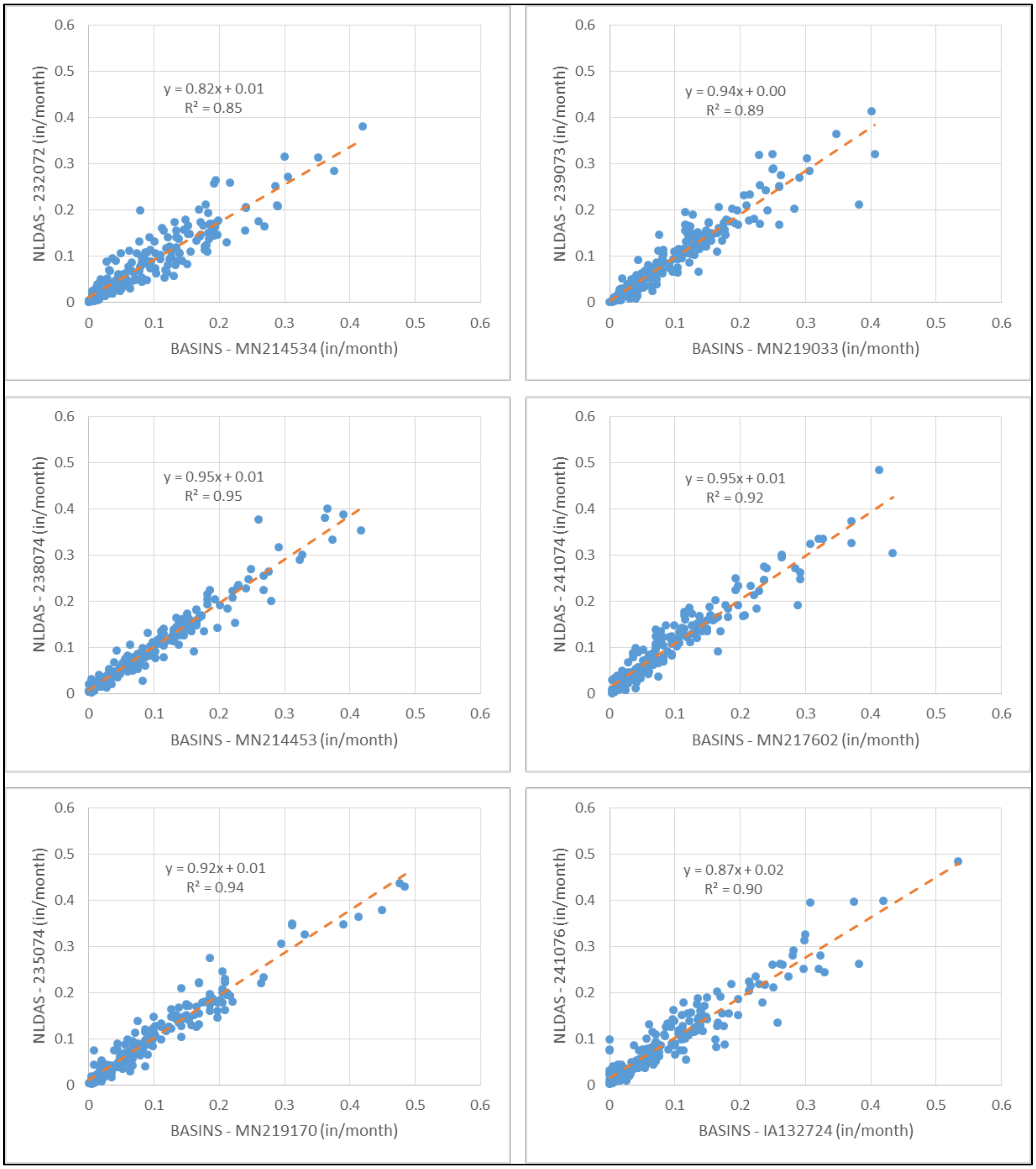

Figure 2-7. Comparison of BASINS and NLDAS Monthly Precipitation

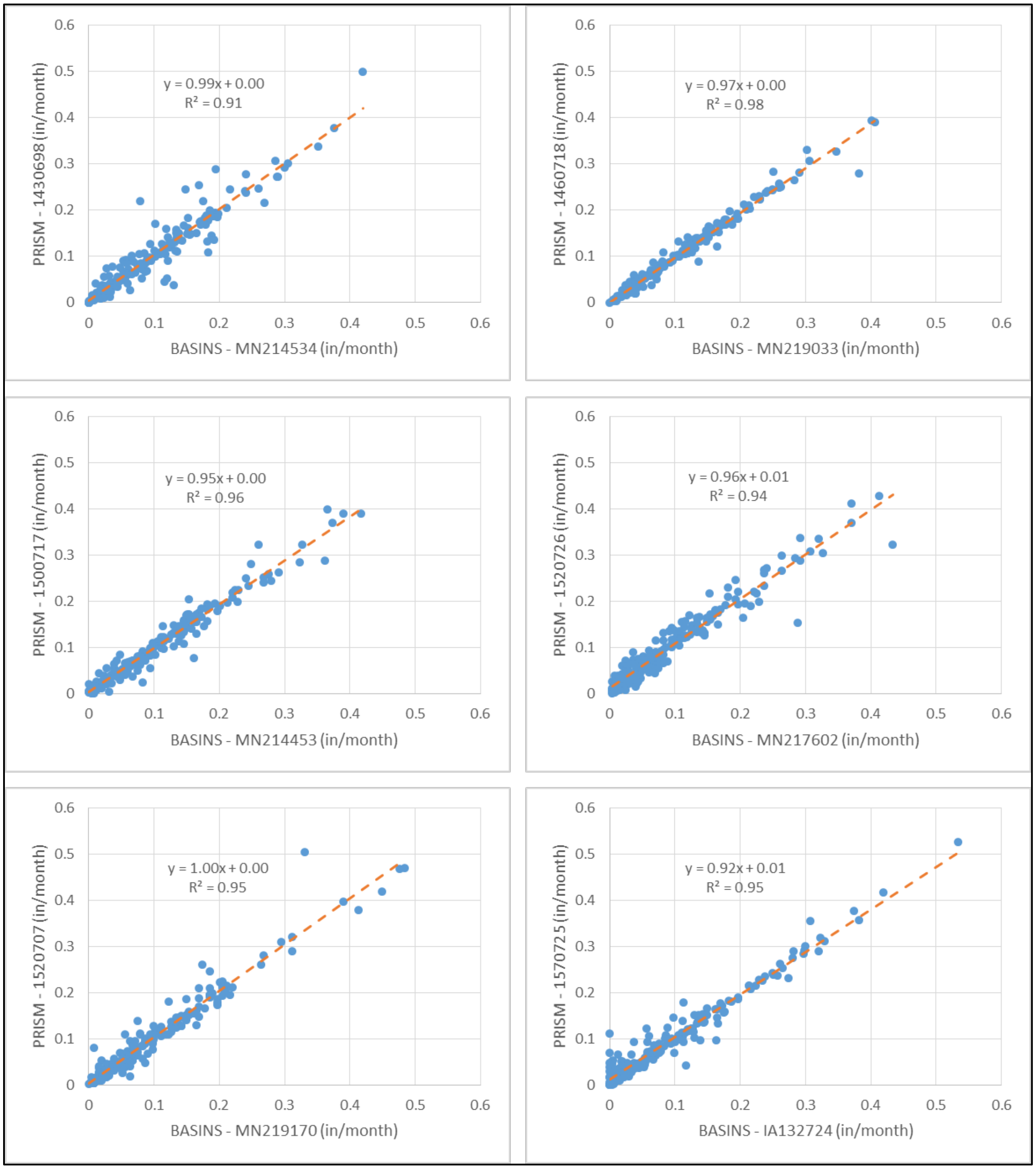

Figure 2-8. Comparison of BASINS and PRISM Monthly Precipitation

Each of the individual NLDAS or PRISM grid cells could be used to represent a weather station, but that would result in the number of HRUs exceeding the upper bound of 999 for an HSPF application. As a result, aggregation of the individual grid cells was required by regions of similar climate. The precipitation in the watershed did not exhibit a strong spatial pattern and the variation in total annual precipitation is also not large. Thus, meteorological data could be aggregated to a single weather region, but that approach would limit the ability to vary parameters by specific regions. As a result, weather regions in the watershed model were created to correspond to the boundaries of HUC10 watersheds. Figure 2-9 shows the weather regions for the Des Moines River watershed HSPF model

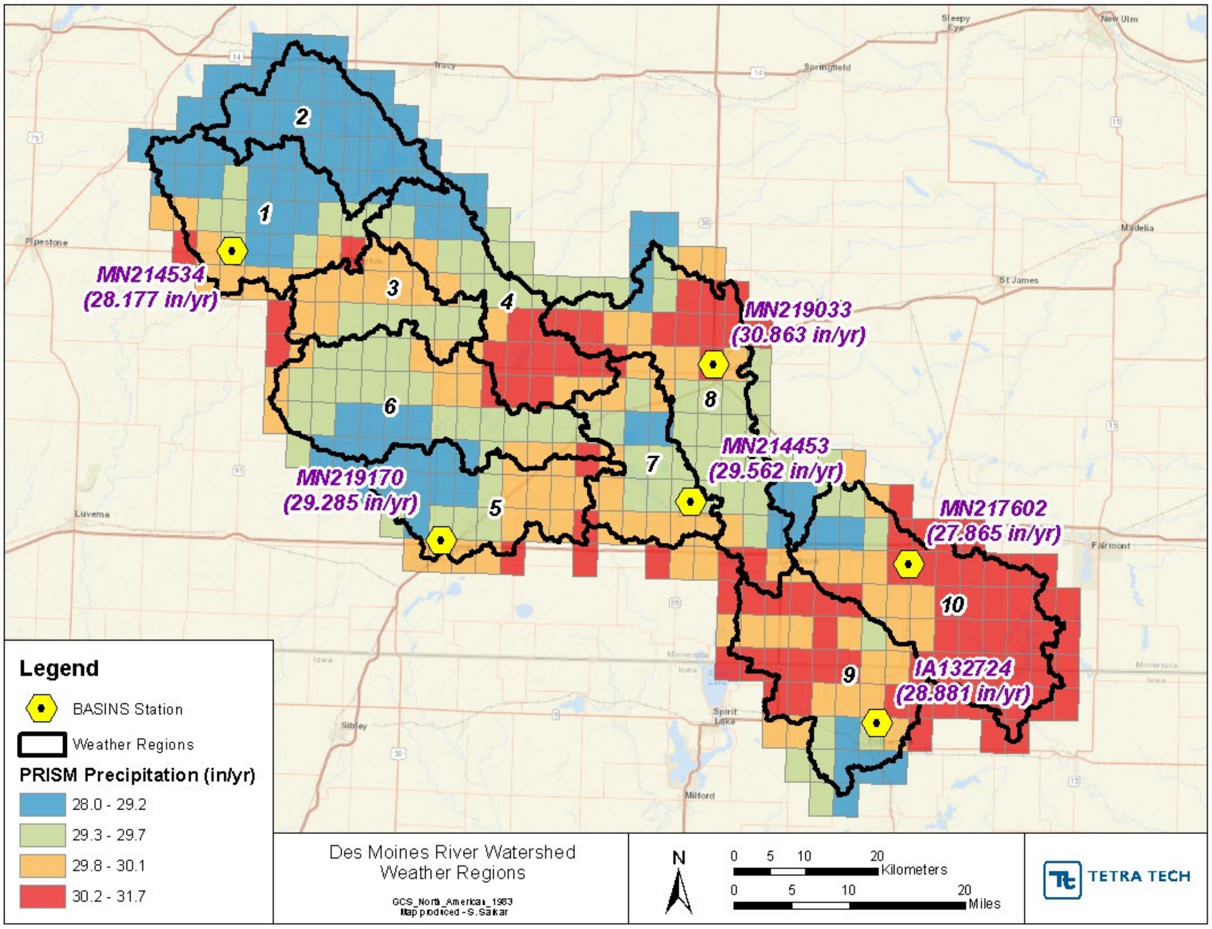

Figure 2-9. Weather Regions for the Des Moines River Watershed Model

2.2.2 Auxiliary Weather Series

NLDAS directly provides matched and consistent estimates of precipitation, air temperature, wind, and solar radiation. NLDAS also provides potential evapotranspiration (PET) calculated by a Penman energy balance method, although this is not directly used, as discussed below. Two variables required by HSPF – dewpoint temperature and cloud cover – are not directly available from NLDAS. These variables were calculated as follows:

Cloud cover is back calculated from the relationship of Davis (1997) describing the ratio of ambient solar radiation at the surface (E_{surf}) to radiation from a cloudless sky ($E_{cloudless}$):

$$\frac{E_{surf}}{E_{cloudless}} = 1 - 0.6740\, C^{2.854},$$

where, C is the fractional cloud cover. $E_{cloudless}$ is a function of latitude and time of year and is calculated with the WDMUtil tool distributed with BASINS.

NLDAS does not provide dewpoint temperature, but does provide specific humidity. We estimate dewpoint by the following method:

1. Calculate vapor pressure (e, mb) as a function of atmospheric pressure (p, mb) and specific humidity (q) from definition of q as a function of the mixing ratio, yielding

$$e = \frac{q\,p}{0.622 + 0.378\,q}$$

2. Use e to calculate dewpoint ($Td[C]$, °C) from e by solving the NOAA equation for e as a function of $Td[C]$:

$$Td[C] = Log_{10}\left(\frac{e}{6.11}\right) x \left[\frac{237.3}{7.5 - e/6.11}\right]$$

3. Convert to dewpoint in °F:

$$Td[F] = 32 + Td]x\,9/5$$

4. Ensure consistency with local daily air temperature data minimum (T_{min}, °F):

$$Td[F] = Max(T_{min}, Td[F])$$

Dewpoint temperature is used in the calculation of PET, so some small inaccuracies in daily PET may be introduced, although these should average out over high and low pressure weather cycles. Dewpoint temperature is also used for the calculation of the effective temperature at which precipitation becomes snow (SNOTMP = TSNOW + (AIRTMP - DEWTMP)*(0.12 + 0.008*AIRTMP)).

As noted above, NLDAS provides an estimate of PET calculated by the modified Penman method of Mahrt and Ek (1984). However, this is not a focus of NLDAS because NLDAS is designed to run a variety of Land Surface Models (LSMs; such as the NOAH model), most of which generate their own energy-based ET estimates. PET is provided only because one of the LSMs (SAC-SMA, the Sacramento soil moisture accounting model) does require it as an input (http://ldas.gsfc.nasa.gov/nldas/NLDAS2forcing.php; accessed 9/2/2015). On investigation it turns out that the PET that NLDAS reports is the PET calculated by the North American Regional Reanalysis (NARR) dataset (Mesinger *et al*., 2006). NARR is documented to have a large positive bias in the estimation of shortwave radiation (Xia *et al*., 2012). NLDAS corrects the NARR shortwave radiation estimates using satellite-based estimates, but the PET estimate ported from NARR is not corrected. In addition, NARR is at a coarser spatial scale than NLDAS and the PET estimates may be off in areas with strong edge effects.

Experiments conducted by Tetra Tech during development of other Minnesota HSPF models concluded that the NLDAS/NARR reported PET values were unreasonably high in some areas (due to the shortwave radiation bias) and exhibited too great a variation from the coastline to the interior (in part this is likely due to the downscaling of coarser-grid NARR data). Further, the PET time series provided by NLDAS did not match the seasonal pattern of Penman Pan ET calculated at individual weather stations.

Based on these observations it is desirable to recalculate PET, rather than using the PET reported by NLDAS/NARR. We therefore calculated Penman Pan PET using inputs from NLDAS (including the corrected shortwave radiation) and applying the standard approach from BASINS that has been implemented in most other Minnesota HSPF models. The Penman Pan ET calculated in this way does provide a reasonable match to the individual weather station results.

2.3 Model Segmentation and Reach Network

2.3.1 Subwatershed Delineation

12-digit Hydrologic Unit Code (HUC12) boundaries and associated stream centerlines (at 1:24,000 scale) from the Minnesota Department of Natural Resources (MNDNR) were used as the base layers for delineation. Cuts were made to the existing HUC12 boundaries at the locations of flow and water quality monitoring stations with data for the simulation time-period (1993-2014). Additional cuts were made for impaired reach segments and waterbodies (shown in Figure 2-10), and for certain lakes explicitly represented in the model. Criteria used to choose lakes for explicit representation in the model are discussed in Section 2.3.2.

Sub-delineations of the HUC12 boundaries were carried out in ArcGIS using NHDPlus version 2 catchments boundaries (Horizon Systems Corporation) as guide. The subbasins delineated for the Des Moines HSPF model are shown in Figure 2-11.

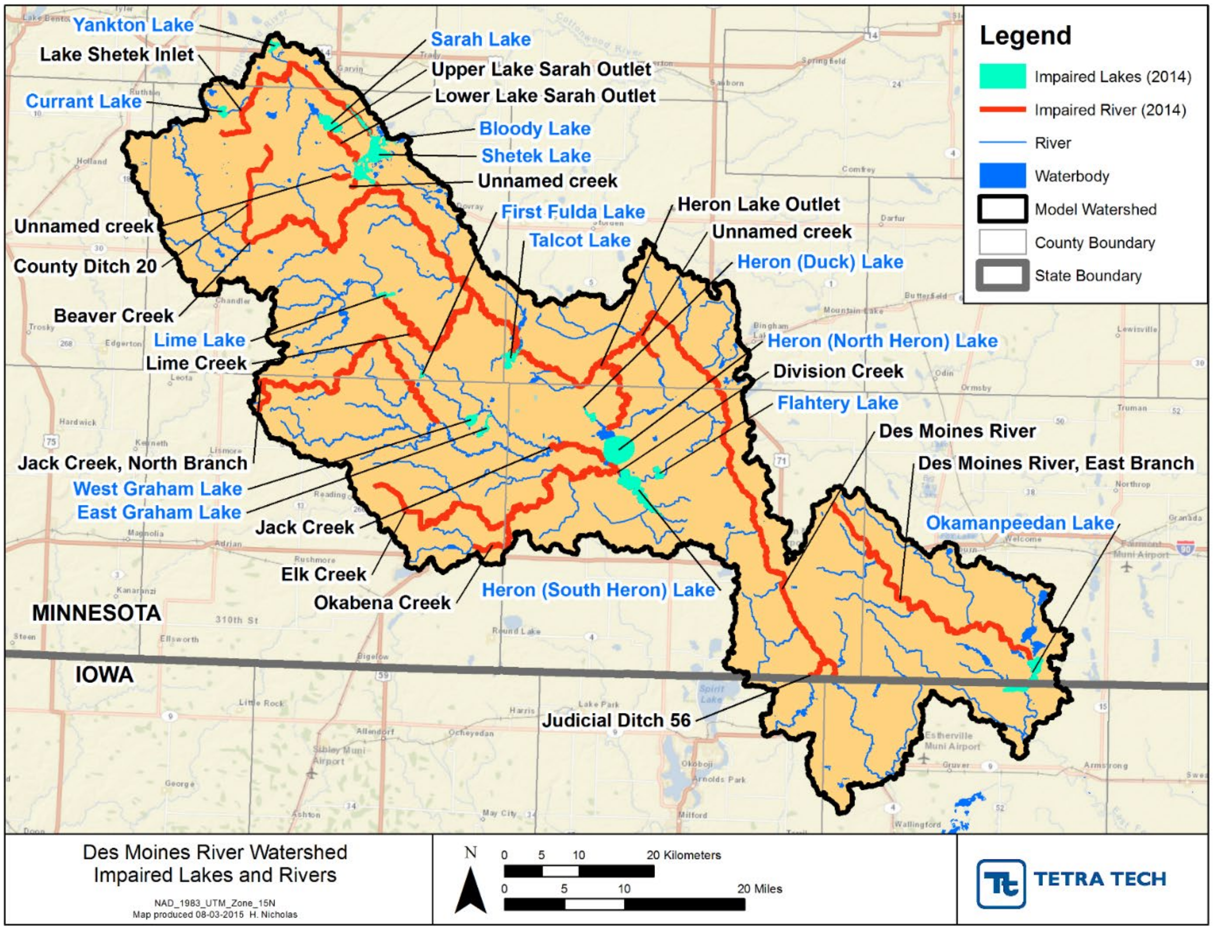

Figure 2-10. Impaired Stream Segments and Lakes in the Des Moines River Watershed Model

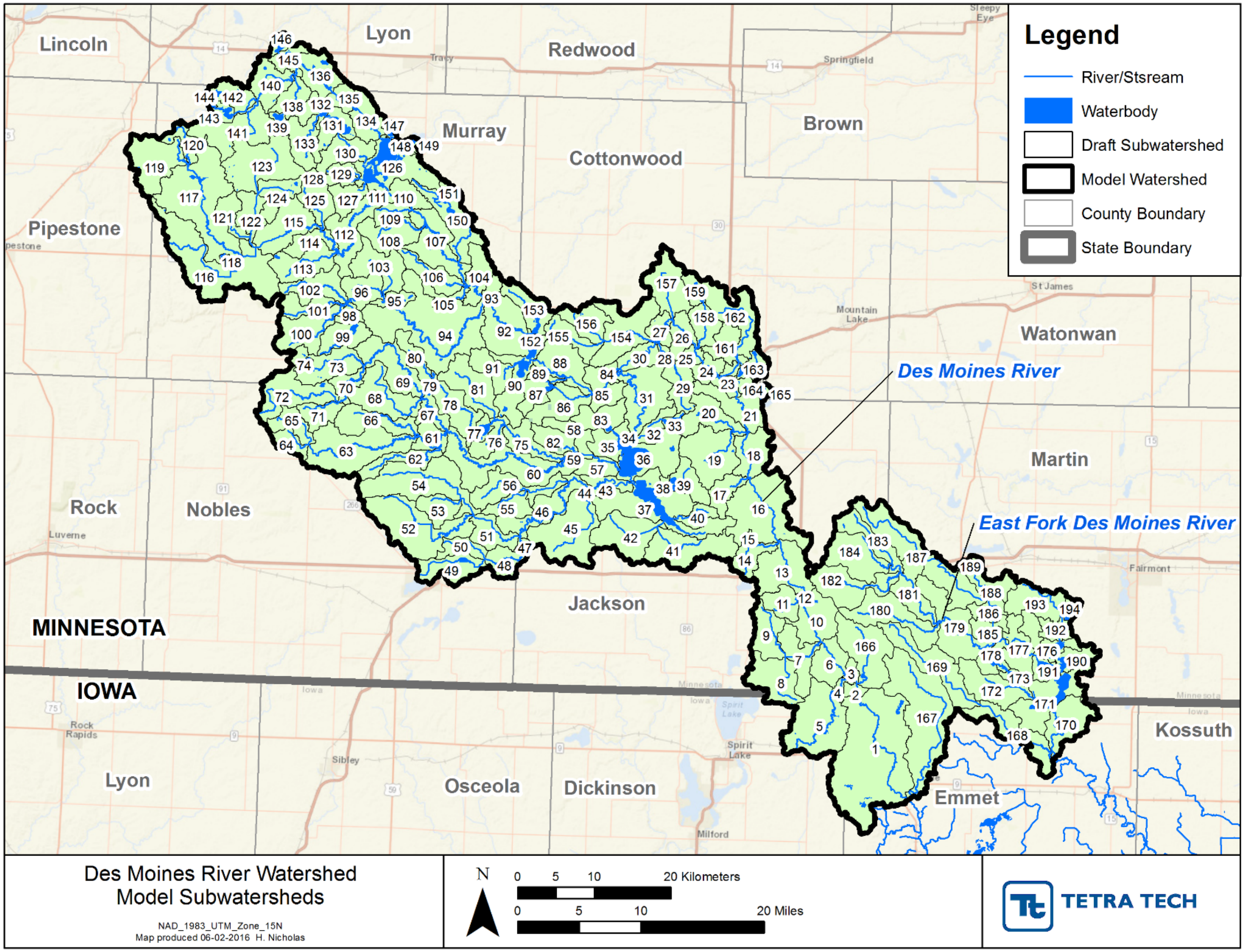

Figure 2-11. Delineated Subbasins for the Des Moines River Watershed Model

2.3.2 Stream Reach and Lake Delineation

HSPF represents a single main channel (reach) or lake waterbody for each model subbasin. These reaches carry the same identifying number as the subbasin. Lower order tributaries at finer spatial scales are not explicitly represented in the model and are accounted for implicitly in the upland simulation.

The study area contains lakes of varying sizes, many of which do not have available bathymetric data. It was not feasible to represent every lake as an explicit lake segment in the model. (Those that are not will be represented as a water/wetland land use). Selection of lakes for explicit representation followed the general procedure outlined in AQUA TERRA (2012).

The process began with the 2012 MPCA Assessment lakes (24 lakes). NHD was then queried for lakes greater than 200 acres, which added another 9 lakes not assessed by MPCA. MPCA bathymetry data provided information for another 3 lakes, which were less than 200 acres, for a total of 36 lakes.

Lakes were first screened as to whether they are located in-line on a HUC12-scale stream reach. The area cumulative distribution of 14 inline lakes was plotted (Figure 2-12). This distribution shows an approximate inflection point of 519 acres. AQUA TERRA (2012) suggests using this inflection point as a cutoff value for selecting lakes for explicit representation. Eight lakes met the acreage criteria.

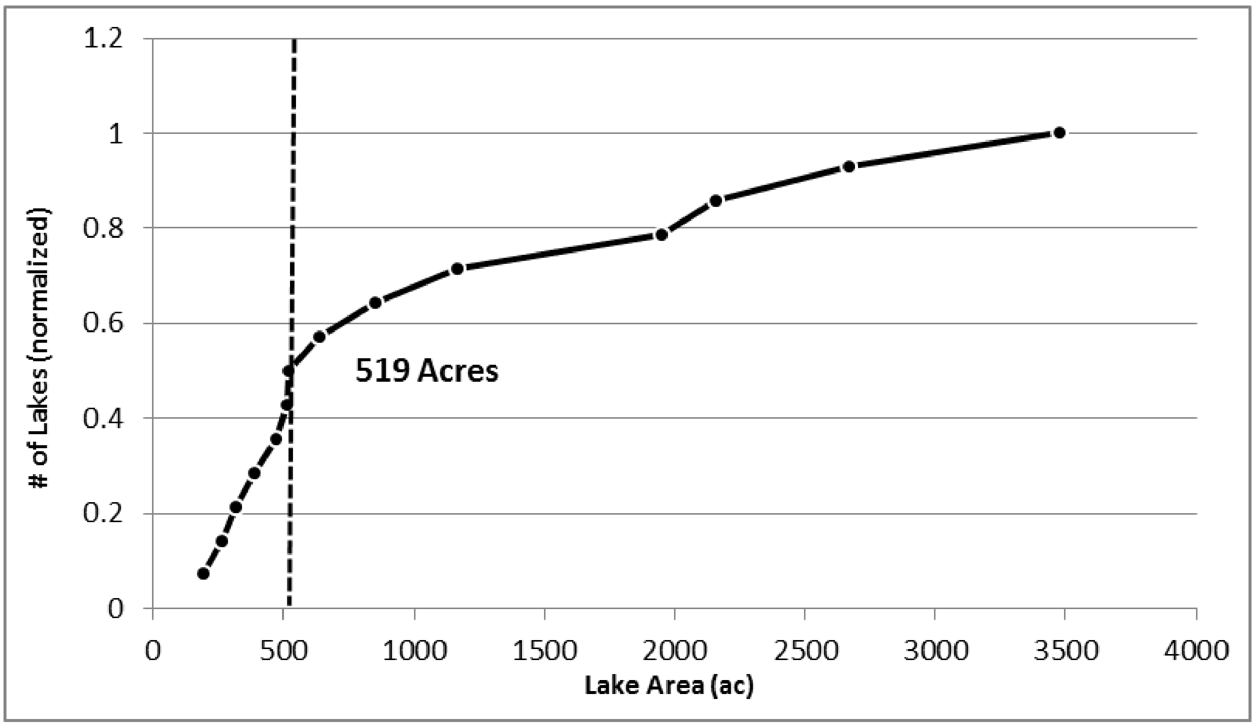

Figure 2-12. Cumulative Distribution of Surface Area of In-line Lakes in the Minnesota Portion of the Des Moines River Watershed

There are 22 lakes not in-line with HUC12 scale stream reaches. Many of these lakes are small and the distribution does not have a clear point of inflection. The size criterion adopted for in-line lakes (519 acres) was thus also applied to this set of lakes (none were larger than 519 acres). Eight lakes with nutrient impairments are modeled explicitly in addition to the 8 lakes identified in the inflection/inline analysis. Six additional lakes have bathymetry data and were added to the set for representation in the model. This methodology resulted in the selection of a total of 22 lakes for explicit simulation in the model (Table 2-4 and Figure 2-13).

Table 2-4. Lakes Represented Explicitly in the Des Moines River Watershed Model

Name	Assessed by MPCA in 2012	Impaired in 2014	Lake area (acres)	Bathymetry
Bloody	Yes	Yes	261.1	Yes
Bright	Yes	No	638.5	Yes (digitized from PDF)
Clear	Yes	No	260.8	Yes
Cottonwood	Yes	No	154.0	Yes
Currant	Yes	Yes	390.9	Yes (digitized from PDF)
East Graham	Yes	Yes	469.1	Yes (digitized from PDF)
First Fulda*	Yes	Yes	118.9	Yes (digitized from PDF)
Flahtery (Flaherty)	Yes	Yes	417.0	Yes
Fox	No	Not assessed	172.1	Yes
Heron (Duck)	Yes	Yes	188.7	Yes (digitized from PDF)
Heron (North)	Yes	Yes	3,204.4	Yes (digitized from PDF)
Heron (South)	Yes	Yes	2,670.2	Yes (digitized from PDF)
Kinbrae	No	Not assessed	98.4	Yes
Lime	Yes	Yes	318.1	Yes (digitized from PDF)
Okamanpeedan	Yes	Yes	2,159.8	Yes (digitized from PDF)
Sarah	Yes	Yes	1,164.2	Yes
Shetek	Yes	Yes	3,477.3	Yes
Summit (Cottonwood County)	No	Not assessed	64.1	Yes
Summit (Murray County)	Yes	No	74.0	Yes
Talcot (Talcott)	Yes	Yes	844.0	Yes
West Graham	Yes	Yes	519.1	Yes
Yankton	Yes	Yes	396.4	Yes

* Second Fulda storage is included in the stage-discharge relationship of First Fulda Lake.

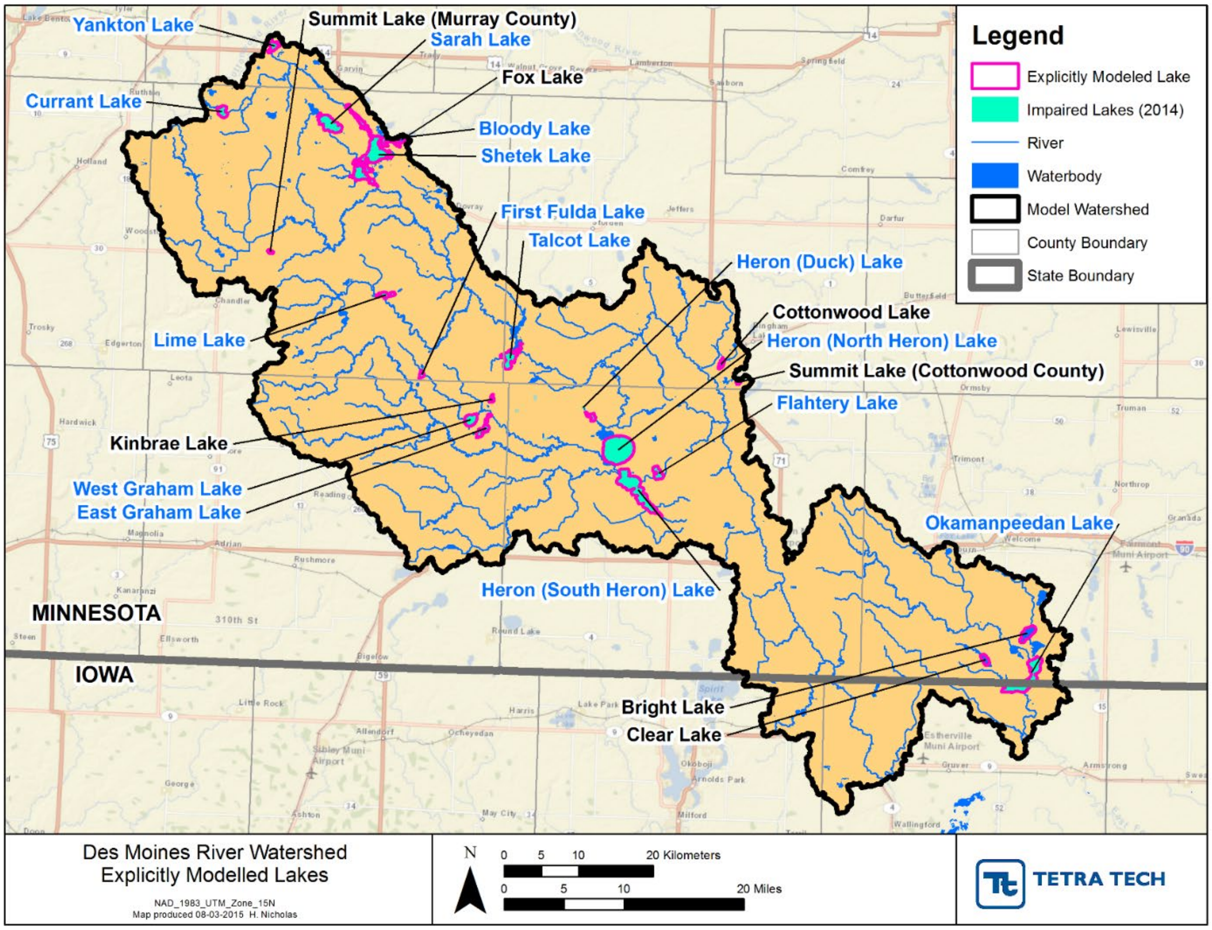

Figure 2-13. Lakes Represented Explicitly in the Des Moines River Watershed Model

2.3.3 Representation of Groundwater-Surface Water Interactions

A USGS study in southwest Minnesota (Adolphson 1983) shows that West Fork Des Moines River and its tributaries are underlain by a highly productive alluvial aquifer with thickness ranging up to 100 feet (Figure 2-14). The alluvial aquifer is located in sand and gravel deposited in glacial outwash channels. Recharge to this aquifer occurs from precipitation, seepage from streams, and groundwater inflow from adjacent areas while discharge occurs via wells and springs, seepage to streams, and groundwater outflow (Adolphson 1983; Cowdery 2005).

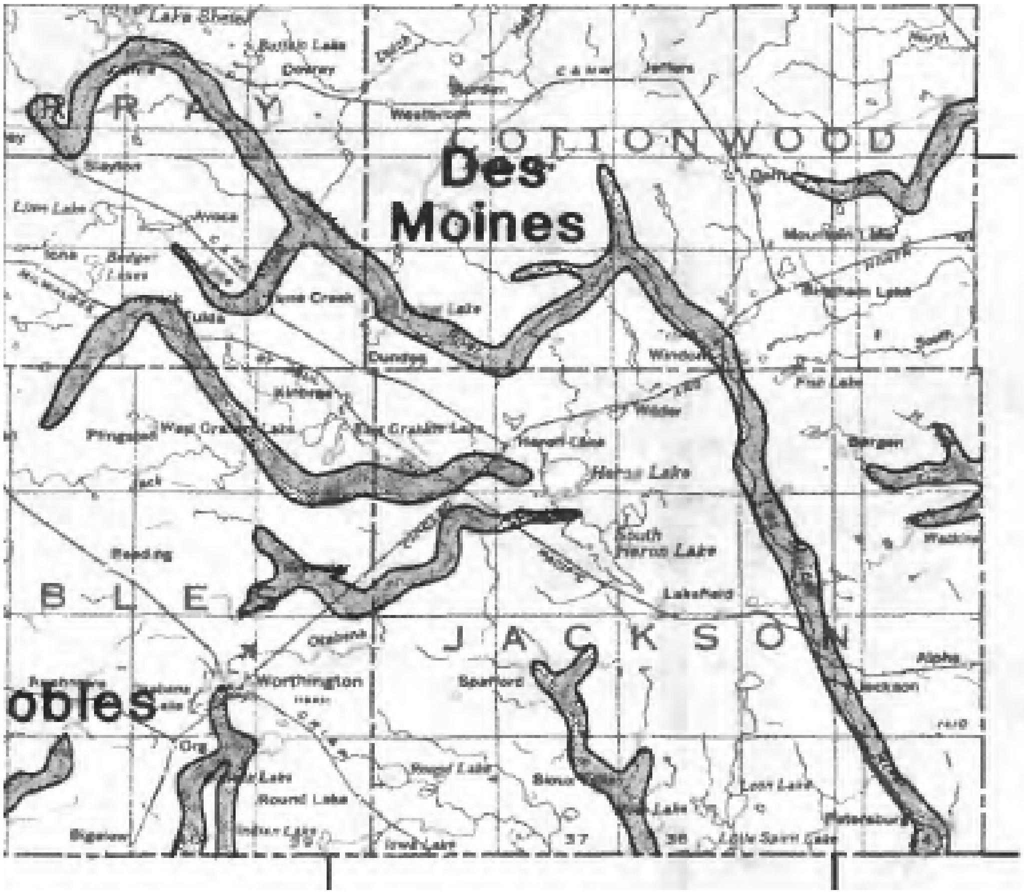

Figure 2-14. Surficial Aquifers in the West Fork Des Moines River Watershed

(Detail extracted from Figure 4 in Adolphson, 1983)

Examination of the observed streamflow at monitoring gages in the watershed shows that exchanges with the alluvial aquifer has significant impacts on streamflow in the West Fork Des Moines River and its tributaries. While the representation of detailed groundwater-surface water interactions is not possible within the framework of the HSPF model, an approximate representation is viable using a network of groundwater reaches.

In the current model setup, a groundwater reach has been assigned to stream reaches that overlay the alluvial aquifer. A groundwater reach receives recharge from all surface reaches underlain by the alluvial aquifer and discharges to the next downstream groundwater reach (representing groundwater outflow) and to downstream surface reaches (representing seepage into streams). Groundwater reaches do not receive meteorological inputs.

Each surface reach in the model that overlies the alluvial aquifer has been assigned two outflows. The first outflow goes to the next downstream surface reach (which is generally the standard setup of an HSPF model) while the second outflow goes to the groundwater reach representing recharge to the alluvial aquifer. The second outflow to the aquifer is small (less than 10 cfs) and was varied as a calibration parameter.

The surficial, alluvial aquifer overlies a deeper, bedrock Cretaceous aquifer located in discontinuous, basal sandstone beds in the Dakota Formation and the overlying Codell Sandstone Member of the Carlile Shale of the Colorado Group (Woodward and Anderson, 1986). Permanent loss of flow to deep groundwater is simulated as a fraction of the flow in the alluvial aquifer where it intersects the Cretaceous aquifer in south-east Cottonwood County and Jackson County. The simulation of losses to the Cretaceous aquifer is generally in agreement with Woodward and Anderson (1986) who state that

> *"Water in the Cretaceous aquifer is confined by overlying shale and by overlying till as much as 700 feet thick. Locally where the drift is permeable and thin and where the shale is missing, water in the aquifer is unconfined. Groundwater moves away from the Sioux Quartzite Ridge north toward the Minnesota River, south toward Iowa, and eastward toward the Mississippi River. Recharge to the aquifer primarily is by infiltration of precipitation that percolates through the overlying drift and underflow in the aquifer from South Dakota."*

2.3.4 Reach Hydraulics

Movement of sediment in stream networks, including transport, scour, and deposition rates, is determined by flow energy. HSPF does not directly solve hydraulic momentum equations for flow routing, but rather specifies information on the relationship between stage, discharge, and geometry through Functional Tables (FTables). The calculation of boundary shear stress from the FTable information is a key component of the simulation of sediment transport.

HSPF is a water balance (hydrologic) model and not a hydraulic model. HSPF represents stream reaches as one-dimensional fully mixed reactors and, while maintaining mass balance, does not explicitly conserve momentum. To simulate the details of hydrograph response to storm events HSPF relies on Function Tables (FTables) that describe the relationship of reach discharge, depth, and surface area to storage volume. At stable median flow conditions the model results are not particularly sensitive to the details of the FTable specification, as outflow tends to approximate the net inflows; however, the shape of the response to storm event peaks can be highly sensitive to FTable details. Given the interest of MNDNR in evaluating the distribution of flows in streams in Minnesota there is an increasing need to refine HSPF basin-scale model FTables.

By default, the BASINS version of HSPF estimates FTables by applying predetermined regressions against drainage area, but this approach does not take into account site-specific characteristics (such as obstructions) and is based on data from sites in ecoregions different from those found in Minnesota.

The optimal approach for hydraulics in HSPF is to incorporate information from a detailed hydraulic model, such as HEC-RAS, but such models are generally not available for the Des Moines River watersheds and creating such models is not part of the scope for this task. The current model has been developed using the default approach for FTables provided with BASINS, which estimates FTables by applying predetermined regressions against drainage area. This approach could be refined through analysis of flow gage rating curves, cross section measurements, and information on bridges and culverts. Such refinements would be expected to improve the high flow simulation as FTables primarily affect the details of the hydrograph shape.

2.3.5 Lake Storage and Outflow

Lakes and reservoirs typically have outflows that are determined by dam/weir characteristics or active management. Thus, lake FTables represent a different class of analyses than stream reach FTables, and essentially need to be addressed on a site-specific basis as a first priority. Site-specific FTables are calculated for lakes. These are based on specific characteristics of individual lakes/dams and take precedence over any other methods for creating FTables.

Where available, lake bathymetric data were used to characterize stage-storage relationships based on the elevation contour polylines contained in each dataset (MNDNR, 2002). Maximum or average depth data were obtained for the remaining lakes that lack bathymetry. The maximum depths (or inferred maximum depth consistent with the average depth) were used to estimate the lake volume at various stages based on the assumption that depth is a function of distance from the lake's shoreline. This was done by converting lake polygons to raster format and using the following linear transformation (Hollister and Milstead, 2010):

$$Z = D*Z_{max} / D_{max}$$

where Z is the depth for any given raster cell; D is the Euclidean distance from the shoreline, including islands; Z_{max} is the measured maximum depth for a given lake; and D_{max} is the maximum distance from the shoreline of a given lake. The lake depth raster dataset was then summed to calculate the lake volume:

$$Lake\ volume\ = \sum_{i=1,j=1}^{n} Cell\ area * Depth\ i,j$$

Storage volumes above the lake surface outlet level were estimated using LiDAR data. The outflows associated with various water depths were estimated using a rectangular weir equation. Lakes with natural outlets were approximated using a broad-crested weir assumption. The dimensions of the weir were determined from details provided by MNDNR hydrologists or, lacking direct information, from examination of aerial imagery.

2.4 Point Source Discharges

Permitted point sources are present in the Des Moines River watersheds and were investigated for inclusion in the HSPF model. A variety of municipal and industrial sources discharge to surface waters of the study area. The majority of the permits are for stabilization ponds that discharge small flows on a seasonal or intermittent basis. There are three major dischargers in the watershed. The Worthington Industrial WWTP and Worthington WWTP discharge to the Okabena Creek while the Windom WWTP discharges to the West Fork Des Moines River. The remaining dischargers are considered minor.

Permitted point sources are listed in Table 2-5 . Those without surface discharges are shown in shaded text at the bottom of the table. Several minor industrial discharges with minimal flow were not included in the model; those that are included are mapped in Figure 2-15.

Table 2-5. Permitted Point Source Discharges in the Des Moines River Watershed

NPDES ID	Name	Type	Model Reach	Average Discharge (MGD)
MNG640102	Alpha WTP Industrial	Minor Industrial	Not in model	0.001
MNG580165	Avoca & Iona WWTP	Minor WWTP Pond	94	Intermittent
MN0021750	Brewster WWTP	Minor WWTP Pond	46	Intermittent
MNG580006	Ceylon WWTP	Minor WWTP Pond	172	Intermittent
MN0064645	Country Pride Services	Minor Industrial	Not in model	No flow data
MNG580221	Currie WWTP	Minor WWTP Pond	110	Intermittent
IA3215001	Dolliver STP	Minor Industrial	Not in model	Cooling water, Iowa
MN0056103	Dunnell WWTP	Minor WWTP	169	0.07

NPDES ID	Name	Type	Model Reach	Average Discharge (MGD)
MNG580188	Fulda WWTP	Minor WWTP Pond	94	Intermittent
IA3000105	Green Plains Superior LLC	Minor Industrial	Not in model	Minimal flow
MN0067385	Heron Lake BioEnergy LLC	Minor Industrial	31	0.18
MNG580189	Heron Lake WWTP	Minor WWTP Pond	35	Intermittent
MN0033375	Hubbard Feeds Inc - Worthington	Minor Industrial	49	<0.01
MNG580063	Jackson WWTP	Minor WWTP Pond	10	Intermittent
MNG580061	Lake Wilson WWTP	Minor WWTP Pond	117	Intermittent
MN0020427	Lakefield WWTP	Minor WWTP	37	0.22
MNG790110	Magellan Pipeline Co - Albert Lea	Minor Industrial	Not in model	No flow data
MN0033693	MDNR Lake Shetek State Park	Minor WWTP Pond	Not in model	No recorded discharge
MN0050288	Okabena WWTP	Minor WWTP	43	
MNG640077	Red Rock Rural WS - Windom WTP No 1	Minor Industrial	Not in model	0.02
MN0024872	Sherburn WWTP	Minor WWTP	188	0.14
MNG580191	Slayton WWTP	Minor WWTP Pond	112	Intermittent
MN0022217	**Windom WWTP**	**Major WWTP**	**22**	**1.05**
MN0031178	**Worthington Industrial WWTP**	**Major WWTP**	**49**	**1.92**
MN0031186	**Worthington WWTP**	**Major WWTP**	**49**	**1.79**
MN0063398	*Beecks Gravel & Excavating Inc*		*Minor (no surface discharge)*	
MNG490274	*Buffalo Ridge Concrete Inc - Edgerton*		*Minor (no surface discharge)*	
MNG490249	*Consolidated Ready Mix - Windom div of GCC*		*Minor (no surface discharge)*	
MNG490046	*Duininck Bros Inc - Aggregate*		*Minor (no surface discharge)*	
MN0070271	*Dundee WWTP*		*Minor (no surface discharge)*	
MNG580101	*Garvin WWTP*		*Minor (no surface discharge)*	
MNG490295	*Hansen Concrete Co*		*Minor (no surface discharge)*	
MNG490003	*Knife River Central Minnesota*		*Minor (no surface discharge)*	
MNG490019	*McLaughlin & Schulz Inc*		*Minor (no surface discharge)*	
MNG490131	*OMG Midwest Inc/Southern MN Construction Co Inc*		*Minor (no surface discharge)*	
MN0067482	*PM Beef Holdings LLC Windom*		*Minor (no surface discharge)*	

NPDES ID	Name	Type	Model Reach	Average Discharge (MGD)
MNG490093	*RA Muecke Sand & Gravel Inc*		*Minor (no surface discharge)*	
MNG820031	*Red Rock Rural WS - Lake Augusta WTP*		*Minor (no surface discharge)*	
MNG490103	*Rupp Construction Co Inc*		*Minor (no surface discharge)*	
MNG490079	*Sweetman Sand & Gravel Inc*		*Minor (no surface discharge)*	
MNG120045	*The Toro Co - Windom*		*Minor (no surface discharge)*	
MNG580200	*Wilmont WWTP*		*Minor (no surface discharge)*	

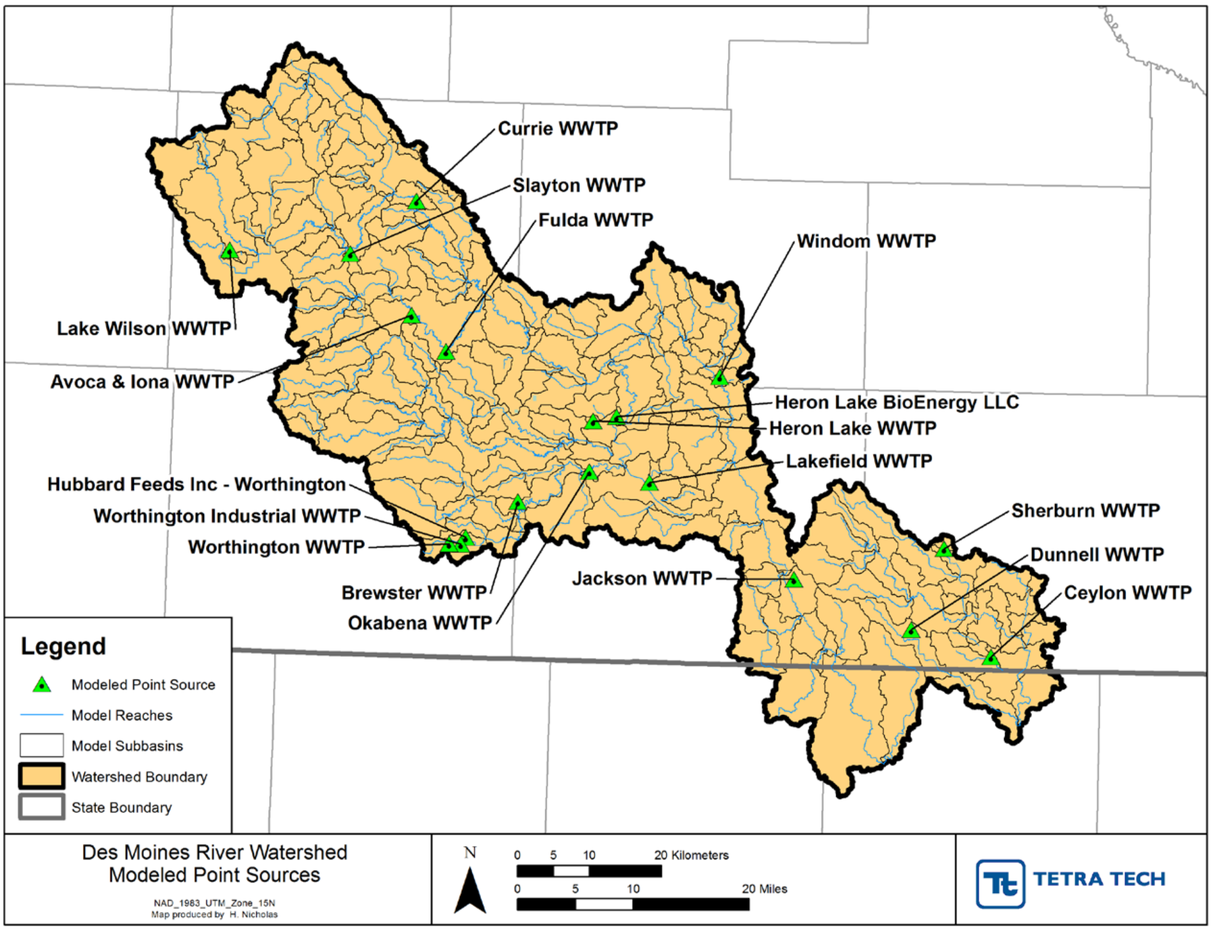

Figure 2-15. Permitted Point Source Discharges in the Des Moines Watershed Model

3 Model Calibration and Validation Approach

3.1 Flow and Water Quality Data

3.1.1 Flow Gaging Data

There are several MNDNR and USGS gages with daily flow records in the Des Moines River watershed for the model simulation period (1/1/1993 - 12/31/2014). Flow data for each gage operated by MNDNR or MPCA were obtained from HYDSTRA; data for gages operated by USGS were downloaded from the NWIS system. The periods of record for selected gages are shown in Table 3-1 and the locations are shown in Figure 3-1.

Table 3-1. Flow Gages in the Des Moines River Watershed Model Domain

USGS ID	HYDSTRA ID	Name	Begin	End	Model Reach
05476000	51107001	Des Moines River at Jackson, MN	6/1/1909	10/8/2015	13
05476500	(see note)	Des Moines River at Estherville, IA	10/1/1951	5/11/1995	1
			1/1/2004	9/30/2014	
05474975	51092001	Jack Creek near Heron Lake	4/1/2003	10/8/2014	58
05474915	51093001	Okabena Creek near Okabena	4/11/2003	8/10/2014	44
05474990	51017001	Heron Lake Outlet near Heron Lake	4/9/2003	10/8/2014	34
	51011001	West Fork Des Moines River near Windom	4/1/2003	10/1/2005	23
	51021001	West Fork Des Moines River near Heron Lake	3/26/2003	10/1/2005	85
	51026001	West Fork Des Moines River near Avoca, CSAH 6	3/22/2013	10/8/2015	92
	51065001	West Fork Des Moines River near Avoca, CSAH 7	3/26/2003	10/1/2005	107
	51055001	Lime Creek near Lime Creek	4/1/2003	10/1/2005	94
	51078001	Lake Shetek Outlet near Currie	4/1/2003	10/1/2005	126
	51069001	Beaver Creek near Currie	10/1/2001	11/2/2008	111
	53008001	Martin County Ditch 11 near Dunnell	3/28/2009	10/24/2010	186
05476900	53014001	Fourmile Creek near Dunnell	3/28/2009	10/24/2010	180

Note: USGS gaging for Des Moines River at Estherville, IA ended in 1995. Subsequent records have been maintained by the U.S. Army Corps of Engineers.

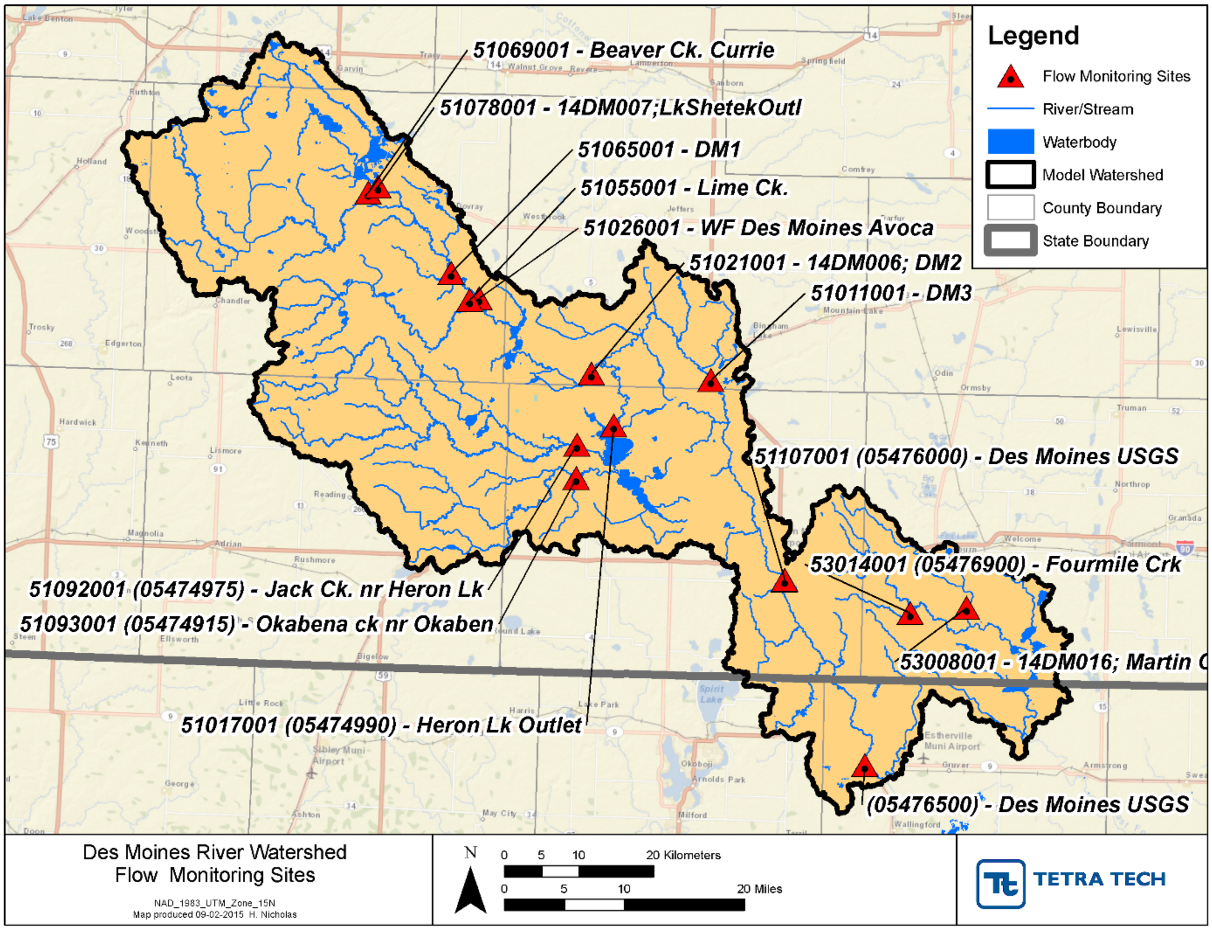

Figure 3-1. Flow Gages in the Des Moines River Watershed Model Domain

3.1.2 Water Quality Data

Water quality data have been collected at many locations within the Des Moines River watershed. Most of these data are available in EQUIS, and MPCA provided a full download of all stations. Despite the volume of data, stations that have collected significant amounts of nutrient data over a time period coincident with the model simulation period are few and an even smaller number are at or near flow gaging stations. The model segmentation was designed to line up with available flow gage locations and monitoring sites known to have large amounts of water quality data; however, some stations with small to moderate amounts of monitoring data were not usable for calibration because a major tributary or point source discharge enters the model segment between the monitoring station and the downstream end of the segment, or because they were on tributaries or lakes that were too small for explicit inclusion in the basin-scale models. In other cases, multiple closely located EQUIS stations were combined for use in model calibration.

Ultimately, 10 locations (represented by 12 EQUIS stations) were selected as primary model calibration locations in the Des Moines River watershed. These locations are summarized in Table 3-2 and displayed in Figure 3-2. For West Fork Des Moines River and its tributaries, most of these stations are affected to some extent by large lakes. Okabena Creek and Beaver Creek are two locations that are exceptions. There are however two major point source dischargers in the Okabena Creek watershed that have significant impacts on the instream nutrient concentration observed at the EQUIS gage. The tributaries in the East Fork Des Moines River are generally not impacted by lakes or large point sources and most of the larger lakes on the East Fork are downstream of monitoring sites.

Table 3-2. Water Quality Calibration Locations for the Des Moines River Watershed Model

Location	Model Reach	EQUIS Station(s)
Beaver Creek near Currie	111	S002-005
Lake Shetek Outlet near Currie	126	S002-006
West Fork Des Moines River at Avoca	107	S002-008
Jack Creek near Heron Lake	58	S001-557
Okabena Creek near Okabena	44	S001-568
Heron Lake Outlet	34	S002-009
West Fork Des Moines River near Windom	23	S000-894
West Fork Des Moines River near Jackson	13	S000-297,S004-359,S005-936
Martin County Ditch near Dunnell	186	S005-572
Fourmile Creek near Dunnell	180	S005-027

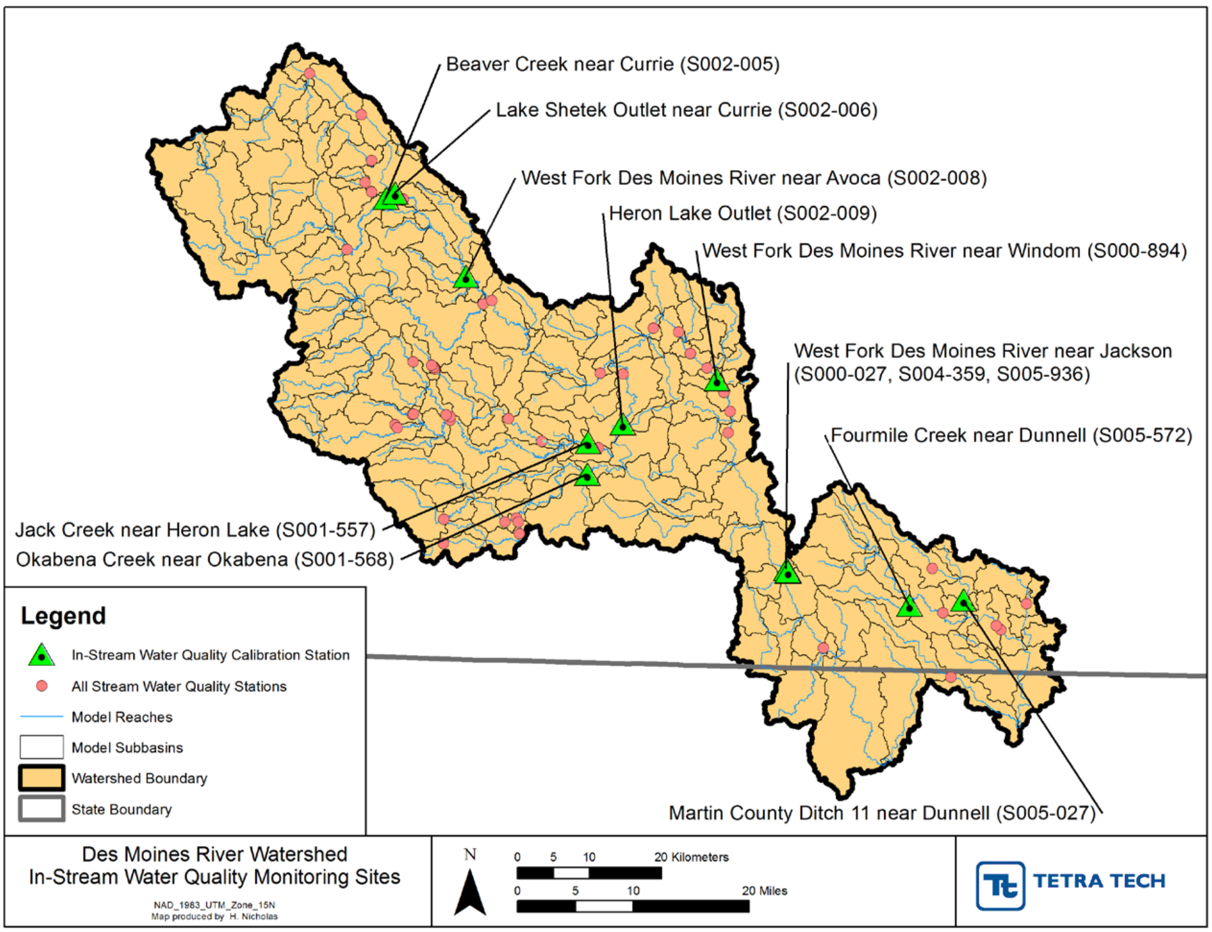

Figure 3-2. Water Quality Monitoring Locations used for the Des Moines River Watershed Model

The West Fork Des Moines River watershed has several large eutrophic lakes that affect downstream water quality and complicate the modeling effort. Most of these lakes have reported growing season chlorophyll *a* concentrations exceeding 100 µg/L. It is important to note that monitoring for solids in the watershed is for total suspended solids (TSS), which is a measure of all filterable solids, including both inorganic sediment and organic solids. HSPF predicts suspended sediment that represents inorganic sediment (e.g., sand, silt, and clay) and can also implicitly represent organic detritus derived from the land surface or channel erosion, but does not account for organic solids produced within the water column by algal growth. In most watersheds this is a small fraction of total suspended solids and can safely be ignored in the sediment calibration. This is not the case for the high algal densities found within and downstream of the eutrophic lakes in the West Fork Des Moines River watershed, where algal biomass may contribute 20 – 25 mg/L of TSS. In these areas, observed TSS should be compared to the sum of suspended sediment (SSED) and phytoplankton biomass (PHYTO) predicted by HSPF.

3.2 Hydrology Calibration Approach

The level of performance and overall quality of hydrologic calibration is evaluated in a weight of evidence approach that includes both visual comparisons and quantitative statistical measures. The calibration proceeds in a sequential manner through (1) general representation of the overall water balance, (2) calibration of snow depth, (3) assurance of consistency with satellite-based estimates of actual evapotranspiration (ET) and soil moisture, and (4) detailed calibration relative to flow gaging for seasonal flows, shape of the flow duration curve, and hydrograph shape.

Key parameters for hydrologic calibration and information on their potential ranges are as described in BASINS Technical Note 6 (USEPA, 2000). Initial values of key parameters were related to soil and climatological properties where appropriate. Specifically, infiltration rates (INFILT) were initialized (and subsequently varied by HSG), while initial values of lower zone nominal soil storage capacity (LZSN), upper zone soil storage capacity (UZSN), and interflow inflow (INTFW) were set based on annual average rainfall, consistent with USEPA (2000). Seasonal patterns based on vegetative cover (MON-LZETPARM, MON-INTERCEP, and MON-MANNING) and snow simulations were initialized based on past experience with Minnesota models.

A key aspect of hydrology in this part of Minnesota is the use of tile drains to enhance drainage of cropland. Tile drains alter the natural hydrologic regimes, can enhance sediment and nutrient delivery to the stream network, and can increase the erosive power of flows instream (Schottler et al., 2014). HSPF does not have an explicit representation of tile drains and it is not feasible to represent individual drain lines in a basin-scale model. In addition, the density of installed tile lines is not available in spatial coverages. The effects of tile drainage on hydrology can be effectively represented in the model through an enhanced interflow component that provides rapid subsurface movement of flow, as has been successfully demonstrated for the Minnesota River basin (Tetra Tech, 2009) as well as in an earlier USGS HSPF application to the Heron Lake watershed (Jones and Winterstein, 2000). This is accomplished in the model through use of a high value of the monthly interflow inflow parameter, for which a value around 3 during the growing season works well for the Des Moines River watershed (Jones and Winterstein used 3.4 in their model). The drainage systems include both subsurface tile and tile drains with surface inlets to drain depressions. As a result, the net effects of tile drainage are represented in the model as a combination of quicker interflow and slower groundwater discharge. Tetra Tech (2009) demonstrated that this representation is generally consistent with maximum drainage coefficients in the range of 0.25 in/day from tiled fields.

Given the inherent errors in input and observed data and the approximate nature of model formulations, absolute criteria for watershed model acceptance or rejection are not generally considered appropriate by most modeling professionals. Yet, most decision makers want definitive answers to the questions—"How accurate is the model?" and "Is the model good enough for this evaluation?" Consequently, the current state of the art for model evaluation is to express model results in terms of ranges that correspond to “very good,” “good,” “fair,” or “poor” quality of simulation fit to observed behavior. These characterizations inform appropriate uses of the model: for example, where a model achieves a good to very good fit, decision-makers often have greater confidence in having the model assume a strong role in evaluating management options. Conversely, where a model achieves only a fair or poor fit, decision makers may assume a much less prominent role for the model results in the overall weight-of-evidence evaluation of management options.

For HSPF and similar watershed models, a variety of performance targets have been documented in the literature, including Donigian *et al*. (1984), Lumb et al. (1994), Donigian (2000), and Moriasi *et al*. (2007). Based on these references and past experience, the HSPF performance targets for simulation of hydrology are summarized in Table 3-3. Model performance is generally deemed fully acceptable where a performance evaluation of “good” or “very good” is attained. It is important to clarify that the tolerance ranges are intended to be applied to mean values, and that individual events or observations may show larger differences and still be acceptable (Donigian, 2000).

The model calibration generally attempts to achieve a good balance between the relative error metrics and the Nash-Sutcliffe coefficient of model fit efficiency (NSE; Nash and Sutcliffe, 1970). Unlike relative error, NSE is a measure of the ability of the model to explain the variance in the observed data. Values may vary from $-\infty$ to 1.0. A value of NSE = 1.0 indicates a perfect fit between modeled and observed data, while values equal to or less than 0 indicate the model’s predictions of temporal variability in observed flows are no better than using the average of observed data. The accuracy of a model increases as the value approaches 1.0. Moriasi et al. (2007) suggest that achieving a relative error on total volume of 10 percent or better and an NSE of 0.75 or more on *monthly* flows constitutes a good modeling fit for watershed applications.

It should be noted that many of the available gage records in these watersheds operate only on a seasonal basis, so that full evaluation of seasonal statistics (or, indeed, evaluation of the total water balance) is not possible. In addition, where winter gaging records are available they are typically imprecise and generally rated poor or fair by USGS due to interference from ice cover.

Table 3-3. Performance Targets for HSPF Hydrologic Simulation (Magnitude of Annual and Seasonal Relative Mean Error (*RE)*; Daily and Monthly NSE)

Model Component	Very Good	Good	Fair	Poor
1. Error in total volume	≤ 5%	5 - 10%	10 - 15%	> 15%
2. Error in 50% lowest flow volumes	≤ 10%	10 - 15%	15 - 25%	> 25%
3. Error in 10% highest flow volumes	≤ 10%	10 - 15%	15 - 25%	> 25%
4. Error in storm volume	≤ 10%	10 - 15%	15 - 25%	> 25%
5. Winter volume error (JFM)	≤ 15%	15 - 30%	30 - 50%	> 50%
6. Spring volume error (AMJ)	≤ 15%	15 - 30%	30 - 50%	> 50%
7. Summer volume error (JAS)	≤ 15%	15 - 30%	30 - 50%	> 50%
8. Fall volume error (OND)	≤ 15%	15 - 30%	30 - 50%	> 50%
9. NSE on daily values	> 0.80	> 0.70	> 0.60	≤ 0.60
10. NSE on monthly values	> 0.85	> 0.75	> 0.65	≤ 0.65

3.3 Sediment Calibration Approach

Sediment is one of the more difficult water quality parameters to calibrate in watershed models because observed instream concentrations depend on the net effects of a variety of upland and stream reach processes, only some of which are directly observed. Further, conditions in one stream reach may depend strongly on erosion and deposition patterns in the upstream reaches. Thus mass balance checks need to examine every reach in the model. Sediment calibration was undertaken in accordance with AQUA TERRA (2012) as well as the guidelines BASINS Technical Note 8: *Sediment Parameters and Calibration Guidance for HSPF* (USEPA, 2006). Sediment calibration required an iterative approach. The first step in calibration involves setting channel erosion to values that achieve a reasonable fit to observations when upland erosion is at rates consistent with the literature and soil survey data. The upland simulation is then further tuned. Next, the long-term behavior of sediment in channels is constrained to a reasonable representation in which degradation or aggradation amounts are physically realistic and consistent with available local information. Finally, results from detailed local stream studies are used to further ensure that the model provides a reasonable representation in specific areas.

The upland parameters for sediment were related to soil and topographic properties. HSPF simulates sediment yield to streams in two stages. First, HSPF calculates the detachment rate of sediment by rainfall (in tons/acre) as $DET = (1 – COVER) \cdot SMPF \cdot KRER^{JRER}$, where *DET* is the detachment rate (tons/acre), *COVER* is the dimensionless factor accounting for the effects of cover on the detachment of soil particles, *SMPF* is the dimensionless management practice factor, *KRER* is the coefficient in the soil detachment equation, *JRER* is the exponent in the soil detachment equation, which is recommended to be set to 1.81, and *P* is precipitation depth in inches over the simulation time interval. Direct addition of sediment (e.g., from wind deposition) is also added via the parameter *NVSI*. Actual detached sediment storage available for transport (*DETS*) is a function of accumulation over time and the reincorporation rate, *AFFIX*.

The transport capacity for detached sediment from the land surface (*STCAP*) is represented as a function of overland flow: $STCAP = KSER \cdot (SURS + SURO)^{JSER}$, where *KSER* is the coefficient for transport of detached sediment, *SURS* is surface water storage (inches), *SURO* is surface outflow of water (in/hr), and *JSER* is the exponent for transport of detached sediment.

DET is similar in concept to the Universal Soil Loss Equation (USLE; Wischmeier and Smith, 1978), which predicts sediment detachment as a function of is the rainfall erosivity, *RE*, a soil erodibility factor, *K*, a length-slope factor, *LS*, a cover factor, C, and a practice factor, *P*: $DET = RE \cdot K \cdot LS \cdot C \cdot P$.

USLE predicts sediment loss from one or a series of events at the field scale, and thus incorporates local transport as well as sediment detachment.

There are two approaches that may be pursued from this point. One is to develop a formal approximation between the HSPF *KRER* and the USLE *K* factor as was done in Tetra Tech (2009). The other approach is to simply assume *KRER* = *K*, as is recommended in USEPA (2006). In theory, *KRER* ought to approximate the product of *K* and the *LS* factor, multiplied by a constant. However, slope is also a key factor in determining the depth of surface runoff and storage, which determines the sediment overland transport capacity in HSPF, so the approach of deriving *KRER* from *K* and *LS* may encounter complications in practice. In areas of generally low slopes variation of *KRER* with slope is expected to be small and the relationship will tend toward linear. Therefore, it is sufficient to use the approach recommended in USEPA (2006) and equate *KRER* and *K*, as was done for this model. The major difference between the two approaches is in the practical definition of the reincorporation rate, *AFFIX*, which will assume different values in order to achieve a stable seasonal cycle of *DETS*.

Once *KRER* is established, the primary upland calibration parameter for sediment is *KSER*, which determines the ability of overland flow to transport detached sediment. HSPF can also simulate gully erosion in which sediment generated from the land surface is not constrained by rainfall detachment. Strong evidence of gully erosion does not exist in the Des Moines River watershed. However, based on discussions experts with local knowledge of the watershed limited gully erosion was simulated for regions where the slope was generally greater than 3%.

The simulation also accounts for the transport of sediment through tile drains with surface inlets. The Des Moines River watershed model uses a simplified approach to this component of the model developed by RESPEC (2014) for updates to the Minnesota River model, in which GENER statements are used to route a fraction of the detached sediment on the soil surface to stream reaches as a function of the water storage on the soil surface. The flow and sediment moving through tile drains also transports both dissolved and sediment-associated nutrients and other pollutants.

Key parameters controlling channel erosion, deposition, and sediment transport within streams and rivers are as follows (USEPA, 2006):

KSAND: Sand transport is represented with a power function based on average velocity, such that carrying capacity for sand = *KSAND* x $AVVEL^{EXPSND}$. *KSAND* is set to 0.1 and *EXPSND* to 2 to start calibration and adjusted to improve the comparison between simulated and observed suspended sediment concentrations at flows where cohesive silt and clay sediments do not scour as well as to ensure a reasonable evolution of sand storage over time,.

TAUCD: HSPF calculates bed shear stress (TAU) during each model time step for each individual reach. The critical bed shear stress for deposition (lb/ft^2) represents the energy level below which cohesive sediment (silt and clay) begins to deposit to the bed. Initial values of TAUCD for silt and clay were estimated by reach by examining the cumulative distribution function of simulated shear stress and setting the parameter to a lower percentile of the distribution in each reach segment, as recommended by USEPA (2006). The 20th percentile was used for clay and the 25th percentile for silt.

TAUCS: The critical bed shear stress for scour (lb/ft^2) represents the energy level above which scour of cohesive sediment begins. Initial values of TAUCS were set, as recommended, at upper percentiles of the distribution of simulated shear stress in each reach (the 90th percentile for clay and the 95th percentile for silt). Values for some individual reaches were subsequently modified during calibration.

M: The erodibility coefficient of the sediment (lb/ft^2-d) determines the maximum rate at which scour of cohesive sediment occurs when shear stress exceeds TAUCS. This coefficient is a calibration parameter. It was initially set to 0.004 for silt, 0.003 for clay, and adjusted during calibration in some reaches.

An example of the distribution of shear stress versus flow for Okabena Creek is shown in Figure 3-3. The notch that appears in the profile around 1,000 cfs represents the reduction in cross-section averaged shear stress that occurs when the flow spreads overbank into the flood plain.

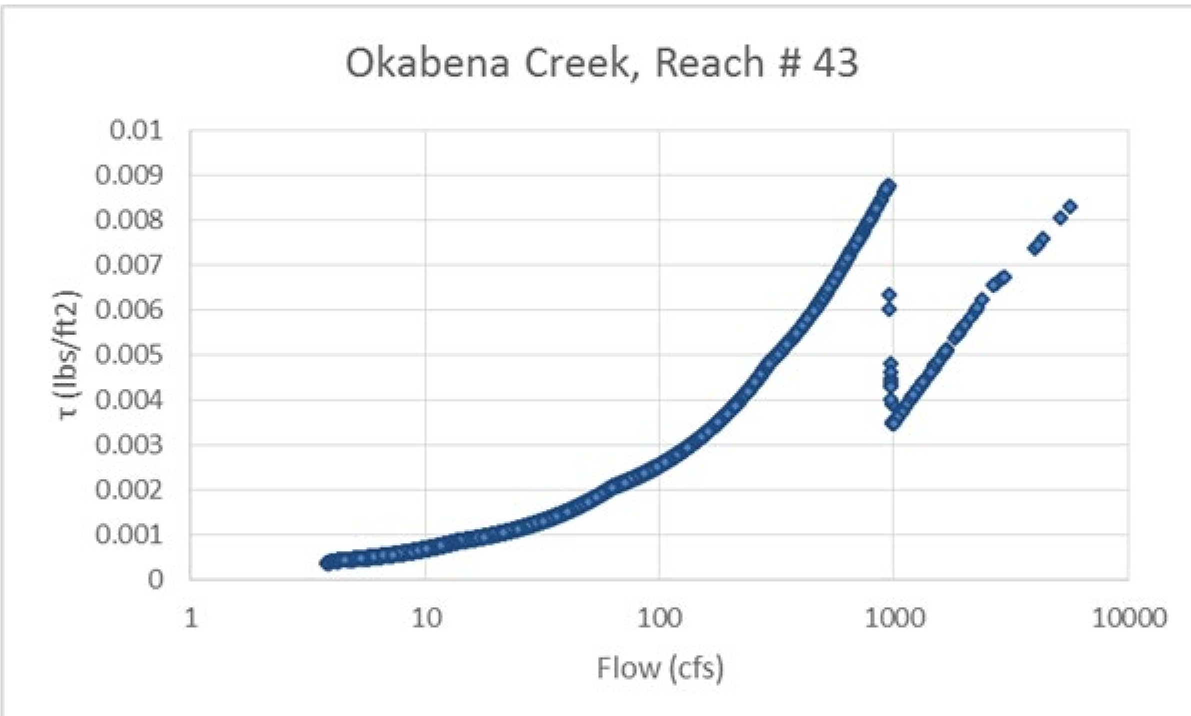

Figure 3-3. Shear Stress Distribution for Okabena Creek (Model Reach 43)

An important issue for sediment calibration is representing the correct division between sediment derived from uplands and sediment derived from reach scour. In some Minnesota watersheds, radionuclide analysis using ^{210}Pb and ^{10}Be, both of which are derived from the atmosphere and decay over time into more stable forms, has been used to identify the fraction of sediment that derives from upland sources in recent contact with the atmosphere. Such information is not available for the Fork Des Moines River watersheds at this time, but could potentially be used to further refine sediment calibration in the future.

Calibration for sediment and other water quality parameters differs from calibration for hydrology in that pollutant concentrations are in most cases not continuously monitored. Instead, observations typically provide measurements of conditions at a point in time and point in space via a grab sample. The discrete nature of these samples presents problems for model calibration: A sample that represents a point in time could have been obtained from a system where conditions are changing rapidly over time – for instance, the rising limb of a storm hydrograph. Such samples cannot be expected to be matched by a model prediction of a daily average concentration. On the other hand, there may be large discrepancies between dynamic model predictions of hourly concentrations and data that are a result of small timing errors in the prediction of storm event flow peaks. Spatially, grab samples reflect conditions in one part of a stream reach (which may or may not be composited over the width and depth of a cross section). HSPF model results, in contrast, represent average concentrations over the length of a stream reach which is assumed to be fully mixed. Model predictions and field observations inevitably have some degree of mismatch in space and time and, even in the best models, will not fully match. Accordingly, a statistical best fit approach is needed.

Performance targets for sediment calibration, based on Donigian (2000), are summarized in Table 3-4. These performance targets are evaluated for both concentration and load, where load is estimated from concentration, on paired data, and should only be applied in cases where there are a minimum of 20 observations. Model performance is generally deemed acceptable where a performance evaluation of “good” or “very good” is attained.

Table 3-4. Performance Targets for HSPF Sediment Simulation (Magnitude of Annual and Seasonal Relative Average Error (*RE*) on Daily Values)

Model Component	Very Good	Good	Fair	Poor
1. Suspended Sediment	≤ 20%	20 - 30%	30 - 45%	> 45%

3.4 Water Quality Calibration Approach

Water quality simulation depends on the simulation of hydrology and sediment transport. This section addresses the calibration and validation of the model simulation of water temperature, dissolved oxygen, nutrients, and algae.

Although not a primary focus of the modeling effort, water temperature simulation is important in the watershed model for several reasons: water temperature affects many biologically mediated processes that influence water quality in the streams, and the temperature of the water determines how it will mix when it enters the lake.

Daily average water temperature in shallow flowing streams is largely controlled by air temperature. Temperature cycles within the day, however, may be strongly affected by heat gain from incoming solar radiation and heat loss due to longwave back radiation. Both of these effects are controlled by the extent of cover and shading on the stream in addition to meteorological variables such as solar radiation and cloud cover.

A detailed diel simulation of stream water temperature is a complex undertaking. The timing and magnitude of heat fluxes are controlled by a variety of factors such as stream orientation and vegetative and topographic shading angles that cannot be fully represented in a basin-scale HSPF model. For example, a stream oriented east-west is likely to be exposed to unshaded solar radiation for a longer part of the day than a stream oriented north-south. Stream shading varies over the course of the year as canopy density changes, and may also change over time as trees grow, are cut, fall due to ice and wind storms, or due to fire. HSPF approximates all these complex details through the assignment of a temporally constant “surface exposed” (CFSAEX) factor that represents the average fraction of tree-top solar radiation reaching the water surface. Given these issues, the stream temperature calibration was checked for reasonableness, but not constrained to achieve specific statistical targets.

Loading of nutrients that may support excess algal growth is an important concern. Dense algal growth is an aesthetic and biological concern and also affects the oxygen balance over the course of the day through oxygen production by daytime photosynthesis and depletion by nighttime respiration. The major nutrients controlling algal growth are phosphorus and nitrogen. Both are simulated in detail in the model. Minor nutrients (e.g., silica, iron) may also play a role in determining algal response but are not simulated in the watershed model. The first step in a sequential process for nutrient calibration is to verify that unit area loading rates were reasonable compared to literature values. Next, calibration to instream observations is carried out to refine the simulation. Plant growth has an important effect on nutrient balances during low flow conditions and serves to convert inorganic nutrients into organic forms; therefore, nitrogen and phosphorus species must be calibrated simultaneously with algae.

The approach taken is to simulate three components in loading from the land surface as general quality constituents (GQUALs): inorganic nitrogen (nitrate, nitrite, and ammonia), inorganic phosphorus (total orthophosphate), and organic matter. Each of these constituents is then partitioned at the point of entry into the stream network:

- Inorganic nitrogen is partitioned into dissolved nitrate, dissolved ammonium, and sorbed ammonium. Fractions of the dissolved constituents are set to reproduce observed data, while sorption of ammonium is simulated using equilibrium partitioning assumptions (the model connects inorganic N from the land surface to dissolved N in the stream reach, but equilibrium partitioning to the sorbed form occurs instantaneously). Assignment of total inorganic nitrogen from the land surface to nitrate and ammonium at the point of entry to the stream is represented by a constant ratio throughout the model, but differs for agricultural land and impervious surfaces. Partitioning of ammonium between dissolved and sorbed forms depends on local suspended sediment concentrations. A small portion of the inorganic N is routed directly to organic N to represent uptake by heterotrophic organisms in low order streams (a process not explicitly simulated by the model).
- Inorganic phosphorus is partitioned into dissolved and sorbed fractions using equilibrium partitioning assumptions. As with ammonium, the fraction that becomes sorbed depends on the local suspended sediment concentration,
- Organic matter (biomass) is partitioned into labile and refractory organic carbon, organic nitrogen, and organic phosphorus components. Initial specifications were based on expected stoichiometry of forest litter, and then revised during calibration to achieve agreement with observed concentrations.

All three upland components (inorganic nitrogen, inorganic phosphorus, and organic matter) may be loaded through either surface flow or subsurface flow (interflow and groundwater discharge). The HSPF GQUAL algorithms do not maintain a full mass balance of subsurface constituents (which would require a groundwater quality model); rather, the user specifies concentration values, which may vary monthly, for interflow and groundwater. Surface washoff loading is considered from both pervious and impervious surfaces.

Inorganic phosphorus loading from pervious surfaces is simulated as a sediment-associated process because of the strong affinity of orthophosphate for soil particles. Surface loading of inorganic phosphorus is thus determined by a potency factor applied to sediment load, which may vary on a monthly basis to reflect changes in surface soil concentration associated with the annual growth cycle. (While this reflects the physical basis of surface loading of inorganic phosphorus, it does mean that any errors in the simulation of sediment loading will also affect estimates of inorganic phosphorus loading.) Subsurface flow pathways are assumed primarily to load small amounts of dissolved inorganic phosphorus. Organic matter is also simulated as a sediment-associated load from pervious surfaces, as this primarily represents the erosion of humus, leaf litter, and other detritus.

In contrast to phosphorus, inorganic nitrogen is highly soluble, and loading in surface runoff may occur independent of sediment movement (particularly where fertilizer is applied). Further, much of the nitrate load in surface runoff represents input from atmospheric deposition. Therefore, inorganic nitrogen loading from pervious surfaces is represented via a buildup-washoff process in which the user specifies a rate of accumulation, an accumulation limit, and a flow rate sufficient to remove 90 percent of the accumulated material.

As noted above, representation of plant growth is a necessary part of the nutrient calibration process. HSPF contains routines for simulating planktonic (floating) and benthic (attached) algae. Growth, respiration, and death processes are affected and potentially limited by the availability of light, availability of inorganic nutrients, water depth, and water temperature. Because HSPF represents stream segments as one-dimensional, fully-mixed reactors, the predictions of algal response are averages throughout the stream segment volume. Planktonic and benthic algae simulations differ primarily in the way that the attenuation of light availability is calculated. For plankton light availability is calculated as the average over the euphotic depth, such that all phytoplankton are assumed to be mid-depth in the reach or the middle of the euphotic zone, whichever is smaller, then adjusted to the full volume of the reach. Benthic algae are assumed to be at the average depth of the reach. These simplifying assumptions can distort the actual response in some situations. For deeper reaches, especially lakes, the phytoplankton simulation results are an average over the reach volume, which does not match well with chlorophyll *a* observations collected from the photic zone. When the average depth is large relative to the light extinction rate benthic algal growth will be simulated as minimal, whereas significant growth may actually occur in the shallower edges of the lake or stream. The scheme does not include a representation of floating or emergent rooted macrophytes. While these can sometimes be successfully approximated with the benthic algae routines, the light availability calculations for benthic algae are not appropriate to these types of macrophytes and the program does not consider that floating/rooted macrophytes can exchange gases with the atmosphere and obtain nutrients from the sediment.

The dissolved oxygen simulation considers reaeration, the decay of organic matter (carbonaceous biochemical oxygen demand), oxidation of ammonia and nitrite N, sediment oxygen demand, and algal photosynthesis and respiration.

For most water quality constituents, it is unreasonable to propose that the model predict all temporal variations in concentration and load. The model should, however, provide an accurate representation of long-term and seasonal trends in concentration and load, and correctly represent the relationship between flow and load. To ensure this, it is important to use statistical tests of equivalence between observed and simulated concentrations, rather than relying on a pre-specified model tolerance on difference in concentrations.

Ideally, average errors and average absolute errors should both be low, reflecting a lack of bias and high degree of precision, respectively. In many cases, the average error statistics will be inflated by a few highly discrepant outliers. It is therefore also useful to compare the median error statistics.

General performance targets for water quality simulation with HSPF are also provided by Duda et al. (2012) and are shown in Table 3-5. These are calculated from observed and simulated daily concentrations, and should only be applied in cases where there are a minimum of 20 observations.

Table 3-5. Performance Targets for HSPF Water Quality Simulation (Magnitude of Annual and Seasonal Relative Average Error (RE) on Daily Values)

Model Component	Very Good	Good	Fair	Poor
Temperature	≤ 7%	8 - 12%	13 - 18%	> 18%
Water Quality/Nutrients	≤ 15%	15 - 25%	25 - 35%	> 35%

Evaluation of water quality simulations presents a number of challenges because, unlike flow, water quality is generally not monitored continuously. Grab samples at a point in space and time may not be representative of average conditions in a model reach on a given day due to either spatial or temporal uncertainty (i.e., an instantaneous measurement in time may deviate from the daily average, especially during storm events, while a point in space may not be representative of average conditions across an entire model reach). Where constituent concentrations are near reporting levels, relative uncertainty in reported results is naturally high. Accurate information on daily variability in point source loads is also rarely available.

Evaluation of relative average error is recommended, but averages are prone to biasing by one or a few extreme outliers. Therefore, it is also useful to examine median relative errors, which are less influenced by outliers.

The performance targets for water quality simulation may be applied to either concentrations or loads. Concentrations provide the most natural metric, but error magnitude may be unduly influenced by variability at low flow conditions that has little effect on cumulative loading downstream. Loads are more meaningful for impacts in downstream lakes, harbors, and estuaries but are not directly observed and need to be estimated from flow and concentration – both uncertain. Tests on loads are performed in two ways: on paired data (observed and simulated daily average concentration multiplied by flow) and on complete time series of monthly loads. For the latter approach, “observed” monthly loads are estimated using the USACE FLUX32 program (a Windows-based update of the FLUX program developed by Walker, 1996), and are themselves subject to significant uncertainty.

Additional statistical tests are also applied as part of a weight-of-evidence examination of the water quality calibration. Two-sample *t*-tests are reported on the differences in mean concentration and mean load, with higher probability values indicating less chance that the measures are systematically different. A problem with the *t*-test is that the test is on a null hypothesis that the mean difference is exactly equal to zero, not whether the difference is physically meaningful. Therefore, a low value on the *t*-test (rejection of the null hypothesis) is generally considered of practical significance only when the mean difference is greater than 10 percent. Additional graphical tests are also performed to ensure that errors in the prediction of load and concentration do not exhibit strong correlations relative to flow magnitude and season.

4 Hydrology Calibration and Validation Results

4.1 Snow Calibration

Snow accumulation and melt is a key component of the water balance in northern watersheds. Daily snow depth and snow water equivalent as simulated by the HSPF model were compared to observed snow datasets available from the National Snow and Ice Data Center (NSIDC). The NSIDC Snow Data Assimilation System (SNODAS) data products integrate snow data from satellites, ground observations and aircrafts to provide estimates of snow cover and associated parameters (Carroll *et al.*, 2001). Snow depth and snow water equivalent are available from September 2003 to present at a spatial resolution of 1 km by 1 km and a temporal resolution of 1 day for the Continental United States (CONUS). HSPF simulated daily time-series were compared to SNODAS aggregated by weather regions.

Values of parameters in the SNOW-PARM1 and SNOW-PARM2 blocks of the HSPF model were configured by weather region as part of the calibration process for snow. The calibrated values of these parameters are provided in Table 4-1. Summary statistics of snow calibration for depth and water equivalents are provided in Table 4-2. Graphical comparison are shown in Figure 4-1 and Figure 4-2 to and represent average monthly modeled snow depth and snow water equivalents against SNODAS estimates, respectively.

Modeled snow depth and snow water equivalent are generally within 15% of the SNODAS estimates indicating reasonable performance. The calibrated model shows a good fit for all weather regions in the watershed. Modeled snow depth and snow water equivalent are however approximately 20% more than SNODAS estimates in weather region 2 (Lake Shetek sub-watershed).

The fit to snow depth and snow water equivalent is approximate as uncertainties exist in the interpretation of remotely sensed data for the SNODAS dataset. It is also important to note that snow fall and melt in the model are highly sensitive to ambient air temperature. Small inconsistencies in air temperatures may have potentially significant impacts on snow behavior, including whether precipitation is interpreted by the model as snow.

Table 4-1. HSPF Snow Calibration Parameter Values

Parameter	Description	Calibrated Value	Recommended Range
SHADE	Fraction shaded from solar radiation	0.5 (Forest) 0.25 (Water/Wetlands) 0.1 (All other land-covers)	0 - 0.8
SNOWCF	Snow gage catch correction factor	1.2 (for WRG 5) and 1.0 (for all others)	1.0 - 2.0
COVIND	Snowfall required to fully cover surface	0.1 - 0.8	0.1 - 10.0
RDCSN	Density of new snow	0.12	0.05 - 0.30
TSNOW	Temperature at which precipitation becomes snow	32.0	30.0 - 40.0
SNOEVP	Snow evaporation factor	0.10	0.0 - 0.5
CCFACT	Condensation/convection melt factor	1	0.5 - 8.0
MWATER	Water storage capacity in snowpack	0.1	0.005 - 0.2
MGMELT	Ground heat daily melt rate	0.001	0.0 - 0.1

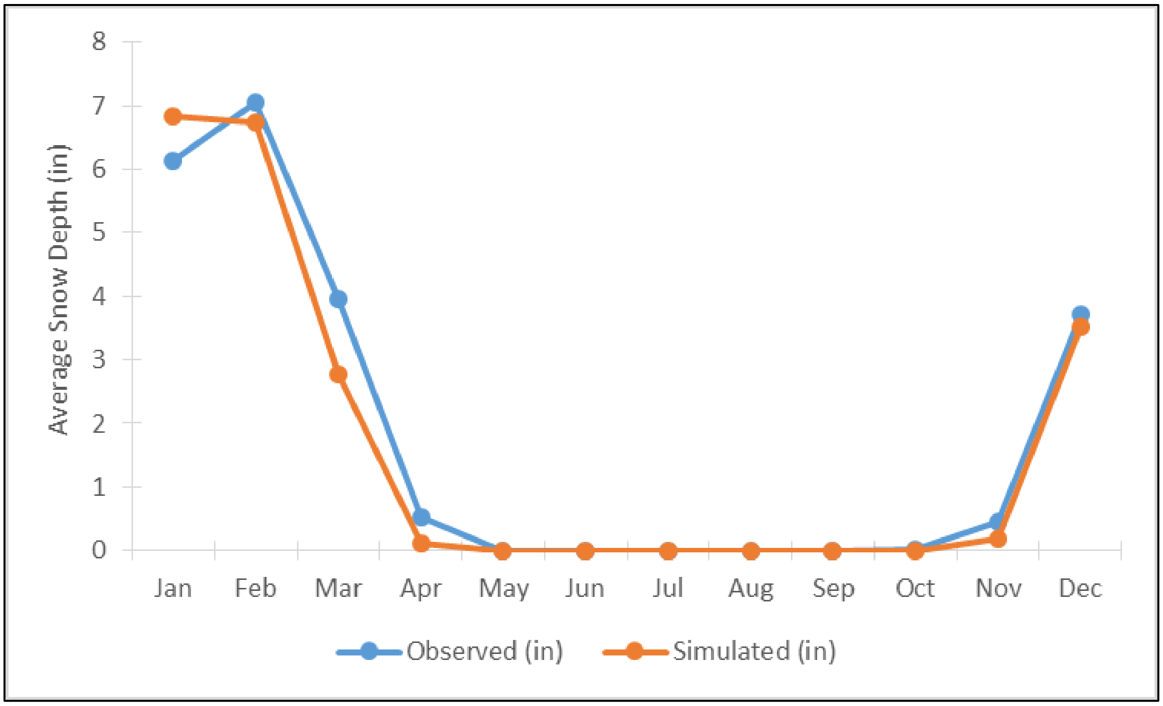

Figure 4-1. Mean Monthly Snow Depth in the Des Moines River Watershed (10/1/2003 – 12/31/2014)

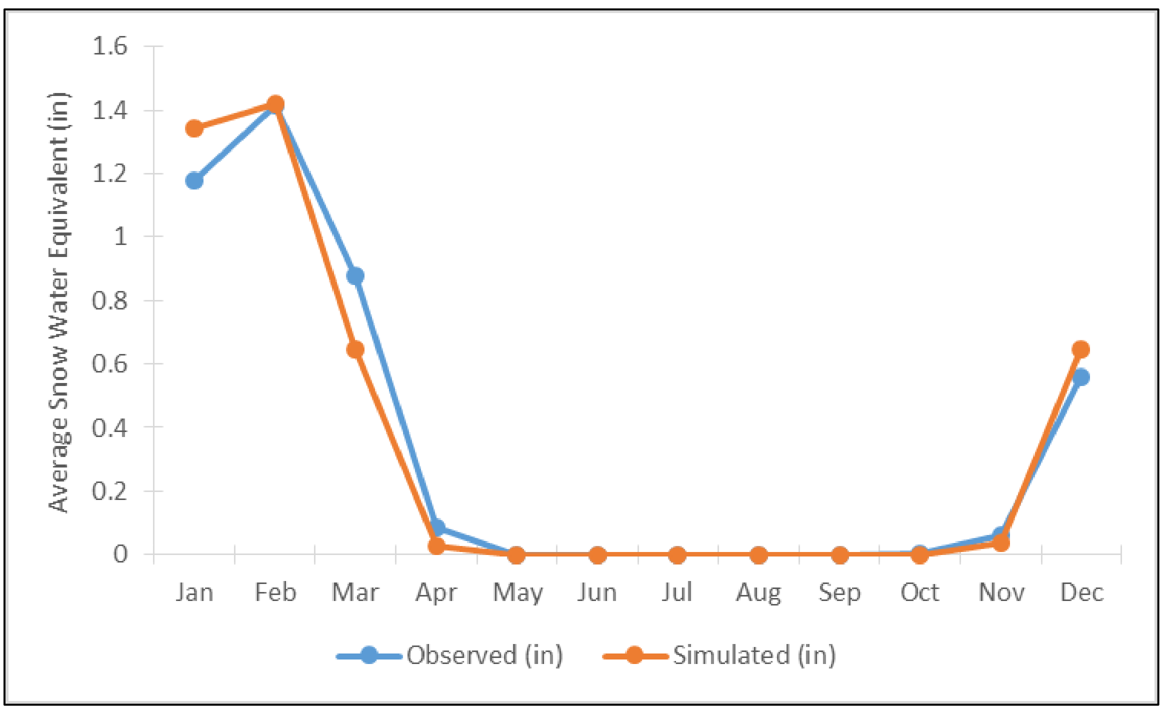

Figure 4-2. Mean Monthly Snow Water Equivalent in the Des Moines River Watershed (10/1/2003 – 12/31/2014)

Table 4-2. Summary of Snow Calibration Results

Weather Zone #	Snow Depth			Snow Water Equivalent		
	Total Error*	Daily R^2	Daily NSE	Total Error	Daily R^2	Daily NSE
1	-11.7%	0.86	0.73	-10.9%	0.85	0.76
2	-21.5%	0.87	0.70	-20.1%	0.87	0.74
3	-15.0%	0.88	0.78	-11.6%	0.87	0.80
4	-4.4%	0.88	0.79	-5.8%	0.85	0.79
5	-8.8%	0.86	0.84	-13.5%	0.82	0.81
6	-5.4%	0.89	0.82	-5.5%	0.86	0.82
7	10.0%	0.83	0.80	-0.1%	0.79	0.77
8	8.8%	0.84	0.79	-0.3%	0.81	0.80
9	7.8%	0.82	0.80	2.5%	0.77	0.75
10	9.0%	0.85	0.83	4.2%	0.82	0.79
Watershed	**-1.9%**	**0.88**	**0.84**	**-5.0%**	**0.85**	**0.84**

* Total error is calculated as the Δ = (simulated - observed)/observed

4.2 Constraints on Soil Moisture Balance and Evapotranspiration

Evapotranspiration is the sum of evaporation from soil, water, and leaf surfaces and transpiration of soil water by plants. Evapotranspiration (ET) is the largest component of the water balance and is thus crucial to hydrologic calibration. Actual ET is often unconstrained in watershed models due to a lack of observed data. This issue was addressed for the Des Moines River watershed model through the use of remotely sensed ET data. The MODIS Global Evapotranspiration Project (MOD16) provides estimates of global terrestrial ET by using satellite remote sensing data at a spatial scale of 1 km^2 grid and at temporal scales of 8-days, months, and yearly totals from 2000 to 2010. It is important to recognize that MODIS does not directly measure evapotranspiration. Rather, an algorithm that considers MODIS land cover, albedo, leaf area index, and enhanced vegetation index is combined with daily meteorological data from NASA's Global Modeling and Assimilation Office reanalysis datasets using a Penman-Monteith type of approach (Mu et al., 2011). A validation study (Velpuri et al., 2013) showed that MODIS was able to estimate monthly ET within about 25 percent based on comparison to FLUXNET studies. These data are thus imprecise, but are useful to check that modeled ET patterns are realistic.

Monthly ET estimates for the watershed were extracted from the global MOD16 dataset. The gridded data were then aggregated to the level of the weather regions. The aggregated monthly data were compared to actual ET (TAET) simulated by the model and used to inform the pan coefficients used to convert Penman Pan PET to land surface PET in the model. Pan evaporation coefficients for all weather regions were set from 0.70 to 0.75. The pattern of observed monthly evapotranspiration was also used to refine the MON-INTERCEP and MON-LZETPARM blocks in the HSPF model.

Table 4-3 provides a summary comparison of simulated ET versus MODIS estimates. Figure 4-3 shows mean monthly simulated evapotranspiration in comparison with MODIS estimates for the Des Moines River watershed model. In general, the simulated ET is similar to that estimated by MODIS except in the spring and early summer months. MODIS predicts a slower ramp up of ET in the spring and summer. This may be because the MODIS algorithm relies on leaf area whereas a significant part of the total evaporation during early periods of snow melt and plant growth may come directly from the soil surface.

Table 4-3. Summary of Evapotranspiration Calibration Results

Weather Region #	Total Error*	Monthly R^2
1	-4.7%	0.74
2	-0.4%	0.76
3	-5.5%	0.71
4	-13.8%	0.73
5	-1.4%	0.68
6	-8.0%	0.70
7	-7.0%	0.75
8	-7.5%	0.77
9	-6.2%	0.77
10	-6.2%	0.73
Watershed	**-6.3%**	**0.74**

* Total error is calculated as the Δ = (simulated - observed)/observed

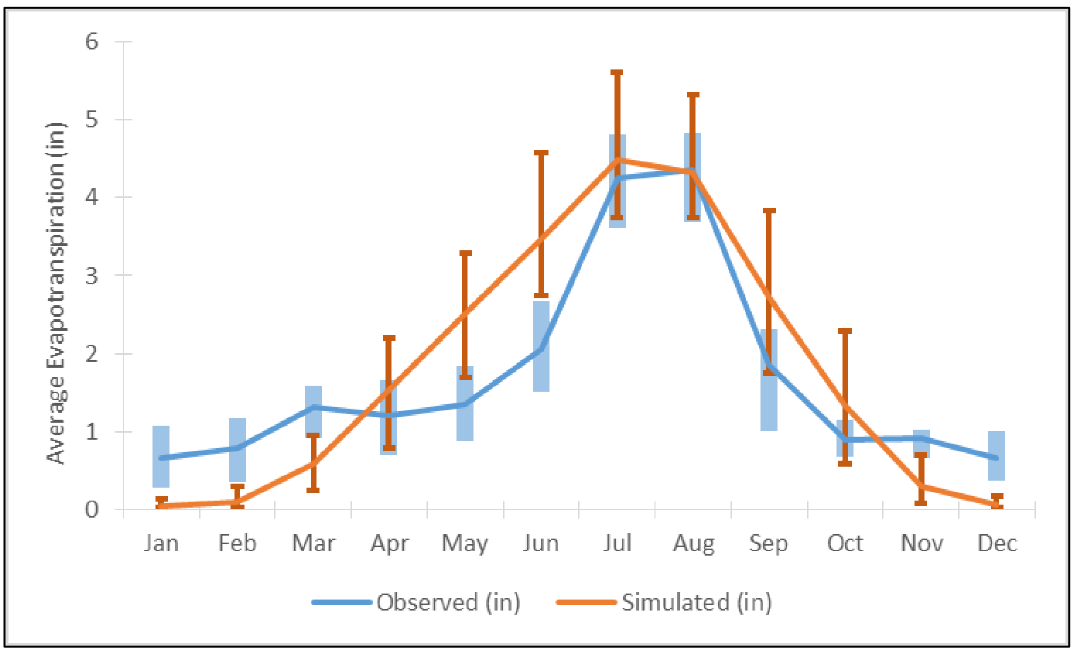

Figure 4-3. Comparison of Average Monthly Simulated Evapotranspiration to MODIS Estimates for Des Moines River Watershed Model Domain[1]

[1] The error bars on the chart show the range of observed and simulated monthly evapotranspiration while the solid lines represent the averages.

4.3 Flow Calibration

Calibration and validation for streamflow focused on the periods of 2002–2014 and 1995–2001, respectively. Calibration was completed by comparing time-series model results to gaged daily average flow. Key considerations in the hydrology calibration were the overall water balance, the high-flow to low-flow distribution, storm flows, and seasonal variations. The criteria in Table 3-3 were used to evaluate the quality of model fit.

Model parameter adjustment followed the guidance and ranges in USEPA (2000) and AQUA TERRA (2012). Calibrated values of some of the key hydrologic parameters are summarize in Table 4-4. The full set of model parameters is available in the User Control Input (uci) model file supplied electronically.

Table 4-4. Selected Hydrology Parameters for Pervious Land

<table>
<tr><th>Land Cover</th><th>HSG</th><th>LZSN</th><th>INFILT</th><th>AGWRC</th><th>KVARY</th><th>DEEPFR</th><th>UZSN</th></tr>
<tr><td rowspan="2">Forest, Grass, Pasture</td><td>A,B</td><td rowspan="2">5</td><td>0.4</td><td rowspan="9">0.92 – 0.99 (by region)</td><td rowspan="2">1.5 – 3.5 (by region)</td><td rowspan="9">0</td><td rowspan="2">0.7 - 0.9</td></tr>
<tr><td>C,D</td><td>0.1</td></tr>
<tr><td>Wetland</td><td>NA</td><td>1.8</td><td>0.5</td><td>0</td><td>2.5</td></tr>
<tr><td>Urban</td><td>All</td><td rowspan="6">5</td><td>0.1</td><td rowspan="6">1.5 -3.5 (by region)</td><td>0.3</td></tr>
<tr><td rowspan="2">Conv. Tillage</td><td>A,B</td><td>0.4</td><td rowspan="2">0.2 – 0.5</td></tr>
<tr><td>C,D</td><td>0.1</td></tr>
<tr><td rowspan="2">Conserv. Tillage</td><td>A,B</td><td>0.4</td><td rowspan="2">0.4 – 0.6</td></tr>
<tr><td>C,D</td><td>0.1</td></tr>
<tr><td>Manured Crops</td><td>All</td><td>0.1</td><td>0.7 – 1.2</td></tr>
</table>

Only two of the gages are maintained continuously through the winter and many of the other gages have seasonal records for two years only, complicating the calibration effort. Table 4-5 summarizes the calibration results for gages in the Des Moines River watershed. Detailed analyses of all gages are provided in Appendix B.

While there are many gages in the watershed, the majority have only operated for a few years, and most report data only seasonally. Rating curves are also imprecise for many of these stations due to continual shifting of bed forms. This lends considerable uncertainty to the calibration. The short operational period of most gages also means that there are limited data for temporal validation. Hydrologist's notes accompanying the gage records show various periods in which there were equipment failures or the rating curve was suspect due to unstable channel conditions. Large percentage errors in low flows can arise in response to a small difference in actual flows in streams that have low summer flows, while highest flow errors can be unduly influenced by a single large event that requires extrapolation of the rating curve.

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Table 4-5. Summary of Flow Calibration Results

Statistic	Lake Shetek Outlet (51078001)	Beaver Creek (51069001)	Lime Creek (51055001)	W Fork Des Moines, Avoca (51065001)	W Fork Des Moines, Heron Lake (51021001)	Jack Creek, Heron Lake (51092001)	Okabena Creek (51093001)	Heron Lake Outlet (51017001)	W Fork Des Moines, Windom (51011001)	W Fork Des Moines, Jackson (USGS 05476000)	Martin County Ditch, Dunnell (53008001)	Fourmile Creek, Dunnell (53014001)	Des Moines River at Estherville, IA ((USGS 04576500)
Error in total volume (%)	0.93	9.12	7.27	9.14	16.85	-2.00	-1.29	-2.88	12.80	-2.55	-22.11	14.01	6.74
Error in 50% lowest flows (%)	17.31	46.76	82.52	57.11	100.72	48.67	50.41	23.31	16.54	8.04	17.51	12.85	7.93
Error in 10% highest flows (%)	-11.39	6.37	7.76	5.48	9.00	-2.61	-15.40	4.79	14.37	-0.52	-40.74	1.09	7.15
Seasonal volume error - Summer (%)	30.26	127.40	109.07	32.45	38.20	26.87	24.22	11.02	40.39	5.45	-5.60	69.60	27.07
Seasonal volume error - Fall (%)	0.00	ND	ND	ND	0.00	-3.79	-25.79	-0.26	ND	-6.39	0.89	ND	-3.57
Seasonal volume error - Winter (%)	ND	ND	-ND	-9.53	ND	ND	ND	ND	ND	10.17	ND	ND	5.26
Seasonal volume error - Spring (%)	-4.39	-8.82	-11.02	4.21	10.18	-8.36	-4.82	-8.41	-0.79	-8.17	-7.46	-4.11	2.28
Error in storm volumes (%)	-33.37	9.73	20.93	35.71	7.77	23.89	3.72	54.83	46.92	40.35	16.21	65.57	43.88
Error in summer storm volume (%)	16.53	165.92	148.18	28.30	3.21	47.80	21.03	53.72	90.95	47.66	11.38	79.11	62.91
Nash-Sutcliffe Coefficient of Efficiency, (NSE)	0.83	0.72	0.66	0.70	0.83	0.67	0.71	0.56	0.67	0.70	0.35	0.54	0.580
Baseline adjusted coefficient (Garrick), E'	0.64	0.53	0.46	0.65	0.66	0.57	0.51	0.44	0.56	0.57	0.32	0.26	0.503
Monthly NSE	0.97	0.90	0.77	0.96	0.95	0.80	0.81	0.72	0.91	0.77	0.32	0.35	0.730

Lake Shetek Outlet near Currie (HYDSTRA# 51078001)

Seasonal gaging is available for two years only. The streamflow at this location is influenced by Lake Shetek itself and a series of other lakes upstream of the gage, including Bloody, Sarah and Currant. Outflow from Lake Shetek declines towards zero during dry periods. The model performance is generally good for total flow, seasonal flow volumes, and high flows. The model under-predicts low flow and storm flow volumes and the performance is only fair for these flow components. These under-predictions are likely a result of uncertainties associated with the representation of stage-discharge relationship of lakes using FTables. Daily and monthly NSE values indicate very good model performance.

Beaver Creek near Currie (HYDSTRA# 51069001)

Seasonal gaging is available for two years only. The model performance is rated as fair for total flow, and high and low flow volumes but ranges from fair to poor for seasonal flow and storm flow volumes. These discrepancies in streamflow are likely due to the interactions between the groundwater and surface water which are only approximated by the model. It is important to note that the period of record is very short for model evaluation. The percent errors are also inflated by the relatively small magnitudes of observed seasonal flows. The daily NSE indicates good model performance while the monthly NSE indicates very good model performance.

Lime Creek near Lime Creek (HYDSTRA# 51055001)

Seasonal gaging is available for two years only. The model performance is generally good for all flow components except for low flow, summer flow and summer storm volumes. Streamflow in Lime Creek is influenced by the presence of Lime Lake and surficial aquifers and declines to minimal amounts during dry periods. The uncertainties in stage-discharge relationship associated with Lime Lake along with the generic representation of groundwater-surface water interactions are likely causes for inconsistencies between observed and simulated streamflow (see Appendix B). The model performance in terms of daily and monthly NSE is, however, good. It is important to note that model performance here is based on comparison of data from two years for the spring-summer seasons, which is a relatively short time-frame for model evaluation.

West Fork Des Moines River near Avoca (HYDSTRA# 51065001)

Seasonal gaging is available for two years only. The model performance at this location is generally good for all flow components except low flow and summer storm volumes for the relatively short period of available data. The model performance for low flow and storm flow volumes is poor and is likely on account of the uncertainties associated with modeling the groundwater-surface water interactions and propagation of errors from upstream tributaries that include Beaver Creek and Lake Shetek. The daily NSE value indicates fair model performance while the monthly NSE value indicates very good model performance.

West Fork Des Moines River near Heron Lake (HYDSTRA# 51021001)

Seasonal gaging is available for two years only. The model performance at this location is good for all flow components except low flow volumes. The model performance in terms of low flow volume is poor and may be attributable to uncertainties in groundwater-surface water interactions, propagation of error from upstream segments and uncertainties associated with stage-discharge relationships of lakes and impoundments. The daily and monthly NSE values indicate very good model performance.

Jack Creek near Heron Lake (HYDSTRA# 51092001)

This is a long-term seasonal gage, lacking winter data. The model performance varies between good and very good for most of the flow components except low flow volumes. The model performance for low flow volumes is rated as poor and likely due the uncertainties associated with groundwater-surface water interactions and stage-discharge relationships associated with major lakes (East and West Graham, and First and Second Fulda). The daily and monthly NSE values indicate good model performance.

Okabena Creek near Okabena (HYDSTRA# 51093001)

The model performance is generally good to fair for all flow components except that the model also over-predicts low flow volumes. This gage is downstream from two large point source discharges (Worthington Industrial WWTP and Worthington WWTP) that have significant impacts on low flow volumes. These point sources are represented in the model as monthly discharges as daily data are not currently available. The performance at this location may improve from using a daily time-series for the point sources. The model under-predicts one large snowmelt peak in March 2010 which contributes to the relatively large error in winter flow volumes. The under-prediction of this single snowmelt event is also likely responsible for the net under-prediction of total and high flow volumes. It is also apparent that the model is not able to reproduce some of the peak flow volumes in the fall months. This might well be due to inconsistencies in rainfall data from PRISM. The observed flow volumes are also subject to uncertainties due to errors in rating curves constructed in shifting sand channels.

Heron Lake Outlet near Heron Lake (HYDSTRA# 51017001)

This is a long-term seasonal gage, lacking winter data. The model performance is generally very good to good for all flow components except for low flow and storm volumes for which the model performance is poor. The streamflow is influenced by the presence of large lakes (North and South Heron Lakes, Duck, Flaherty and Timber). Inaccuracies in representation of stage-discharge relationship and propagation of error from upstream reaches (Jack Creek and Okabena Creek) are likely causes of inconsistencies observed for this location. In addition the Heron marsh may have an impact on streamflow on this location which is currently being represented as a generic water land use instead of an impoundment with stage-discharge relationship due to a lack of bathymetric data. The model performance is poor and fair in terms of daily and monthly NSE, respectively.

West Fork Des Moines River near Windom (HYDSTRA# 51011001)

Seasonal gaging is available for two years only. The model generally over predicts the different flow components and the overall performance varies between poor and fair for the short period of available data. The alluvial aquifer reaches its greatest thickness in this portion of the Des Moines River (Adolphson 1983) and provides a large amount of water storage. The groundwater reach representation in the HSPF model is tuned to maintain the storage in the groundwater reaches relatively stable at the spatial scale of a weather region, but it is likely that a higher rate of loss from streams to the aquifer is warranted for this portion of the model domain. Well withdrawals from the alluvial aquifer may also impact surface flows. Fully resolving these components would require linking the HSPF watershed model to a groundwater flow model, which could be addressed in future enhancements. The daily NSE value indicates fair model performance while the monthly NSE value indicates very good model performance.

Des Moines River at Jackson (USGS 05476000)

This gage has long-term continuous flow data for the calibration period. The model fit statistics are generally very good to good with the exception of low flow and storm flow volumes. The errors in low flow and storm flow are likely due to propagation of errors from upstream segments and spring snowmelt peaks in some years are under-estimated (Figure 4-4). The daily and monthly NSE values indicate good model performance.

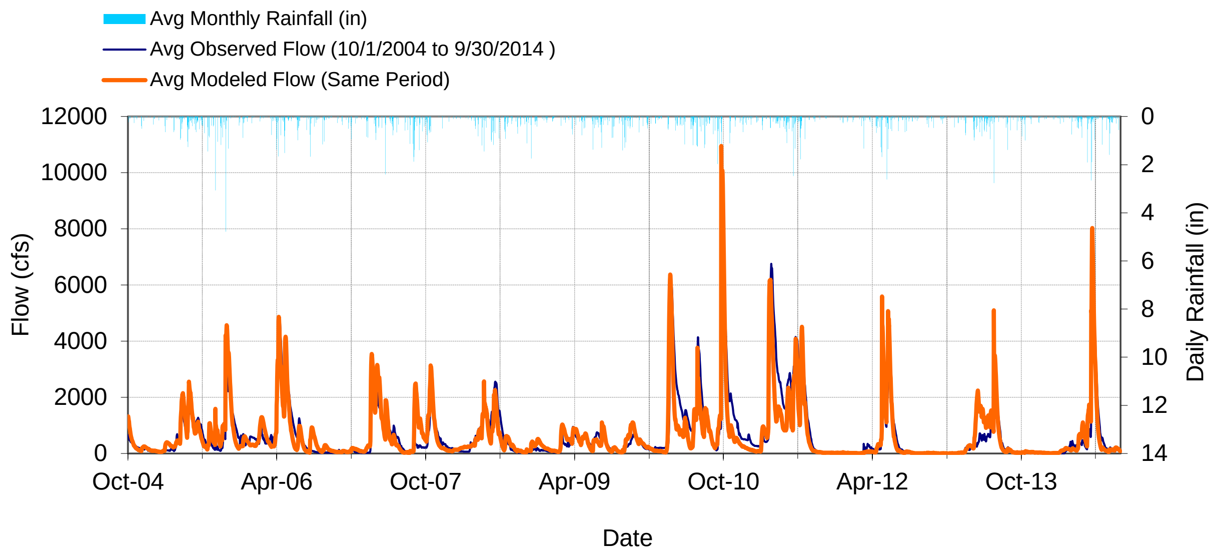

Figure 4-4. Simulated and Gaged Flow, Des Moines River at Jackson (USGS 05476000)

Martin County Ditch and Fourmile Creek (HYDSTRA# 53008001 and 53014001)

The model performance for both gages on the tributaries of the East Fork Des Moines River is poor. It is however important to note that a true evaluation of model performance is not possible at this location due to the paucity of data. Flow data are essentially available for two seasons. In addition, the drainage areas associated with these gages are small and are likely not representative of the streamflow behavior in the larger Des Moines River watershed.

Des Moines River at Estherville, IA (USGS 04576500)

Recent records at this station are maintained by USACE and appear to have occasional discrepancies when compared to upstream gages. Nonetheless, model fit statistics are generally good, except that summer flow volumes and storm volumes appear to be over-predicted. Flow at this station is strongly impacted by interactions between the river and the alluvial aquifer upstream and performance would likely be improved through linkage to a groundwater model. As with the upstream gage on the Des Moines River at Jackson, the large flows of 2010-2011 appear to be somewhat under-predicted, although the model and data otherwise track well (Figure 4-5). See Appendix B for full details.

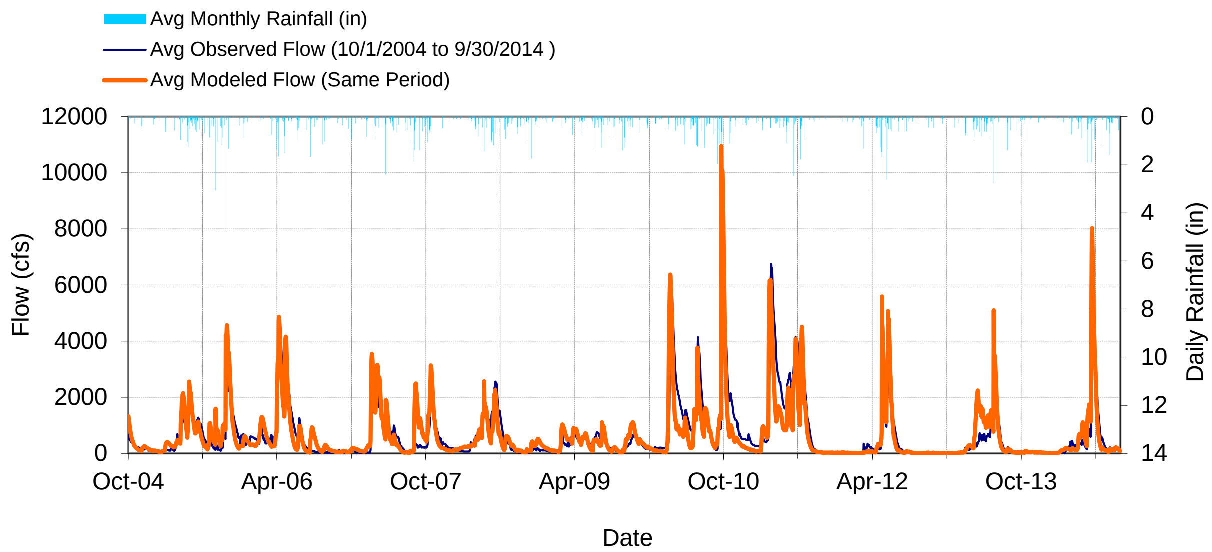

Figure 4-5. Simulated and Gaged Flows, Des Moines River at Estherville, IA (USGS 04576500)

Note: Flow records for this period were maintained by the U.S. Army Corps of Engineers

4.4 Flow Validation

Des Moines River at Jackson is the only site that had data prior to 2002 to pursue hydrology validation. The results for the validation period are similar to those for the calibration period and generally confirm the calibration (Table 4-6). The daily and monthly NSE values indicate good model performance.

Table 4-6. Summary of Flow Validation Results

Statistic	W Fork Des Moines, Jackson (USGS 05476000)
Error in total volume (%)	-9.33
Error in 50% lowest flows (%)	3.68
Error in 10% highest flows (%)	-8.93
Seasonal volume error - Summer (%)	29.27
Seasonal volume error - Fall (%)	-21.29
Seasonal volume error - Winter (%)	19.64
Seasonal volume error - Spring (%)	-23.48
Error in storm volumes (%)	15.02
Error in summer storm volumes (%)	71.80
Nash-Sutcliffe Coefficient of Efficiency, E	0.72
Baseline adjusted coefficient (Garrick), E'	0.57
Monthly NSE	0.79

4.5 Water Balance Summary

The water balance components of the Des Moines River watershed model (excluding evapotranspiration) are shown in Figure 4-6. Shallow or "active" groundwater discharge is the highest contributor of flow from the land surface to the stream network (81%) followed by interflow (15%) and surface runoff (4%). In terms of the disposition of the total water supply from precipitation, ET (at 74%) is the largest component (Figure 4-7), consistent with the findings of Sanford and Selnick (2013) for this region.

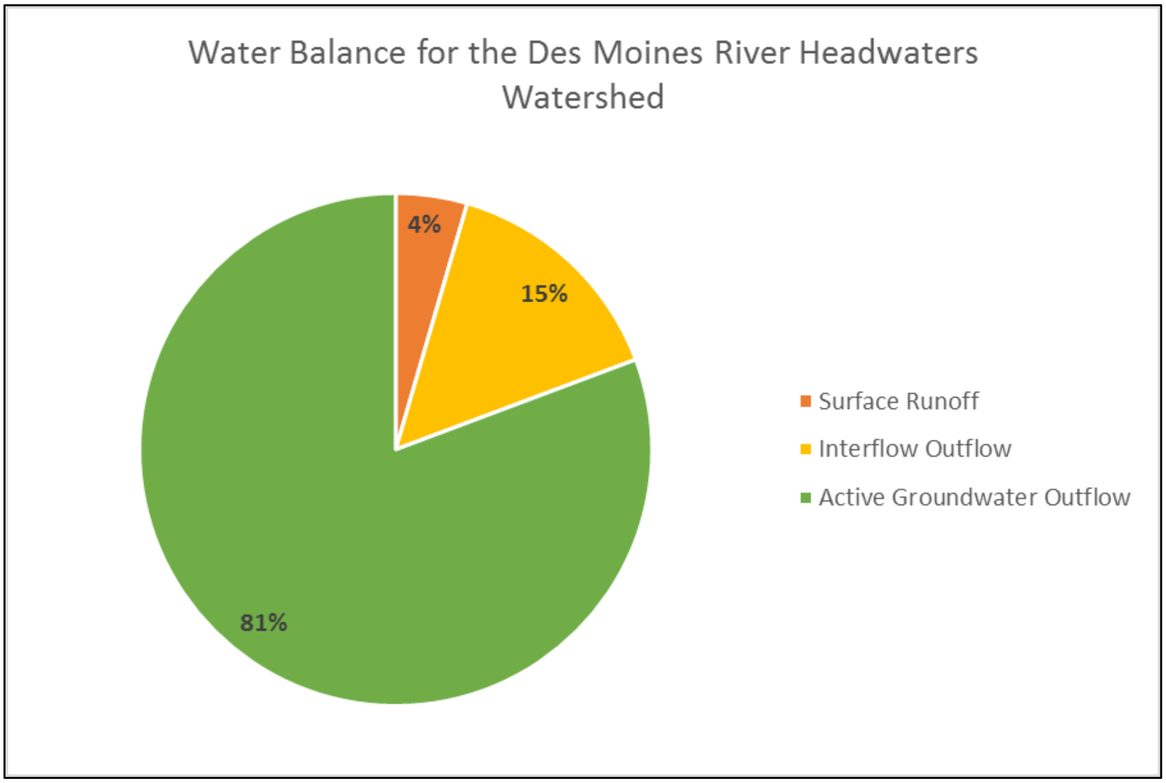

Figure 4-6. Water Balance Components for the Des Moines River Watershed Excluding Evapotranspiration (1995–2014)

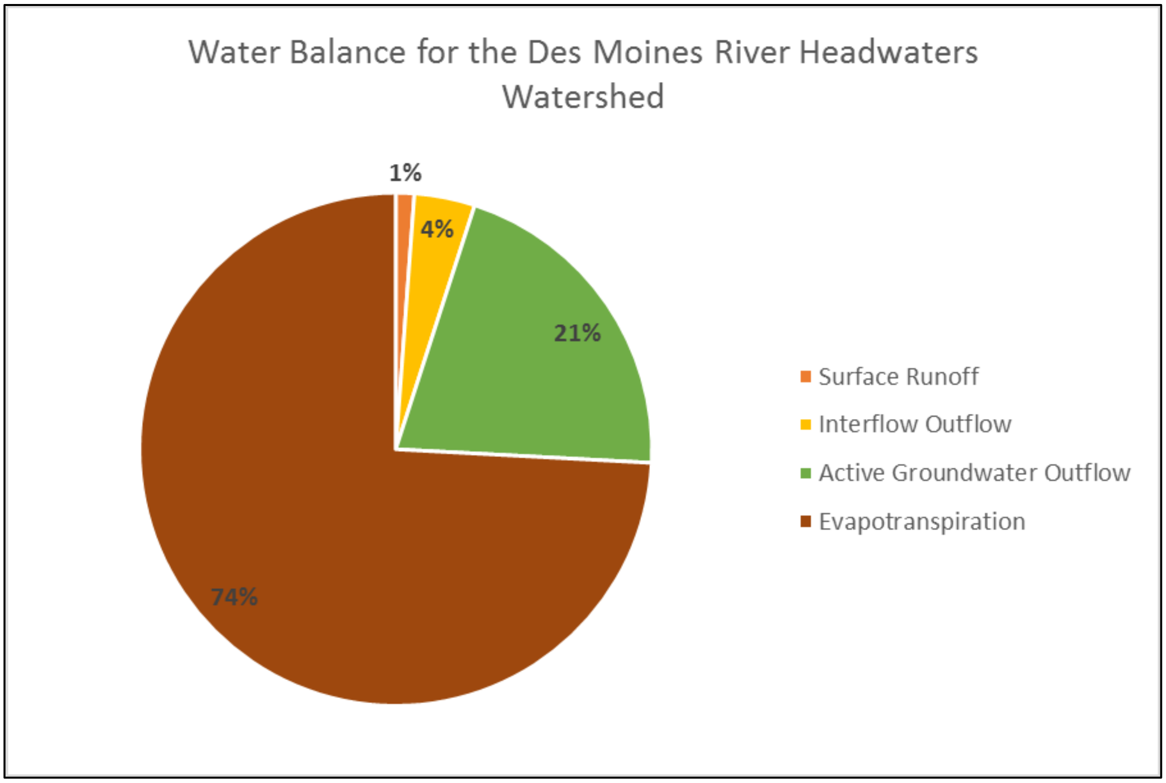

Figure 4-7. Water Balance Components for the Des Moines River Watershed Including Evapotranspiration (1995–2014)

As noted in Section 2.3.3, an important part of the overall water balance in this watershed is the interplay of the surface streams with the alluvial aquifer. Most of the recharge to the alluvial aquifer is simulated as a local loss for a given reach and a gain to the next downstream surface and groundwater reaches. Permanent loss to deep groundwater is simulated as a fraction of outflow coming out of the alluvial aquifer only in parts of the watershed that intersects the Cretaceous aquifer in south-east Cottonwood County and Jackson County. The simulation ensured that the volume of water in the alluvial aquifer did not change significantly over the simulation period. The average change in aquifer storage was within ± 15% from 1993 to 2014.

5 Sediment Calibration

Sediment calibration follows the sequential procedure outlined in Section 3.3. The observed data sets for calibration contain numerous observations, but in some cases cover only limited time periods (refer to Figure 3-2 for locations). 2002 to 2014 was used as the calibration period, while observations prior to 2002 were used for model validation. The validation exercise was carried out only for locations where data were available prior to 2002.

5.1 Upland Sediment Loading Rates

Detailed studies are not available to identify and quantify specific sediment sources in the Des Moines River watershed; however, lessons learned from model development for other watersheds in the Minnesota Corn Belt are applied here to constrain reasonable estimates of sediment loading (Tetra Tech, 2009; Tetra Tech, 2016). An important issue for sediment calibration is representing the correct division between sediment derived from uplands and sediment derived from reach scour. There have been several studies conducted in the last decade to identify and quantify sediment sources in the Minnesota River Basin. These studies, which are a joint effort between the MPCA and researchers, have applied diverse methods to examine sediment balances in the basin including stream gauging, field measurements, aerial surveying, sediment fingerprinting, and sediment budgeting. Radiometric analysis using ^{210}Pb and ^{10}Be, both of which are derived from the atmosphere and decay over time into more stable forms, has been used to identify the fraction of sediment that derives from upland sources in recent contact with the atmosphere. (Such information is not available for the Des Moines River watershed at this time, but could potentially be used to refine sediment calibration in the future.) Results from these studies were used to guide the upland sediment calibration of the Des Moines River watershed model.

Sediment yields vary significantly across the HUC8 watersheds of the Minnesota River basin due to variations in topography and precipitation, but upland yield from field sources is generally in the range of 100 – 200 lb/ac/yr, with higher loads in the southeastern portion of the basin. Somewhat lower loads are expected in the Des Moines River watershed due to lower average annual precipitation and relatively flat topography.

Average annual simulated upland sediment loading rates by land use are summarized in Table 5-1. The average upland loading rate for the entire Des Moines River watershed from 1995 to 2014 as simulated by the model is approximately 95 lb/acre/yr.

The modeled sediment yields vary significantly by landuse. Cultivated cropland is the major landuse in the watershed covering about 82% of the watershed area and accounts for approximately 89% of the upland sediment load. Tillage practices play a key role in cropland sediment loading. When conventional tillage practices are used on fields with relatively poor drainage (HSG C, D), modeled sediment yields average 322 lb/acre/year. Maintaining crop residue cover of 30% or better when conservation tillage is practiced reduces sediment detachment and delivery to streams. As a result, modeled sediment loading rates are 211 lb/acre/year for conservation tillage HRUs with HSG C, D soils. Much lower rates of sediment loading (below 50 lb/ac/yr) are simulated for crops on well-drained soils. These rates are similar to those obtained from HSPF modeling recently completed for the nearby Redwood and Cottonwood River watersheds (Tetra Tech, 2016)..

Table 5-1. Average Upland Sediment Loading Rates (1994-2014) for the Des Moines River Watershed Model

Land Use	Sediment Yield (lb/ac/yr)
Urban Pervious	103.2
Urban Impervious	353.7
Urban Developed	120.8
Cropland - Conservation Tillage (HSG A,B)	34.8
Cropland - Conservation Till (HSG C,D)	218.4
Cropland - Conventional Till (HSG A,B)	68.6
Cropland - Conventional Till (HSG C,D)	331.3
Cropland - Manure Application (HSG A,B)	26.3
Cropland - Drained	120.9
Pasture	39.8
Forest	15.8
Grassland	19.6
Water/wetland	0

5.2 Reach Sediment Mass Balance

Net sediment scour and deposition was analyzed on a reach by reach basis consistent with recommendations in USEPA (2006) to ensure that significant amounts of scour and deposition occur only in areas where reasonably expected. Summary analysis in terms of changes in nominal sediment bed depth (representing both vertical and lateral changes in this one-dimensional model) is shown in Figure 5-1. For reaches that show a net scour over the simulation time-frame, the average change in bed-depth is -0.21 feet with the 10^{th} and 90^{th} percentile being -0.45 and -0.02 feet, respectively. For reaches that show a net deposition the average change in bed-depth is +0.25 feet with the 10^{th} and 90^{th} percentile being +0.01 and +0.48 feet, respectively.

Most reaches for which net sediment accumulation is simulated area lakes that are explicitly simulated in the model. The majority of free-flowing reaches are simulated as experiencing net degradation over the simulation period (mostly due to bank erosion). This is consistent with the TMDL for the West Fork Des Moines River (MPCA, 2008), which identifies streambank erosion and bed scour as significant sources of phosphorus load.

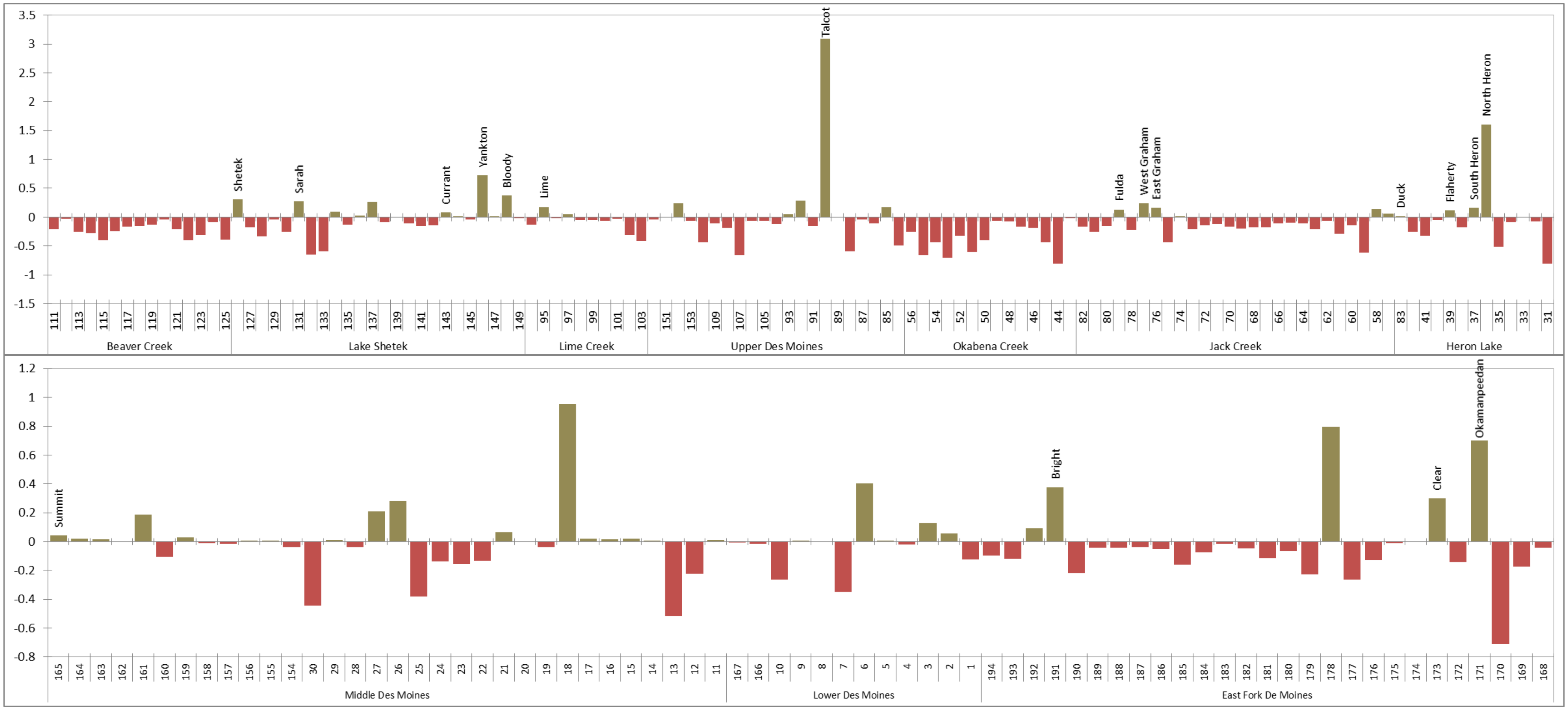

Figure 5-1. Reach Sediment Balance, Des Moines River Watershed Model, 1994-2014

5.3 INSTREAM VERSUS UPLAND SEDIMENT SOURCES

Sediment in the Des Moines River watershed originates from two major source types: Sediment that is detached by rainfall and washed off from upland areas (along with small amounts of gully erosion), and sediment derived from reach bed and bank erosion. The split between upland and instream sources of sediment (prior to accounting for deposition losses in lakes) is shown in Figure 5-2. Approximately 65% of the sediment load in this watershed is simulated as generated from upland sources. (Gully erosion makes up for about 2% of the upland sediment loading.) The remaining 35% of the sediment loading is generated from channel sources. The fractions vary somewhat by individual watershed. For example, the model predicts that approximately 33% and 40% of the total sediment load for the Okabena Creek and Jack Creek watershed comes from channel sources. These total loads are then reduced by the significant amounts of deposition occurring in some of the larger lakes in the watershed.

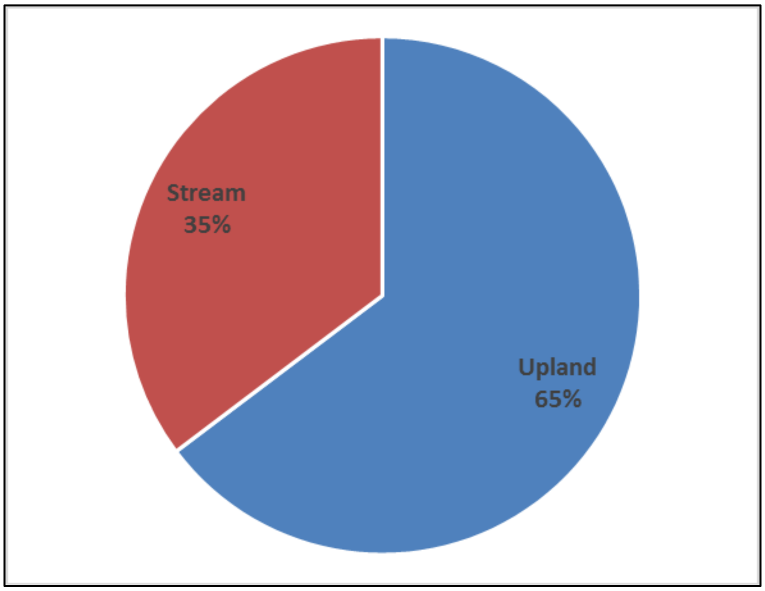

Figure 5-2. Simulated Sediment Sources in the Des Moines River watershed

5.4 CALIBRATION TO OBSERVED TOTAL SUSPENDED SOLIDS DATA

Suspended sediment calibration took place at ten stations and used both visual and statistical approaches. We attempted to replicate the observed time series while at the same time minimizing relative errors associated with both concentration and load (as inferred from concentration and flow). Attention was paid to matching observed and simulated relationships between load and flow through the use of power plots, while also examining the distribution of error terms relative to both season and flow. It is not uncommon for relative error to be strongly leveraged by one or more outliers (especially for load, which tends to be determined by concentrations at high flows); therefore, the median error (which is not sensitive to outliers) is reported as well as the average error.

The detailed calibration process is shown here through an example for the Des Moines River at Jackson monitoring station, while a complete set of graphical and statistical results for all stations is provided in Appendix D. Four years of observations are available at this station. The model appears to track the observed data fairly well, although several high observations are over-estimated by the model. (Figure 5-3). The average and median relative errors on concentration are rated as good based on the criteria in Table 3-4 (9.3% and -8.3%, respectively), while the average and median relative errors on load are 130.7% and -0.4 %, suggesting a discrepancy between model and observations at higher flows. (It is possible that observations at high flows may not accurately capture the total load transported by the river as obtaining representative samples at high flows is challenging and the bedload component is typically omitted.) A log-log power plot (Figure 5-4) shows that the observed and simulated loads have a similar distribution relative to flow; however, the simulation consistently over-estimates in the low flow range which deviates from the observed pattern. This could be due to the presence of three major point sources upstream of this site, which are represented in the model by average monthly loads independent of daily weather and streamflow conditions. The distribution of prediction errors versus flow (Figure 5-5) shows that the errors are generally distributed evenly around zero, although a slight positive bias appears to exist at flows. The outliers noticeable at high flows caused the inflated relative average error on load; however, the overall simulation appears reasonable.

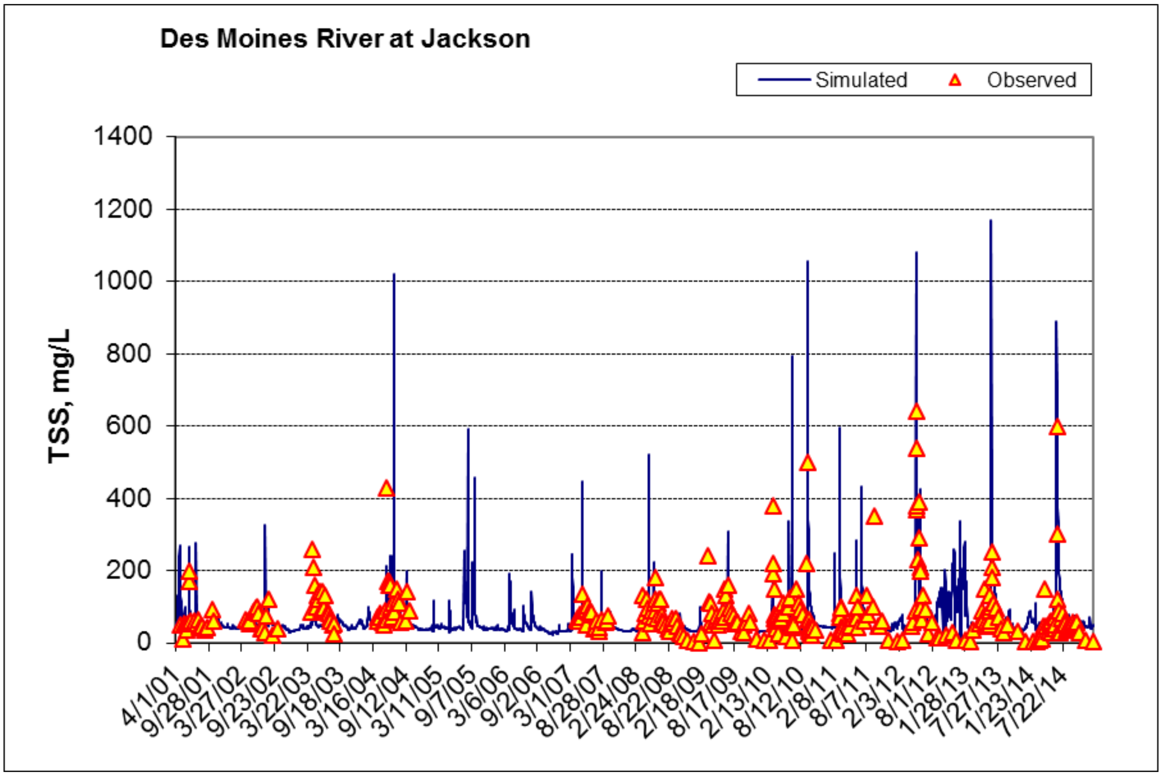

Figure 5-3. Time Series Plot for Total Suspended Sediment, Des Moines River at Jackson

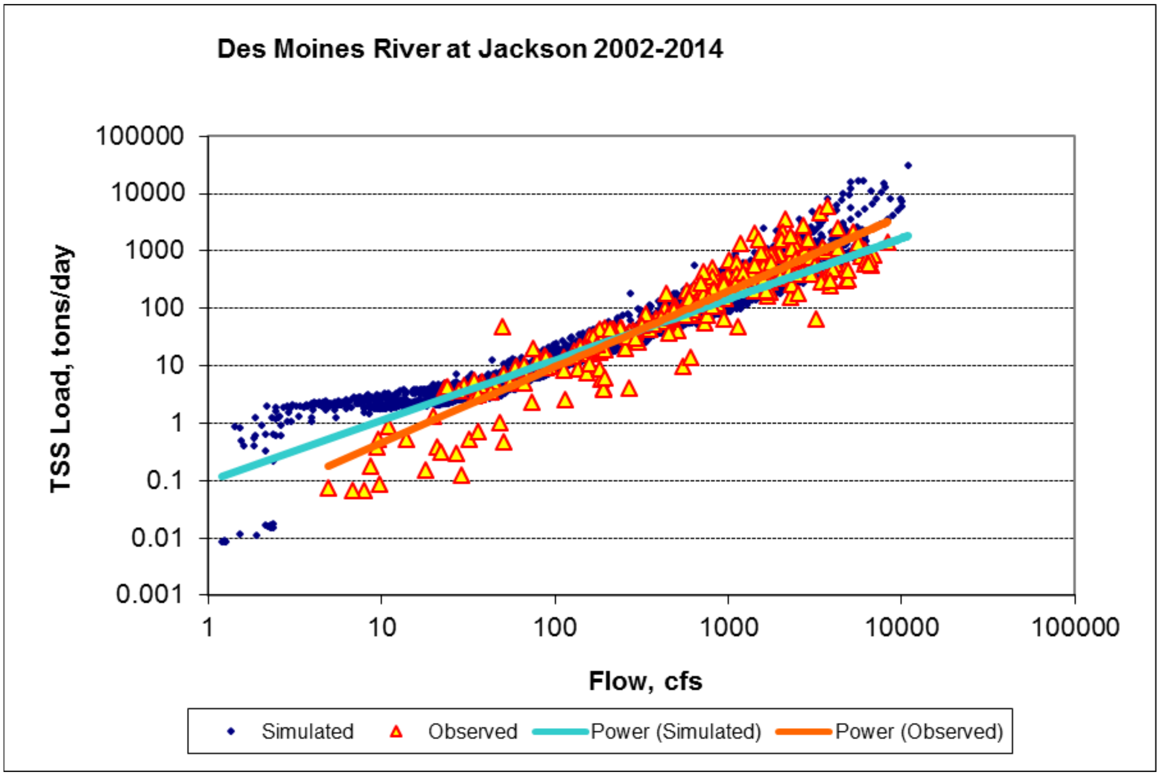

Figure 5-4. Log-log Power Plot of Simulated Total Suspended Sediment Load and Load Inferred from Observed Concentration, Des Moines River at Jackson

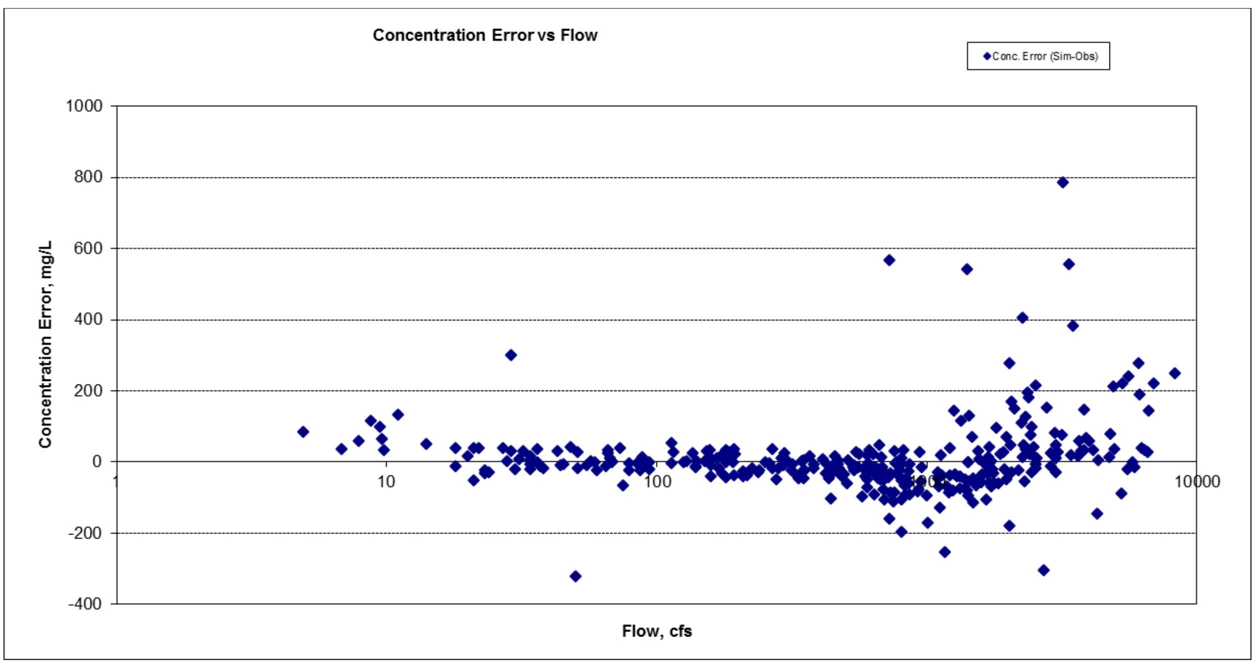

Figure 5-5. Distribution of Concentration Error for Total Suspended Sediment, Des Moines River at Jackson

Sediment calibration proved to be challenging for this watershed. Total suspended solids at a number of tributary stations do not show the expected strong positive correlation with flow, while some stations at the outlet of lakes show a negative correlation (see Figure 5-6 for Heron Lake Outlet). The negative relationship to flow for Heron Lake is in part due to the significant contribution of algal biomass to total suspended solids, which is largest during summer low flow periods of high algal growth, when the load of algal biomass can be of the same magnitude as inorganic sediment (Figure 5-7); however, there may also be other unknown factors that contribute to elevated sediment concentrations at low flows, such as the impact of recreational (and wildlife) activity in the lakes.

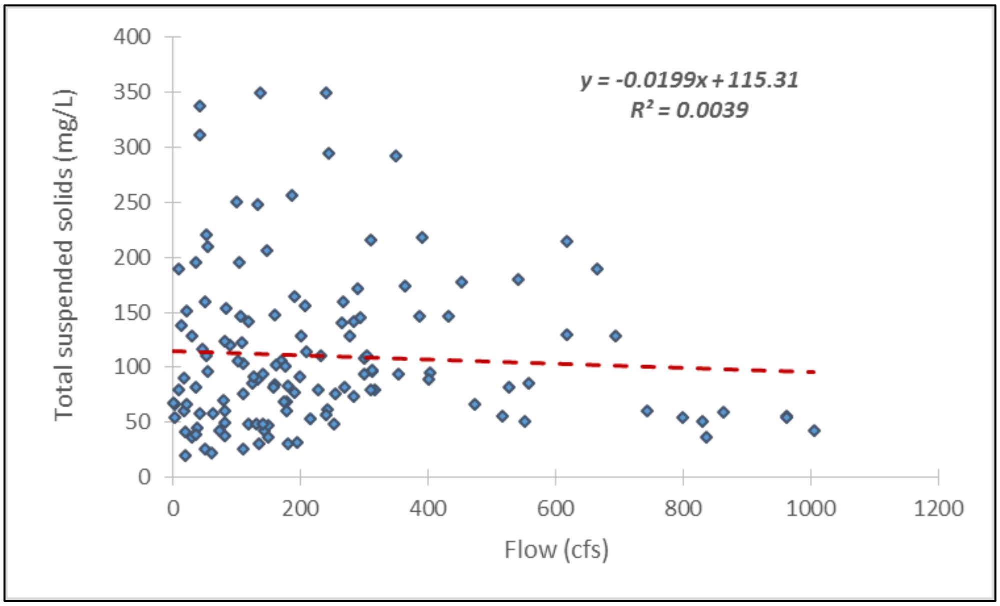

Figure 5-6. Observed Total Suspended Sediment Correlation to Flow at Heron Lake Outlet (S002-009)

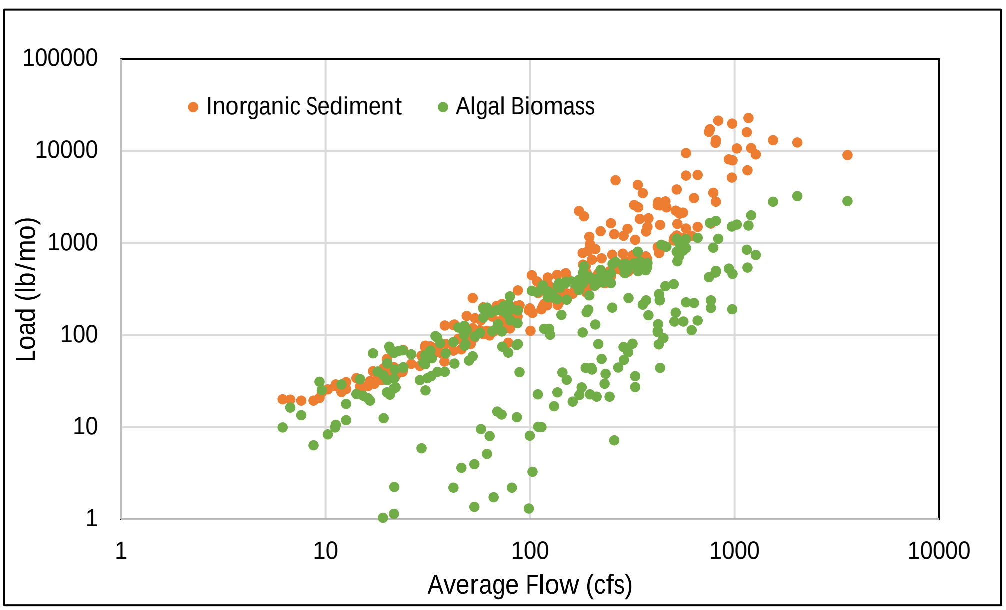

Figure 5-7. Inorganic Sediment and Algal Biomass Components of TSS, Heron Lake Outlet

In addition, the sediment dynamics in lakes in these watersheds are impacted by wind resuspension and dust deposition. Some of these issues have been addressed to a certain extent by using low settling rates for silt and clay, and simulating dust deposition on explicitly simulated lakes. However, the calibration suggests that some of these factors need closer attention to improve model performance.

Nonetheless, given all these challenges, the model performance at the most downstream gage with the longest time-period of observed flow and water quality data seems to perform adequately for sediment.

Summary sediment calibration statistics for all stations are provided in Table 5-2 (the full details are in Appendix D). The fit for concentration is generally within the target range (±25%) for the major stations that have longer periods of record and integrate over larger drainage areas. The fit for average paired loads appears to have a high bias at most locations, while the median relative error on loads is within targets.

Table 5-2. Summary of Sediment Calibration Results

Station	Time-Period	# Samples	Relative Error on Concentration		Relative Error on Loads	
			Average	Median	Average	Median
Beaver Creek (S002-005)	2002-2014	117	-19.09%	-11.20%	27.20%	-0.43%
Lake Shetek (S002-006)	2000-2004,2014	89	26.74%	11.40%	79.70%	4.39%
Jack Creek (S001-557)	2002-2014	256	-20.28%	-16.56%	69.46%	-9.51%
Okabena Creek (S001-568)	2002-2014	309	-23.40%	-28.71%	51.91%	-6.70%
Heron Lake Outlet (S002-009)	2002-2014	260	-6.29%	-22.55%	123.01%	-18.12%
W Fork Des Moines River, Avoca (S002-008)	2001-2004	79	-36.89%	-27.48%	46.96%	-5.16%
W Fork Des Moines River, Windom (S000-894)	2001-2004	79	2.15%	-14.36%	59.44%	-2.24%
W Fork Des Moines River, Jackson (S000-027,S004-359,S005-936)	2002-2014	303	9.29%	-8.25%	130.73%	-0.37%
Martin County Ditch (S005-027)	2008-2014	80	-12.56%	-19.87%	23.25%	-0.91%
Fourmile Creek (S005-572)	2009-2010	54	-2.56%	-21.68%	92.97%	-1.45%

The model performance for sediment in the validation period (Table 5-3) is generally poor for stations with available data. In contrast to the calibration period, loads at several stations are under-estimated relative to the observations from the validation data. Note that during the validation period continuous flow data is generally lacking except for the water quality station at Jackson, so the model performance for flow in this period cannot be tested. The sample sizes are also relatively small, which can lead to undue influences on average relative errors by a small number of outliers. The model performance based on median relative errors on load is rated as very good.

Table 5-3. Summary of Sediment Validation Results

Station	Time-Period	# Samples	Relative Error on Concentration		Relative Error on Loads	
			Average	Median	Average	Median
Beaver Creek (S002-005)	2001-2002	33	-47.96%	-13.85%	-63.01%	-1.42%
Lake Shetek (S002-006)	No Data					
Jack Creek (S001-557)	1997-2002	33	-67.52%	-52.54%	-53.83%	-9.69%
Okabena Creek (S001-568)	1997-2002	39	-53.14%	-43.96%	-25.26%	-3.73%
Heron Lake Outlet (S002-009)	1997-2002	46	-34.08%	-26.39%	21.13%	-14.75%
W Fork Des Moines River, Avoca (S002-008)	No Data					
W Fork Des Moines River, Windom (S000-894)	No Data					
W Fork Des Moines River, Jackson (S000-027,S004-359,S005-936)	2001-2002	33	28.12%	1.75%	94.79%	0.10%
Martin County Ditch (S005-027)	No Data					
Fourmile Creek (S005-572)	No Data					

Some notes regarding individual stations are provided below:

Beaver Creek: This is one of the few locations in the watershed not impacted by large lakes; however, the relatively high stream chlorophyll *a* concentration reported for this site (based on a limited number of observations) suggests that the observed suspended solids concentrations are still impacted by algal biomass.

Lake Shetek: This station drains the Lake Shetek complex consisting of Lake Shetek, Bloody Lake, and Sarah Lake. The observed total suspended solids concentrations do not show a correlation with flow. The model was largely calibrated at this location by reducing the silt and clay settling rates in the lakes. The chlorophyll *a* concentration reported for these lakes are consistently in the 100 μg/L range suggesting that algal biomass has an impact on the total suspended solids concentration. Lake Shetek is also a shallow lake likely impacted by wind resuspension and recreational activities, which are difficult to represent in the HSPF model as a mechanistic process.

Jack Creek: The Jack Creek gage also does not show a strong correlation of suspended sediment with flow, with low flow concentrations in the 100 mg/L range and gradually falling off to approximately 10 mg/L at high flows. The sediment (and water quality) response at this location is impacted by two relatively large lake complexes: Fulda (First and Second) and Graham (East and West). The high reported chlorophyll *a* concentrations for the lakes and the stream suggest that algal biomass has skewed the observed data at low to mid-range flows.

Okabena Creek: The suspended solids data at this location shows a weak positive correlation with flow, but observed concentrations show a downward trend in the high flow range. This location is also impacted by two large point sources discharges (Worthington WWTP and Worthington Industrial WWTP).

Heron Lake Outlet: This water quality monitoring location is directly downstream from the Heron Lake complex consisting of the South and North Heron, Duck and Flaherty lakes. Okabena Creek and Jack Creek drain into Heron Lake. As discussed above, the total suspended solids at this location is negatively correlated with flow. The in-lake TSS concentrations observed close to the outlet are also high, supporting the theory that algal dynamics and wind resuspension have significant impacts.

West Fork Des Moines River, Avoca: The reported total suspended solids concentrations at this location again do not show a strong correlation with flow, suggesting that organic solids from the lakes are an important component of total suspended solids at lower flows, while channel erosion is more important during high flow events. This location is downstream of the Lake Shetek complex and includes the Beaver Creek drainage. The discussion for Beaver Creek and Lake Shetek generally apply to this location as well.

West Fork Des Moines River, Windom: This station integrates the outflow from the Heron Lake watershed and Talcot Lake. The Talcot Lake drainage also includes Lime Lake. Both Lime and Talcot have reported growing season chlorophyll *a* concentrations exceeding 100 µg/L. The observed suspended solids behavior is similar to that observed for the West Fork Des Moines River at Avoca. The distribution of concentration error versus flow does not show an apparent bias at high flows. The low to mid-range flow concentrations are however under-predicted.

Martin County Ditch: This water quality station is located in the headwaters of the East Fork Des Moines River. This location is relatively free from the impacts of lakes. There are however two minor point source dischargers that likely have some impacts on low flow concentrations. The model performance at this location is generally good.

Fourmile Creek: This location drains a relative small area that is free from the impacts of point sources or lakes. The model performance is generally good based on relative errors on concentration. The relative average error on load is impacted by the presence of a few outliers. The model performance based on relative median error on load is very good.

5.5 Comparison to FLUX Load Estimates

A final check on the sediment calibration is comparison of simulated loads to loads estimated from observed flow and concentration data. MPCA's Watershed Pollutant Load Monitoring Network (WPLMN) is designed to obtain spatial and temporal pollutant load information from Minnesota's rivers and streams and track water quality trends. As part of this program, MPCA releases estimates of annual pollutant loads for each 8-digit hydrologic unit code basin. These "observed" monthly loads are estimated using the USACE FLUX32 program (a Windows-based update of the FLUX program developed by Walker, 1996; available at https://www.pca.state.mn.us/water/watershed-pollutant-load-monitoring-network#flux32-8f1620f5), and are themselves subject to significant uncertainty.

The West Fork Des Moines River at Jackson is the only location in the watershed for which FLUX estimates of long-term mass loading rates have been developed. A comparison shown in Table 5-4 and Figure 5-8. For this location, the average simulated load over the period of five years is similar to the FLUX-estimated load, thus confirming the model calibration. The HSPF simulated loads are higher than the FLUX-estimated loads for 2010 and 2011 and lower than the FLUX-estimated loads from 2007 to 2009.

Table 5-4. Comparison of Simulated and FLUX-Estimated Sediment Loads

Year	Simulated Load (t/yr)	FLUX Load (t/yr)
2007	46,318	46,804
2008	25,338	40,770
2009	17,166	27,893
2010	146,549	94,333
2011	89,861	67,957
Average	**65,046**	**55,551**

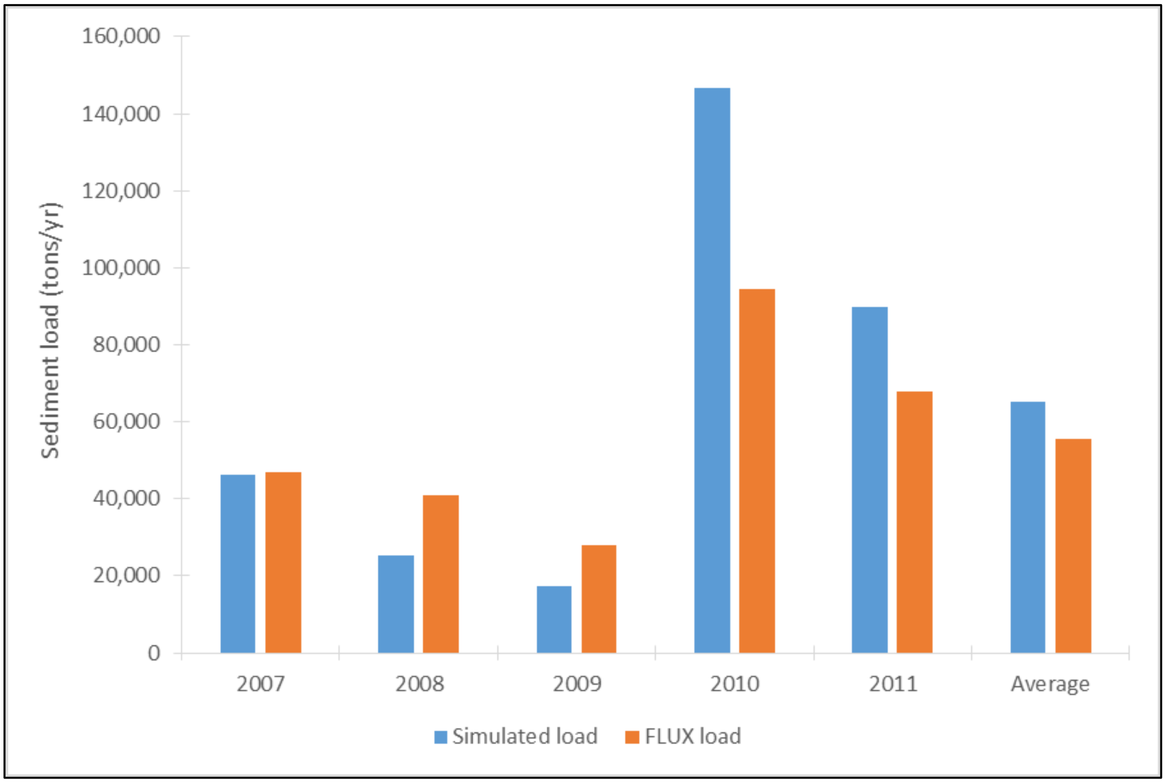

Figure 5-8. Annual HSPF Simulated TSS Load and FLUX Estimated Load, Des Moines River at Jackson

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6 Nitrogen and Phosphorus Calibration

6.1 NUTRIENT MODEL SETUP

The nutrient simulation follows the same general approach used in other Minnesota HSPF models and recommended by AQUA TERRA (2012). Ammonia, nitrate nitrogen, orthophosphate, and generalized organic matter are simulated on the land surface, with the first two being represented by buildup-washoff processes and the second two simulated as sediment-associated using potency factors for pervious land (with a buildup-washoff approach for impervious land). Representation of point source loads of nutrients is described in Section 6.1.2. Full nutrient kinetics are represented instream, including the decay of organic matter, uptake by and release from planktonic and benthic algae, nitrification, denitrification, exchanges with the sediment bed, and sorption to sediment of ammonium and ortho-phosphate.

6.1.1 Nonpoint Sources

The nutrient simulation for the uplands represents inorganic nitrogen, inorganic phosphorus, and organic matter as three distinct constituents. Inorganic phosphorus and organic matter on pervious surfaces are simulated using a sediment potency approach, while inorganic nitrogen on pervious surfaces and all three constituents on impervious surfaces are represented as a buildup/washoff process. Concentrations associated with subsurface flows are also included.

Within the stream reaches the model represents individual nutrient species (ammonia, nitrate, nitrite, organic nitrogen, orthophosphate, organic phosphorus, and organic carbon/BOD). The stream reach module is implemented with full nutrient simulation, including uptake by and release from plankton and benthic algae, decay of organic matter, oxidation of ammonium to nitrite and nitrite to nitrate nitrogen, bed exchanges of dissolved and sorbed nutrients, and ammonia volatilization.

The key parameters controlling the upland nutrient simulation are listed below:

MON-ACCUM: The monthly varying assignment of the build-up or accumulation of a constituent on a particular surface (lb/ac-d).

MON-SQOLIM: The monthly varying upper limit value beyond which a constituent can no longer accumulate on a surface (lb/ac).

MON-IFLW-CONC and **MON-GRND-CONC**: These parameters are used to assign the interflow and groundwater constituent concentrations on a monthly basis. The values for these parameters were estimated from the observed data with consideration of flow regime and then calibrated as necessary.

MON-POTFW: The monthly varying specification of constituent mass per sediment mass (lb/ton). For organic matter the assigned values were around 10^0 to 10^1. The seasonal assignment for organic matter reflects the annual cycle of growth and then litter.

The sediment potency, build-up/washoff, and subsurface flow parameters were initialized for the Des Moines River watershed model based on past experience with models for the Minnesota River basin. A literature review was conducted to establish appropriate ranges for unit-area loading rates of the diverse land use categories found in the watersheds (Table 6-1). The simulated unit-area loading rates were compared to the literature-based ranges and the surface and subsurface flow parameters were revised until reasonable loading estimates were established for TN and TP. Results for the Des Moines River watershed were aggregated and are provided in Figure 6-1 and Figure 6-2.

The mean simulated TN unit loading rate for the cropland category in the Des Moines River watershed is 26.3 lb-N/ac/yr, which is slightly higher than the range reported in Table 6-1 but consistent with the elevated nitrate concentrations observed throughout the watershed. A large proportion of this load (approximately 78%) is nitrite + nitrate nitrogen. In part this is due to tile drainage. In addition, as nitrate concentrations in streams are influenced by groundwater loading and this region is known to have elevated nitrate concentrations in groundwater wells (22 mg/L reported in the only MNDNR observation well in the watershed), a high nitrate loading is appropriate. The developed pervious and impervious mean simulated values are 5.8 lb-N/ac/yr and 14.6 lb-N/ac/yr, respectively. These results are similar to the values reported by the MPCA, which range from 2-17 lb-N/ac/yr for mixed developed land use (MPCA, 2013). The mean simulated TN unit loading rate from grassland and water/wetland land uses are within the literature suggested ranges. The loading rate for pasture and forest land uses are lower than the literature suggested ranges but are expected to have minimal impact on loading since they cover a small fraction of the watershed area.

The simulated TP unit loading rate for the cropland category is 0.77 lb-P/ac/yr. This is approximately in the middle of the range reported in Table 6-1. The simulated dissolved ortho-phosphate fraction is slightly higher than the organic fraction for the cropland category. This is expected because a large amount of the cropland uses artificial drainage which is expected to facilitate ortho-phosphate transport out of the soil. The simulated TP loading rate for developed pervious and impervious areas are 0.232 and 1.011 lb-P/ac/yr, respectively. The impervious loading is on the higher end of the reported range. A higher loading from impervious areas is expected due to atmospheric deposition of phosphorus. The TP loading rates from urban pervious land are lower than the reported range largely as a result of an effort to maintain the stoichiometry between nitrogen, phosphorus, and carbon in organic matter. The simulated TP unit loading rates for pasture, grassland, pasture and water/wetland land uses are within the ranges reported in the literature.

Table 6-1. Reference Ranges for the Nutrient Loading Rates of Diverse Land Use Categories

Land Use	TN (lb-N/ac/yr)	TP (lb-P/ac/yr)	References
Forest	1.97 – 4.2	0.05 – 5	Clesceri et al., 1986; Loehr et al., 1989; MPCA, 2013, MPCA, 2004; Reckhow et al., 1980
Wetland	0.5 – 5	0	MPCA, 2013; MPCA, 2004
Pasture	6.1 – 23	0.11 – 0.43	Clesceri et al., 1986; McFarland and Hauck, 2001; MPCA, 2013; MPCA 2004
Crop	7.5 – 23	0.11 – 1.7	Dodd et al., 1992; Clesceri et al., 1986; Loehr et al., 1989, MPCA, 2013; MPCA 2004
Developed (pervious)	2 – 17	0.8 – 1.02	Loehr et al., 1989; MPCA, 2013; MPCA, 2004; Reckhow et al., 1980
Developed (impervious)	2 – 17	0.8 -1.02	Loehr et al., 1989; MPCA, 2013; MPCA, 2004; Reckhow et al., 1980
Barren	0.5 - 5	ND	MPCA, 2013
Shrub	0.5 - 5	0.05 – 0.12	MPCA, 2013; MPCA, 2004

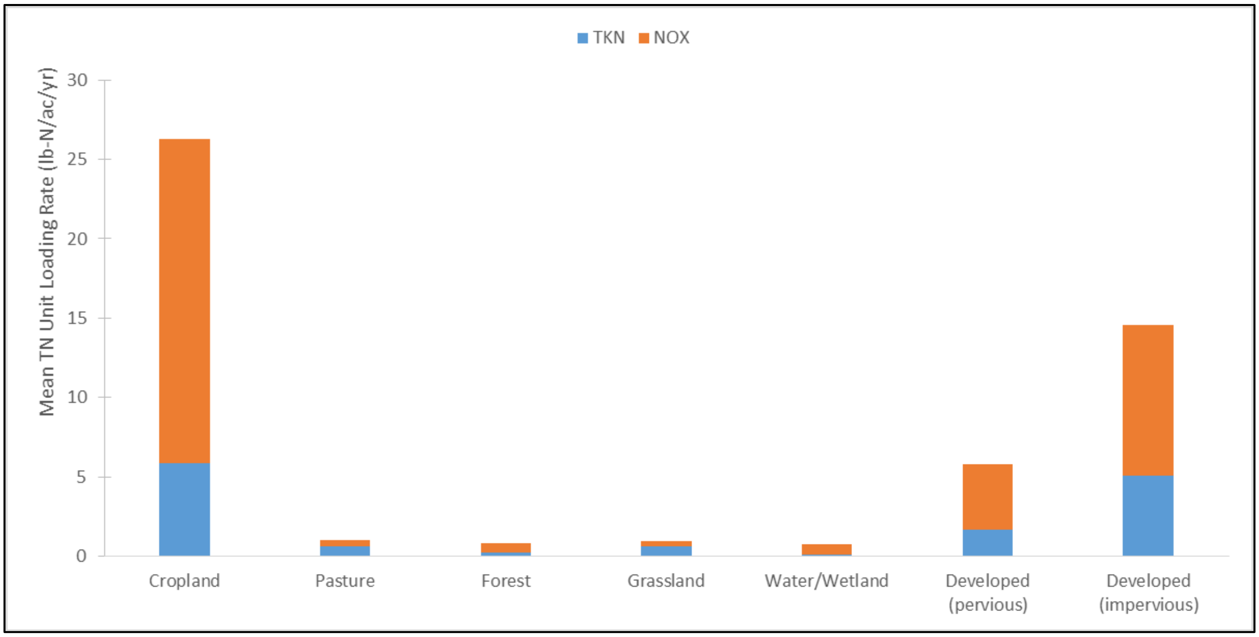


Figure 6-1. Average Simulated TN Loading Rates for Land Use Categories in the Des Moines River Watershed

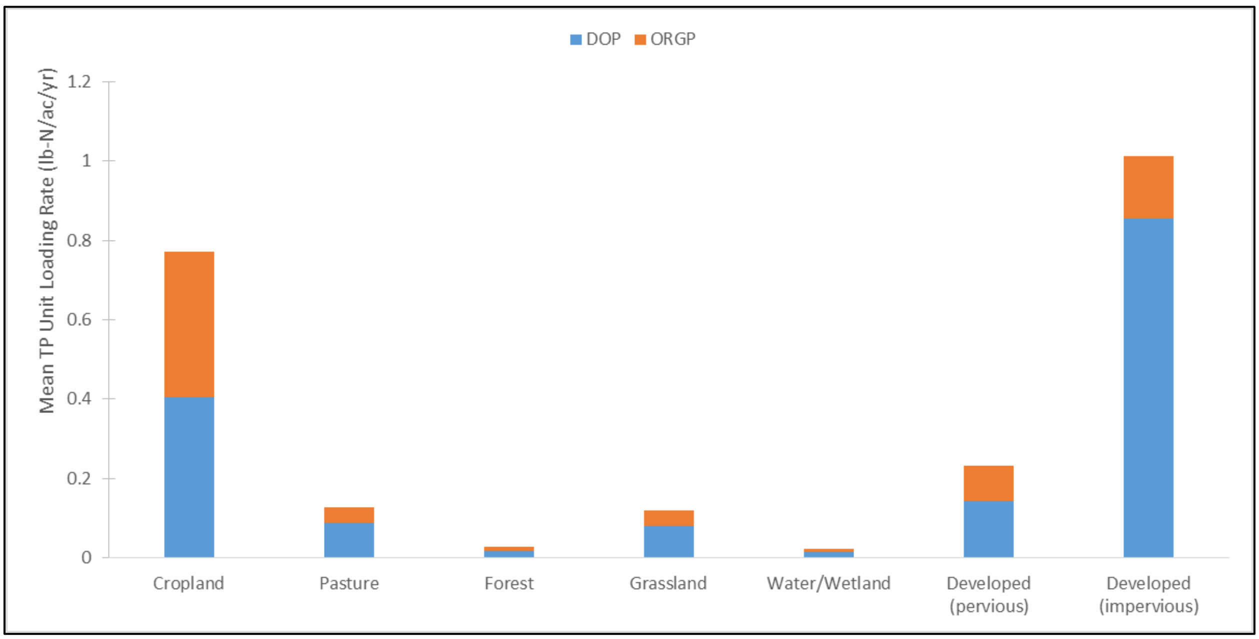


Figure 6-2. Mean Simulated TP Loading Rates for Land Use Categories in the Des Moines River Watershed

6.1.2 Point Sources

Point sources that have discharge permits in the Des Moines River watershed are included in the modeling framework. The Des Moines River model incorporates three major WWTP discharges and eighteen minor point sources. The minor point sources are generally stabilization ponds that discharge a few months every year, although there are some that discharge continuously. The point sources included in the model are documented in Section 2.4 of this report.

Loads from point sources are based reported on monthly flow and load series. Wastewater treatment plant discharges generally have monitored and reported effluent total phosphorus and ammonia concentrations in the Discharge Monitoring Reports (DMRs) commencing in 1995. Effluent nitrite + nitrate (NO_2 +NO_3-N), organic nitrogen, and phosphorus species were generally not reported. Discharge concentrations of these nutrients were based on the type of discharger being represented (e.g. small municipal, industrial, etc.). Mean nutrient concentrations simulated for the point sources are provided in Table 6-2.

Table 6-2. Average Modeled Discharge Concentrations (mg/L) for Point Sources in the Des Moines River Watershed, 1995-2012

Discharge (Major Source or Minor Type)	NH_4-N	NO_2 +NO_3-N	Organic N	TP	PO_4-P	Organic P	CBOD
Windom WWTP.	0.85	37	2.36	7.58	6.82	0.76	3.35
Worthington WWTP	0.28	15	3.58	1.71	1.55	0.16	3.15
Worthington Ind. Ponds	1.36	15	2.6	1.45	1.10	0.34	7
Worthington Ind. Sludge	0.29	100	2.6	10.86	10.46	0.40	3.15
Minor Municipal WWTPs	DMRs	7	2	DMRs	TP is 72.3% PO_4		DMRs
Minor WWTP (Ponds)	DMRs	7	2	DMRs	TP is 59.2% PO_4 Spring TP is 83.5% PO_4 Fall		DMRs
Minor Industrial	1	7	2	DMRs	TP is 72.3% PO_4		DMRs

Notes: Total phosphorus (TP) concentrations for WWTPs are based on discharge monitoring reports (DMRs), while the split between phosphorus species is based on fixed ratio assumptions. Nitrate plus nitrite N concentrations for minor municipal WWTPs are not monitored and are based on MPCA summaries by source type. Very limited nitrate plus nitrite N monitoring is available for the Worthington Industrial sludge discharge. Limited monitoring by MPCA in 2011-2012 showed concentrations in the range of 150 mg/L N. A value of 100 mg/L was assumed for the long term based on comparison to instream monitoring in Okabena Creek.

6.1.3 Channel Sources

Nutrients can be gained or lost through exchanges with the sediment bed – either through releases in the dissolved form or by scour or deposition of nutrients that sorb to sediment. HSPF simulates ortho-phosphate and ammonia as sorbing to sediment and also represents release of dissolved ortho-phosphate, ammonia, and labile organic matter (as BOD, with associated nutrients) from the sediment.

Based on past experience, adsorption coefficients were set for ortho-phosphate as 1,000 ml/g relative to silt and clay and 100 ml/g relative to sand; the corresponding numbers for total ammonia N were 100 and 10 ml/g. Default background sediment bed concentrations for ortho-phosphate are set at 350 mg/kg for silt and clay and 50 mg/kg for sand, and, for total ammonia N, 80 mg/kg for silt and clay and 40 mg/kg for sand.

6.1.4 Atmospheric Deposition

The model simulates wet and dry deposition of ammonia-N and nitrate-N to pervious surfaces, impervious surfaces, and water bodies. In addition, both dry and wet deposition of phosphorus to water is simulated. Atmospheric deposition of phosphorus to the uplands is not simulated because it is implicit in the sediment potency representation of pervious land loading and the buildup/washoff representation of impervious land loading of phosphorus.

Direct phosphorus deposition to surface water is represented in the model. The phosphorus dry deposition rate is specified as 0.270 kg/ha/yr, based on the 2007 update to *Detailed Assessment of Phosphorus Sources to Minnesota Watersheds - Atmospheric Deposition* (Twaroski, et al., 2007. The wet deposition concentration for phosphorus is set at the average concentration for Lamberton, MN (just northeast of the Des Moines River watershed) of 26 µg/L given in the same resource.

Wet deposition concentrations of ammonia and nitrate N (as mg/L) are taken from monthly data recorded at the National Atmospheric Deposition (NADP) station MN27 (Lamberton). Dry deposition rates of ammonia and nitrate N (as lb/ac) are taken from EPA Clean Air Status and Trends (CASTNET) monitoring (https://www.epa.gov/castnet). There are not CASTNET stations within or particularly close to the watersheds studied here, so we use the station at Santee Sioux, in northeast Nebraska (SAN189) for the period after 7/5/2006, filling in earlier dates with monitoring from Perkinstown, WI (PRK134). In all cases, reported data were converted from molar units to mass or mass-based concentration as N.

6.1.5 Regeneration from Lake Sediment

The Des Moines River watershed contains a series of highly eutrophic lakes with high concentrations of nutrients and algae. These lakes appear to have accumulated large stores of nutrients and organic matter over time. While the lakes are predominantly a sink for nutrients, they can also serve as a source of nutrients. In the shallow parts of lakes, turbulence from wind-induced waves or recreational boating can stir up nutrient-laden sediment from the lake bottom. Where the lake water:sediment interface is depleted in oxygen (as could occur during summer thermal stratification or under winter ice cover), phosphorus that is deposited as insoluble complexes with iron hydroxide can become soluble and diffuse back into the water column. Both phosphorus and nitrogen compounds can also be generated from the decomposition of organic matter on the lake bottom.

The Des Moines River watershed model approximates these processes through specification of benthic release rates of ortho-phosphate and ammonia; however, a more precise representation would require a more complex lake model that simulates vertical stratification and turbulent erosion processes. Management of nutrient levels downstream of major lakes may require management of nutrient regeneration in the lakes.

6.2 Nutrient Calibration

Nutrients from point and nonpoint sources are loaded to the stream reaches. Within the stream reaches the model represents the following nutrient species: ammonia, nitrite, nitrate, organic nitrogen, orthophosphate, organic phosphorus, and organic carbon/BOD. The stream reach module simulates instream biogeochemical processes including nutrient uptake and release by plankton and benthic algae, decay of organic matter, nitrification/denitrification, absorption/desorption of nutrients on suspended sediment, and deposition and scour of sediment-stored nutrients.

The nutrient calibration and validation relies on a weight of evidence approach. Upland loading rates are constrained to be in general agreement with literature values (as described in Section 6.1.1), while point source discharges are based on monitoring or recommended assumptions for unmonitored parameters provided by MPCA. Model calibration then adjusts parameters to optimize the fit between model predictions and observations at multiple stations throughout the watershed and the robustness of the fit is checked with validation tests on a different time period. Model performance is then checked against other sources of information, including information developed by MPCA on delivered loads and lake phosphorus balances.

6.2.1 Comparison of Model to Observations

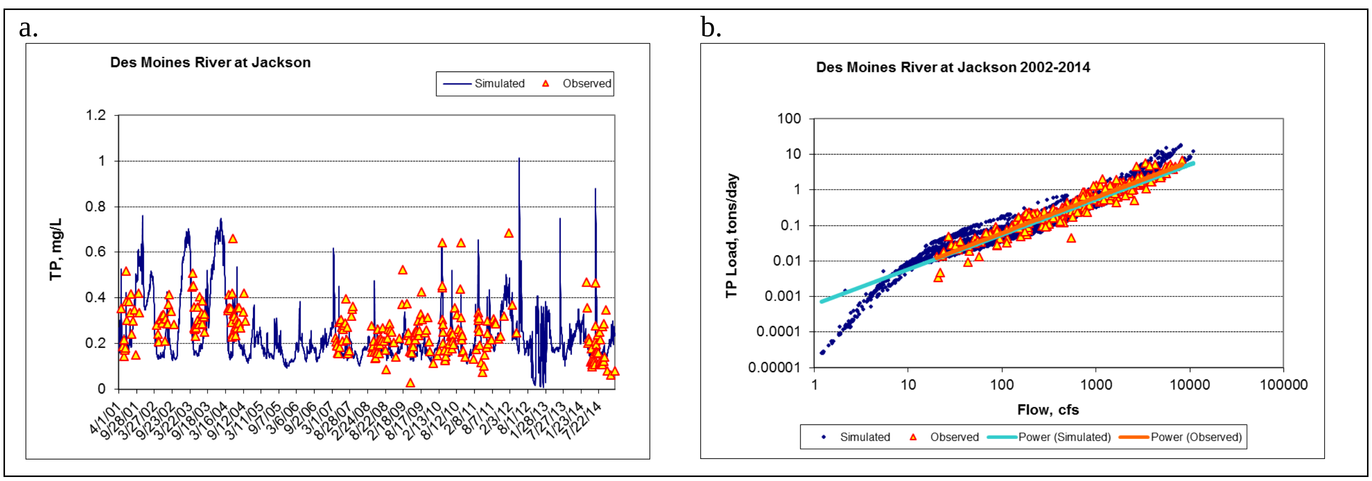
Comparisons between model predictions and sample observations are made in terms of both concentration and inferred load (concentration times simulated or observed flow). Complete graphical and tabular statistical results for each station are provided in Appendix D. Figure 6-3 provides an example of the primary types of calibration plots provided for each monitored nutrient parameter at each site, in this case showing the total phosphorus calibration for the West Fork Des Moines River at Jackson. The four panels in Figure 6-3 are:

a. Standard time series plot, showing the observations and continuous model predictions of daily average concentrations. This shows general agreement, but can obscure biases in the simulation.

b. A power plot comparing the relationship of observed and simulated loads versus flow. The objective here is that the relationship to flow (summarized by the power regression lines) should be similar for the model and observations.

c. A scatterplot of simulated versus observed concentrations shows the degree of spread or uncertainty about the 1:1 line.

d. A plot of the residuals against flow is used to diagnose bias relative to the flow regime. In this case there is a fair balance between over and under-prediction across the range of flows, but some indication of a tendency to over-predict concentrations at the highest flows. A similar plot of residuals versus month is used to diagnose potential seasonal biases.



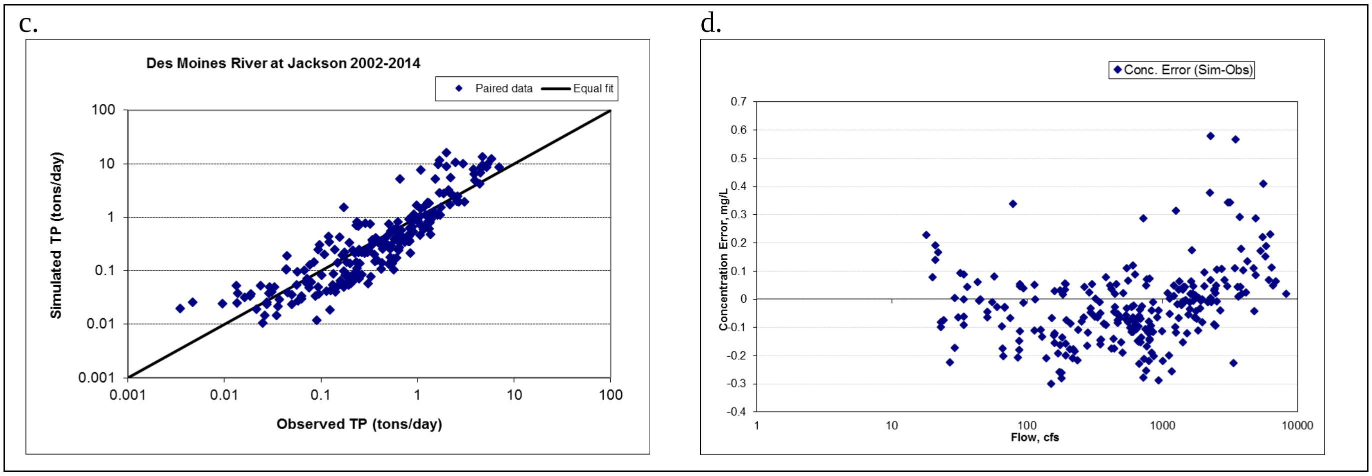


Figure 6-3. Example Calibration Plots for Total Phosphorus, West Fork Des Moines River at Jackson

This section first provides an overview of the results with a focus on total phosphorus, total nitrogen, and nitrate + nitrite nitrogen (nitrate nitrogen is included in the overview because it is often the predominant form of nitrogen and the number of observations for total nitrogen is limited at many stations). Results for individual nutrient species are then summarized, with full results provided in Appendix D.

Summary statistics for the calibration and validation of total phosphorus, total nitrogen, and nitrate + nitrite nitrogen at all stations are provided in Table 6-3 through Table 6-8. Discussion by parameter and individual monitoring site follows the tables.

A majority of the stations are affected by nutrient processes in lakes and several are downstream of the large point sources in Worthington and Windom for which we have imprecise specification of loading rates. Nonetheless, the performance of the model is generally acceptable.

Table 6-3. Summary Statistics for Total Phosphorus Calibration

Station	Time Period	# Samples	Relative Error on Concentration		Relative Error on Loads	
			Average	Median	Average	Median
Beaver Creek	2002-2014	118	28.86%	34.69%	16.88%	7.02%
Lake Shetek	2000-2004,2014	91	21.32%	34.04%	16.09%	12.77%
Jack Creek	2002-2014	256	-8.41%	-3.64%	10.31%	-1.32%
Okabena Creek	2002-2014	309	25.81%	14.73%	-18.94%	4.98%
Heron Lake Outlet	2002-2014	259	-30.99%	-13.58%	10.22%	-10.98%
W Fork Des Moines River, Avoca	2001-2004	80	-12.88%	-4.36%	12.86%	-0.68%
W Fork Des Moines River, Windom	2001-2004	80	-20.39%	-31.10%	-0.72%	-15.16%
W Fork Des Moines River, Jackson	2002-2014	231	-8.94%	-11.06%	53.11%	-5.06%
Martin County Ditch	2008-2014	79	38.87%	14.43%	-38.88%	4.34%
Fourmile Creek	2009-2010	55	35.46%	23.72%	-25.69%	2.20%

Table 6-4. Summary Statistics for Total Phosphorus Validation

Station	Time Period	# Samples	Relative Error on Concentration		Relative Error on Loads	
			Average	Median	Average	Median
Beaver Creek	2001-2002	34	2.33%	23.17%	-29.84%	2.74%
Lake Shetek	No Data					
Jack Creek	1997-2002	34	-60.48%	-42.28%	-53.68%	-10.81%
Okabena Creek	1997-2002	40	127.53%	27.00%	-42.33%	13.24%
Heron Lake Outlet	1997-2002	47	10.69%	-17.25%	-37.97%	-23.43%
W Fork Des Moines River, Avoca	No Data					
W Fork Des Moines River, Windom	No Data					
W Fork Des Moines River, Jackson	2001-2002	34	-15.39%	-22.80%	4.46%	-11.07%
Martin County Ditch	No Data					
Fourmile Creek	No Data					

Table 6-5. Summary Statistics for Total Nitrogen Calibration

Station	Time Period	# Samples	Relative Error on Concentration		Relative Error on Loads	
			Average	Median	Average	Median
Beaver Creek	2002-2014	97	-2.44%	-7.13%	30.21%	0.08%
Lake Shetek	2000-2004,2014	62	-11.57%	-7.24%	7.44%	0.00%
Jack Creek	2002-2014	220	-8.26%	-11.43%	4.30%	-10.13%
Okabena Creek	2002-2014	273	-1.63%	2.72%	7.78%	1.82%
Heron Lake Outlet	2002-2014	222	-14.23%	-12.92%	2.75%	-14.32%
W Fork Des Moines River, Avoca	2001-2004	74	-13.40%	-17.07%	23.68%	-5.33%
W Fork Des Moines River, Windom	2001-2004	71	-22.90%	-19.90%	-0.39%	-11.42%
W Fork Des Moines River, Jackson	2002-2014	276	-11.69%	-9.82%	33.49%	-0.95%
Martin County Ditch	2008-2014	10	18.41%	25.59%	-4.39%	6.06%
Fourmile Creek	No Data					

Table 6-6. Summary Statistics for Total Nitrogen Validation

Station	Time Period	# Samples	Relative Error on Concentration		Relative Error on Loads	
			Average	Median	Average	Median
Beaver Creek	2001-2002	29	-2.63%	-6.81%	23.18%	-3.37%
Lake Shetek	No Data					
Jack Creek	1997-2002	13	-33.89%	-23.08%	-17.58%	-14.77%
Okabena Creek	1997-2002	15	4.91%	-2.22%	-1.03%	-0.37%
Heron Lake Outlet	1997-2002	27	-45.28%	-41.15%	-23.61%	-16.39%
W Fork Des Moines River, Avoca	No Data					
W Fork Des Moines River, Windom	No Data					
W Fork Des Moines River, Jackson	2001-2002	27	-7.86%	-16.44%	13.26%	-15.42%
Martin County Ditch	No Data					
Fourmile Creek	No Data					

Table 6-7. Summary Statistics for Nitrate + Nitrite-N Calibration

Station	Time Period	# Samples	Relative Error on Concentration		Relative Error on Loads	
			Average	Median	Average	Median
Beaver Creek	2002-2014	97	-8.96%	-15.97%	21.94%	-2.71%
Lake Shetek	2000-2004,2014	62	-16.08%	-3.59%	12.16%	0.00%
Jack Creek	2002-2014	221	-11.43%	-16.07%	-0.88%	-11.99%
Okabena Creek	2002-2014	274	-5.73%	-0.81%	5.12%	-0.21%
Heron Lake Outlet	2002-2014	222	-7.79%	-4.07%	3.42%	-10.01%
W Fork Des Moines River, Avoca	2001-2004	74	-18.70%	-24.06%	21.31%	-8.16%
W Fork Des Moines River, Windom	2001-2004	71	-19.37%	-22.24%	-3.33%	-8.06%
W Fork Des Moines River, Jackson	2002-2014	276	-8.26%	-1.64%	25.10%	0.63%
Martin County Ditch	2008-2014	81	10.37%	-0.33%	-13.59%	0.36%
Fourmile Creek	2009-2010	55	3.51%	-1.03%	6.77%	2.00%

Table 6-8. Summary Statistics for Nitrate + Nitrite-N Validation

Station	Time Period	# Samples	Relative Error on Concentration		Relative Error on Loads	
			Average	Median	Average	Median
Beaver Creek	2001-2002	29	-10.08%	-13.74%	22.05%	-8.14%
Lake Shetek	No Data					
Jack Creek	1997-2002	14	-40.53%	-38.76%	-25.18%	-19.04%
Okabena Creek	1997-2002	18	-1.11%	-7.10%	-10.33%	-8.16%
Heron Lake Outlet	1997-2002	27	-39.14%	-15.61%	-16.82%	-6.62%
W Fork Des Moines River, Avoca	No Data					
W Fork Des Moines River, Windom	No Data					
W Fork Des Moines River, Jackson	2001-2002	27	-2.51%	-16.86%	8.56%	-10.66%
Martin County Ditch	No Data					
Fourmile Creek	No Data					

Total Phosphorus: The relative average errors on concentration during calibration are ranked as "good" or "very good" (based on the criteria in Table 3-5) at 5 out of 10 sites, while the relative average errors on load meet these standards at 7 out of 10 sites. Apparent poor performance on the simulation of load during calibration for West Fork Des Moines River at Jackson seems to be due to a limited number of high outliers, as discussed further below. The validation tests confirm model performance for total phosphorus concentrations and load for Beaver Creek, Heron Lake outlet, and West Fork Des Moines River at Jackson, and indeed show improved performance for loads at Jackson. Jack Creek, Okabena Creek (concentration and load), and Heron Lake outlet (load only) show a degradation in performance during the calibration period.

Total Nitrogen: Model performance for total nitrogen during the calibration period is excellent, with average relative errors on concentration receiving a ranking of "good" or "very good" at all nine sites and for load at 7 out of 9 sites. The validation tests confirm the calibration for load, with all six sites receiving "good" or "very good" rankings. Two sites (Jack Creek and Heron Lake outlet) show poorer performance for concentration during the validation period associated with model ability to simulate concentrations at lower flows.

Nitrate + Nitrite Nitrogen: Performance for nitrate + nitrite nitrogen is similar to that for total nitrogen, with 10 out 10 sites receiving "good" or "very good" rankings for concentration and nine out of 10 receiving such rankings for estimated load. The validation generally confirms the calibration, with the exception of Jack Creek and concentrations at Heron Lake outlet.

Beaver Creek: This is one of the few locations in the watershed not impacted by large lakes upstream. For total phosphorus, both the average and median relative error on concentration are higher than 25% during the calibration period, rendering the performance only "fair" at this location based on the criteria established in Table 3-5. The distribution of concentration residuals against flow suggests that most of the over-estimation occurs in the mid-range of flows suggesting that loading associated with tile drainage may be high. The average and median relative error on load are, however, low and the performance is rated as "good". This location is downstream of two minor point sources (Avoca and Wilson) that may impact concentrations during the low flow periods. The model performance for the validation period based on average and median relative error on concentration are "very good" and "fair", respectively.

Based on average and median relative error on load, the model performance is “fair” and “very good”, respectively.

The model performance for total nitrogen is “very good” during the calibration and validation periods based on average and relative error on concentration. The average relative error on load appears to have somewhat of a high bias due to the presence of a few outliers during high flow periods, but is still rated as “good”. The low median relative error on load suggests that the model performance for total nitrogen is “very good”. The model performance for nitrite + nitrate nitrogen is similar to that for total nitrogen. This is expected since nitrite + nitrate nitrogen accounts for a large proportion of the total nitrogen load except where affected by algal growth in eutrophic lakes.

Lake Shetek: This station is located at the outlet of the Lake Shetek complex and is impacted significantly by algal activity within the lake, where the reported chlorophyll *a* concentrations frequently exceed 100 µg/L. The calibration for nutrients at this location consisted of balancing the nutrient responses within the lakes and at the outlet stream station (see Section 8.1). Despite these challenges, the model performance for total phosphorus at this location was rated “good” based on average relative error on concentration but only “fair” based on median relative error on concentration. The model performance based on average and median relative errors on load is general rated as “good”.

For total nitrogen the model performance was “very good” based on relative errors on concentration and load. The model performance for nitrite + nitrate nitrogen is similar to the performance for total nitrogen and is generally rated as “good” to “very good”.

Jack Creek: The water quality response at this location is impacted by two relatively large lake upstream complexes: Fulda (First and Second) and Graham (East and West). The high chlorophyll *a* concentrations reported for these lakes suggest that algal activity may have significant impacts on nutrients in the stream. The model performance for total phosphorus was rated as “very good” for the calibration period based on relative errors on concentration and load. The performance for the validation period was, however, rated as “poor”. The average reported concentration for the validation period at 0.264 mg/L was substantially higher than that for the calibration period at 0.179 mg/L. The reasons for the difference are unknown but could reflect differences in algal activity in the upstream lakes during the validation period as the major lakes upstream of this gage report some of the highest chlorophyll *a* concentrations during the validation period. The under-estimation during the validation period could also be associated with uncertainties in the flow simulation as there are not available flow gage data to validate model hydrology for this.

The model performance for total nitrogen based on relative errors on concentration and load is rated as “very good” for the calibration period. The performance for the validation period is only “good” to “fair”. The performance for nitrite + nitrate nitrogen is similar to that for total nitrogen. The impacts of elevated algal activity in upstream lakes during the validation period and uncertainties associated with flow also apply to nitrogen simulation at this station.

Okabena Creek: The water quality monitoring station at this location is not affected by the presence of large lakes but is downstream from two major point source dischargers - Worthington WWTP and Worthington Industrial WWTP. Worthington Industrial WWTP has an especially large impact on the total phosphorus concentrations at this location for the validation period (before 2002) and the beginning years of the calibration period (2002 - 2014). Prior to August 2004, the average monthly total phosphorus concentration in the wastewater discharge was consistently above 20 mg/L and sometimes as high as 35 mg/L. Since loading from point sources are represented in the model as monthly loads, the model is unable to reproduce the variations in concentrations observed in grab samples due to the daily variations in point source discharge amounts in a given month. Given these challenges, the model performance for total phosphorus during the calibration period is still generally rated as “good” based on relative errors on concentration and load. The performance for the validation period based on average relative errors on concentration and load is rated “poor”, perhaps due to the imprecise representation of point source loads.

The model performance based on median relative errors on concentration and load is rated as “Fair” and “Good”, respectively. It should however be noted that the performance for dissolved ortho-phosphate at this location is generally poorer than total phosphorus because total phosphorus discharge from Worthington Industrial WWTP is largely in the form of dissolved ortho-phosphate (see Appendix D).

The total nitrogen and nitrite + nitrate nitrogen samples collected at this location suggests that discharge of nitrite + nitrate nitrogen from Worthington Industrial WWTP likely has significant impacts. While the facility did not have a permit requirement to report nitrite + nitrate nitrogen concentrations, a limited number of observations reported for this facility after November 2011 are consistently above 150 mg/L. As the samples are limited in number and not available before late 2011, an average nitrite + nitrate nitrogen concentration of 100 mg/L was assumed with the discharge for this facility based on the instream observations. The model performance for total nitrogen and nitrite + nitrate nitrogen for both the calibration and validation periods are rated as “very good” based on average errors on concentration and load.

Heron Lake Outlet: This monitoring site is at the outlet of the Heron Lake complex and is also downstream of the Jack Creek and Okabena Creek watersheds. While the discussions for the Jack and Okabena Creek locations apply to this location, the nutrient calibration at this location is further complicated by algal activity in the Heron Lake complex. The calibration at this location consisted of balancing the model response for nutrients in-stream while ensuring that the nutrient and chlorophyll *a* concentrations in the lakes were reasonable (see Section 8.1). The model performance for total phosphorus for the calibration period is rated only “Fair” based on average relative error on concentration. The under-prediction occurs for the mid-range flows and could likely be related to algal activity. While the model was calibrated to chlorophyll *a* in lakes, the sample size was often too small for a comprehensive evaluation. The performance based on median relative error on concentration is, however, rated as “very good”. The model performance based on relative errors on load is “very good” for the calibration period. For the validation period the model performance is “very good” and “good” based on average and median relative errors on concentrations, respectively. The model performance based on average relative error on load is rated “poor”, but is “good” based on median relative error on load.

The model performance for total nitrogen is “very good” for the calibration period based on relative errors on concentration and load. The performance for nitrite + nitrate nitrogen for the calibration period is similar. The model performance for the validation period for total nitrogen is rated as “poor” based on relative errors on concentration but “good” based on relative errors on load. The performance for nitrite + nitrate nitrogen based on average and median relative concentration errors are “poor” and “good”, respectively. The performance based on relative errors on load is, however, “very good”.

It is important to note that both the South and North Heron Lakes report some of the highest in-lake chlorophyll *a* concentrations during the validation period (exceeding 300 µg/L and 500 µg/L for South and North Heron Lakes, respectively). The inconsistencies with nutrient performance observed for the validation period could also be due to uncertainties in the flow simulation, for which validation data at this location are not available.

Model simulation at this station also shows inconsistencies in the representation of inorganic versus organic species of phosphorus (see Appendix D). The inorganic and organic fractions of phosphorus and nitrogen are highly influenced by algal activity in lakes. In particular, it is not possible to represent any vertical stratification of the distribution of nutrient species in lakes with a one-dimensional model. In addition, the relatively large point source input of nutrients from Worthington Industrial WWTP further complicates the nutrient balance in these lakes.

West Fork Des Moines River, Avoca: In addition to being downstream from the Lake Shetek complex, this location also includes the drainage from Beaver Creek. The model performance for total phosphorus at this location is “very good” for relative average errors on concentration and load. The model performances for total nitrogen and nitrite + nitrate nitrogen are rated as “very good” to “good”.

West Fork Des Moines River, Windom: This water quality station is located downstream of the Great Bend of the river in Cottonwood Co. The station is downstream of Heron Lake and is also affected by Talcot Lake and Lime Lake (on Lime Creek, upstream of Talcot Lake), which have reported chlorophyll *a* concentrations frequently exceeding 100 µg/L. Such high concentrations of algae in waterbodies are expected to have significant impacts on nutrient dynamics in the stream stations. The model performance for total phosphorus is rated as “good” to “fair” based on average relative errors on concentration and “very good” to “good” based on relative errors on load. The model performances for total nitrogen and nitrite + nitrate nitrogen are rated as “good” based on relative errors on concentration and “very good” based on relative errors on load.

West Fork Des Moines River, Jackson: This is the downstream station on the West Fork Des Moines River, and accumulates the conditions discussed for all the previous stations. The model performance at this location is generally rated as “very good” to “good” for total phosphorus, total nitrogen and nitrite + nitrate nitrogen concentrations for both the calibration and validation periods, with poorer performance for the total load estimates. The total phosphorus relative average error on load is rated as “poor” as a result of apparent high bias in simulated concentrations at the highest flows (see panel d in Figure 6-3 above, as well as the details in Appendix D). The nutrients at this location is further impacted by a major point source discharger - Windom WWTP. The low to mid-range flow concentrations of organic phosphorus and nitrogen are under-estimated, likely due to uncertainty in the simulation of algal activity in the Heron Lake complex and Talcot Lake.

Martin County Ditch: The last two stations (Martin County Ditch and Fourmile Creek) have much smaller drainage areas than the stations presented above. Smaller watersheds tend to present an issue in comparing point-in-time water quality measurements to daily average model output as concentrations often change rapidly over storm event hydrographs such that the observations are less likely to be representative of average concentrations. (Comparison of model results at the hour of observation, even where documented, is also sub-optimal as large apparent differences between model and observations result from even a small mis-timing of storm peaks.) The Martin County Ditch location is relatively free from the effects of upstream lakes, but there are two minor point source dischargers that may impact the nutrient concentration. The model performance for total phosphorus is rated as “poor” based on average relative error on concentration and is largely due to over-estimation during low flow periods, possibly associated with uncertainties in the representation of the point source discharges. The performance based on median relative error on concentration is however rated “good”. The performance based on average and median relative errors on load are rated as “poor” and “very good”, respectively.

The model performance for total nitrogen is rated as “very good” to “good” but the total number of samples is insufficient for a rigorous evaluation. The model performance for nitrite + nitrate nitrogen is rated as “very good” based on relative errors on concentration and load.

Fourmile Creek: This location drains a relative small area that is free from the impacts of point sources or lakes. Only nitrite + nitrate nitrogen data were available for this location. The model performance is rated as “very good” based on relative errors on concentration and load.

6.2.2 Comparison of Model to FLUX Estimates of Delivered Load

MPCA’s Watershed Pollutant Load Monitoring Network (WPLMN) is designed to obtain spatial and temporal pollutant load information from Minnesota’s rivers and streams and track water quality trends. As part of this program, MPCA releases estimates of annual pollutant loads for each 8-digit hydrologic unit code basin developed using the FLUX program, as described in Section 5.5. MPCA estimates of nutrient loads at the downstream gage station on the West Fork Des Moines River at Jackson are currently available for calendar years 2007 - 2011. Comparisons between the MPCA FLUX estimates and model simulated results are shown in Figure 6-4 through Figure 6-6, and Table 6-9.

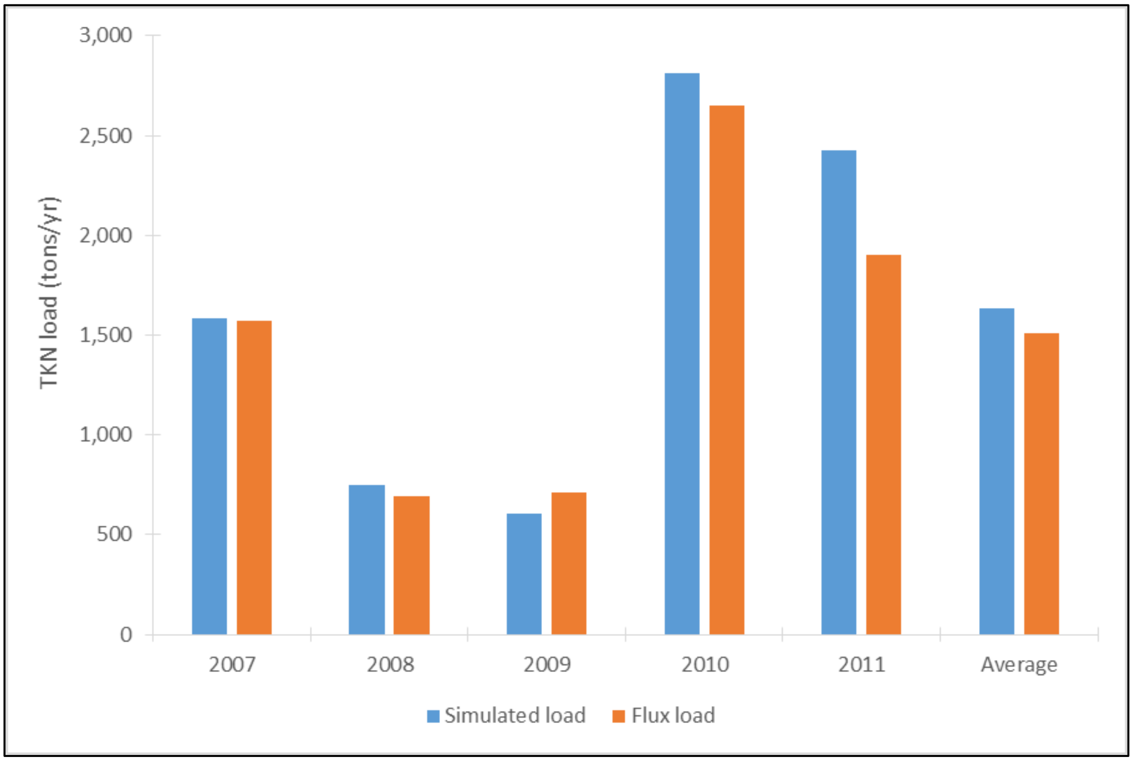


Figure 6-4. Comparison of Model to MPCA FLUX Estimates of Total Kjeldahl Nitrogen (TKN) Load, Calendar Years 2007-2011, West Fork Des Moines River at Jackson

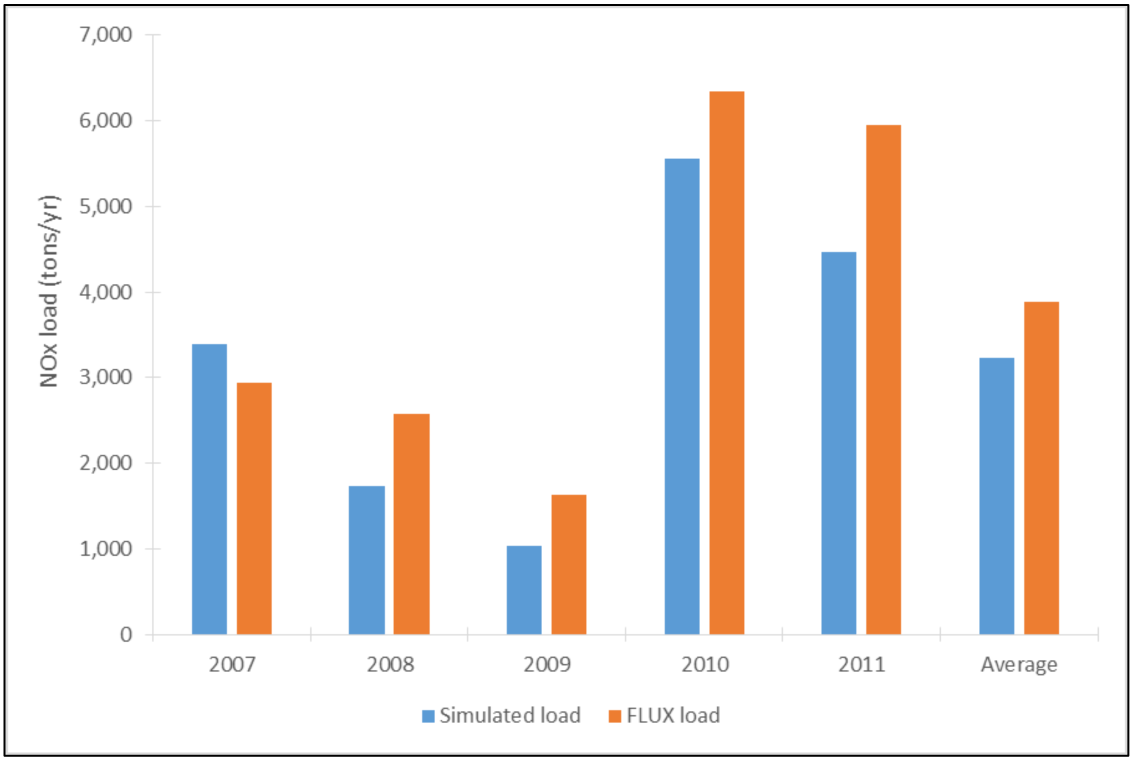


Figure 6-5. Comparison of Model to MPCA FLUX Estimates of Nitrite + Nitrate Nitrogen (NOx) Load, Calendar Years 2007-2011, West Fork Des Moines River at Jackson

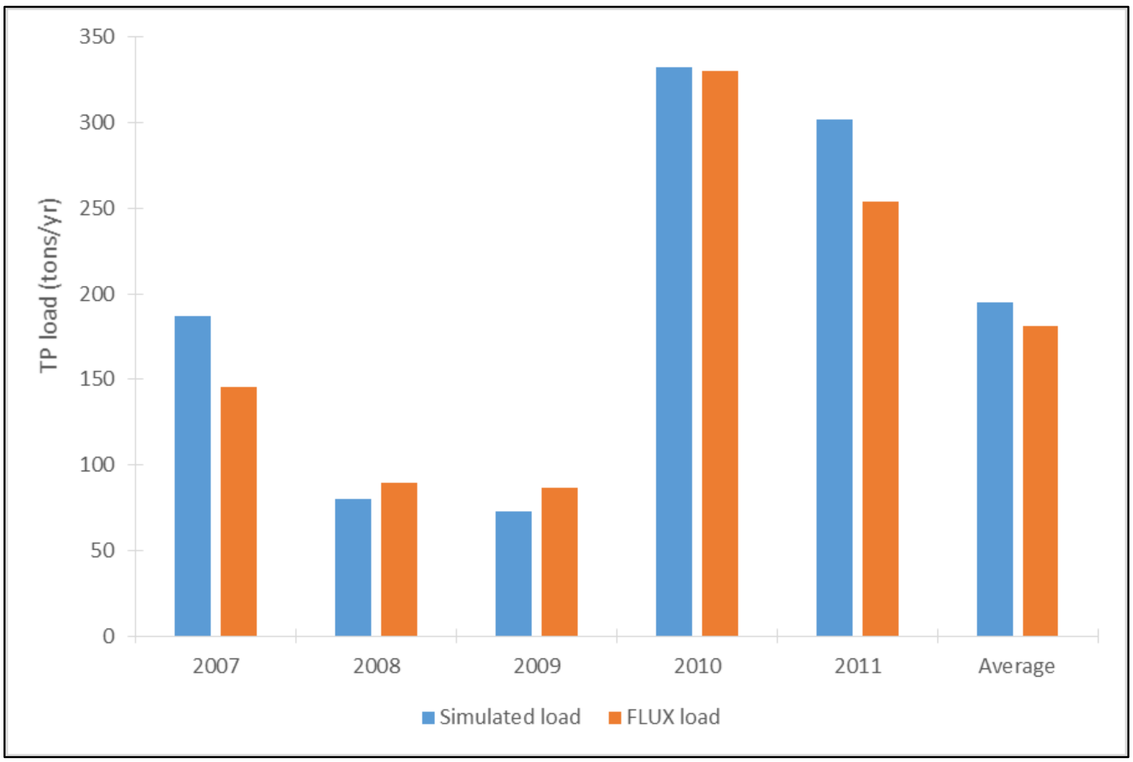


Figure 6-6. Comparison of Model to MPCA FLUX Estimates of Total Phosphorus (TP) Load, Calendar Years 2007-2011, West Fork Des Moines River at Jackson

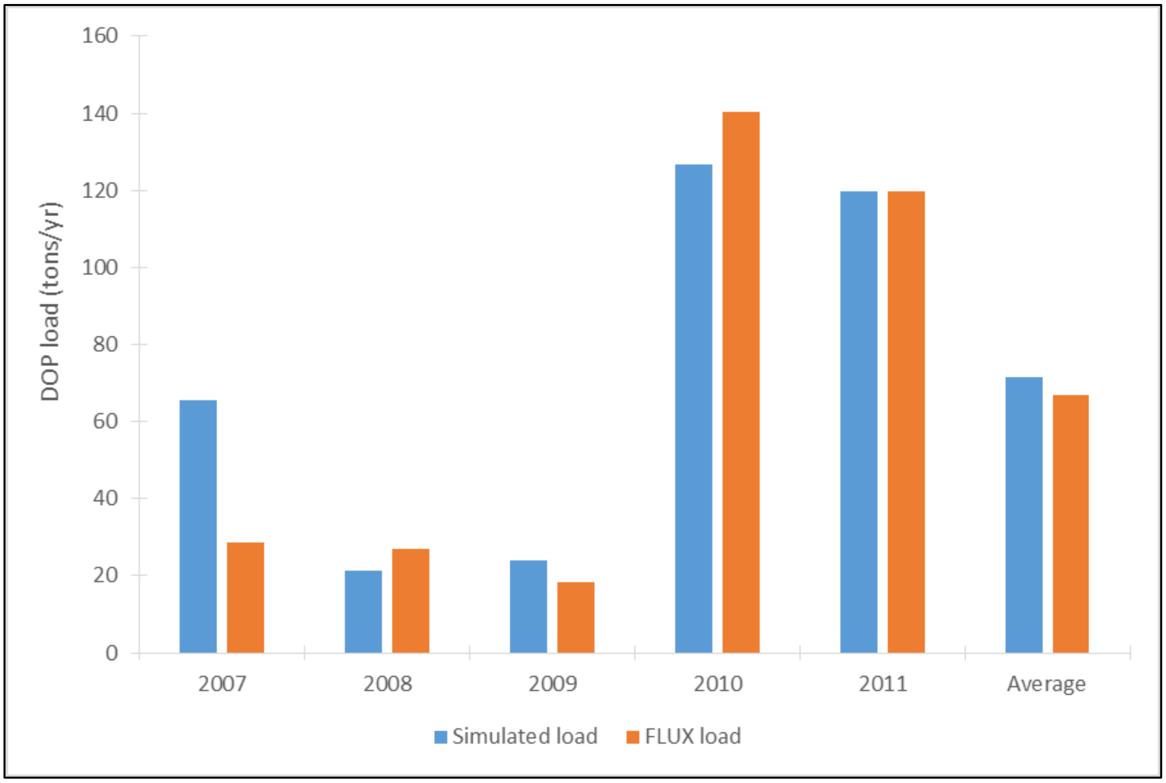


Figure 6-7. Comparison of Model to MPCA FLUX Estimates of Dissolved Ortho-Phosphate (DOP) Load, Calendar Years 2007-2011, West Fork Des Moines River at Jackson

Table 6-9. MPCA FLUX Estimates and Model Simulated Annual Nutrient Loads, Calendar Years 2007-2011, West Fork Des Moines River at Jackson

Year	TKN (tons/yr)		NOx (tons/yr)		TP (tons/yr)		DOP (tons/yr)	
	Simulated	FLUX	Simulated	FLUX	Simulated	FLUX	Simulated	FLUX
2007	1,583	1,573	3,395	2,937	187	145	65	29
2008	751	693	1,735	2,580	80	89	21	27
2009	603	712	1,027	1,631	73	86	24	18
2010	2,810	2,651	5,552	6,338	333	330	127	140
2011	2,429	1,903	4,461	5,957	302	254	120	120
Average	***1,635***	***1,506***	***3,234***	***3,889***	***195***	***181***	***71***	***67***
Difference	***8.6%***		***-16.8%***		***7.7%***		***7.0%***	

As evident from the figures and tables above, the modeled load deviates somewhat from the FLUX loads during some years. The FLUX results are themselves uncertain, and accuracy is limited by the representativeness of the same monitoring data used for model calibration. Based on average load over the 5 year period the match between FLUX and model simulated loads for TKN, TP and DOP are rated “very good”. The model seems to under-predict NOx loads but performance is still rated “good”. The comparison between the simulated and FLUX loads generally confirm the model calibration for nutrients.

It is important to note that both NOx and DOP are very sensitive to plant/algal uptake of inorganic nutrients and release of organic nutrients, much of which occurs in a large number of shallow and eutrophic lakes in the watershed. HSPF does not provide detailed simulation of kinetic processes in waterbodies.

6.2.3 Consistency with Lake Analyses

The Des Moines River watershed has a large number of lakes. Detailed nutrient balance studies are not available for these lakes. There are, however, in-lake monitoring stations for most of these lakes. Nutrient concentrations in ten explicitly simulated lakes in the model were compared to observed concentrations from 1995 - 2014.

HSPF has limited ability to simulate time series of nutrient in lakes because it is a one-dimensional model that predicts the nutrient content of the entire water volume, whereas the observations are mostly summer results from the surface water. Surface and total water volume nutrient concentrations may be quite different in lakes that undergo summer thermal stratification of the water column. A different analytical tool that incorporates a two-dimensional analysis may be needed to interpret nutrient concentrations in these and other stratified lakes (see discussion in Section 9).

The calibration for nutrients in lakes focused on ensuring that the average modeled and observed concentrations for the growing season were in general agreement. In several cases, the simulated nutrient concentrations in lakes needed to be higher than the observed concentrations, but this was required to ensure that the in-lake chlorophyll *a* concentrations were reasonable and nutrient concentrations at downstream stations matched well with the observed and may reflect the difference between surface and total lake volume nutrient content. Average and median nitrite + nitrogen, total Kjeldahl nitrogen and total phosphorus predictions in ten explicitly simulated lakes in the Des Moines River watershed model are compared to available observations for 1995 - 2014 in Table 6-10, Table 6-11 and Table 6-12, respectively.

Table 6-10. Nitrite + Nitrate Nitrogen Concentrations (mg/L) in Selected Explicitly Simulated Lakes of the Des Moines River Watershed

Name	Subbasin	# Samples	Monitored			Simulated		
			Average	Median	Range	Average	Median	Range
Sarah	131	22	0.2	0.2	0-0.4	0.3	0.2	0-2.8
Shetek	126	79	0.3	0.0	0-0.7	0.3	0.1	0-3.8
Lime	95	9	0.8	0.2	0.1-4.2	1	0.1	0-11.6
Talcot	90	9	0.5	0.1	0.1-4.3	0.7	0	0-8.7
Fulda	79	56	1.6	0.2	0-7.9	1.2	0.4	0-10.3
West Graham	77	36	0.6	0.3	0-1.8	0.4	0.1	0-7.1
East Graham	76	35	0.4	0.2	0-1.6	0.3	0.1	0-5.8
South Heron	37	43	0.6	0.0	0-4.4	0.7	0.1	0-8
North Heron	36	28	2.8	0.3	0-7.9	2.1	0.5	0-12
Okamanpeedan	171	12	0.4	0.1	0.1-2.4	0.7	0.2	0-10.1

Table 6-11. Total Kjeldahl Nitrogen Concentrations (mg/L) in Selected Explicitly Simulated Lakes of the Des Moines River Watershed

Name	Subbasin	# Samples	Monitored			Simulated		
			Average	Median	Range	Average	Median	Range
Sarah	131	28	2.0	1.9	1.1-3.9	2.1	2	0.4-4.6
Shetek	126	81	2.2	2.0	1-3.8	2.8	2.8	0.4-6.2
Lime	95	17	3.0	3.3	1.2-5	2.8	2.9	0.4-6.5
Talcot	90	17	3.0	3.1	1.5-5.1	2.7	3	0.2-6.1
Fulda	79	60	2.0	1.9	0.7-4	1.7	1.6	0.3-5.2
West Graham	77	42	1.9	1.7	1.1-5	1.6	1.4	0.1-4.2
East Graham	76	41	2.3	2.3	1.2-3.2	1.8	1.8	0.3-4.6
South Heron	37	44	5.2	4.7	2.1-12.3	4.3	4.2	1.2-7.7
North Heron	36	29	4.7	4.8	1.1-9.6	3.6	3.5	1.6-7.6
Okamanpeedan	171	25	3.0	2.7	1-4.8	3.6	3.6	1.2-5.7

Table 6-12. Total Phosphorus Concentrations (mg/L) in Selected Explicitly Simulated Lakes of the Des Moines River Watershed

Name	Subbasin	# Samples	Monitored			Simulated		
			Average	Median	Range	Average	Median	Range
Sarah	131	43	0.1	0.1	0-0.5	0.2	0.2	0.1-0.5
Shetek	126	106	0.1	0.1	0-0.2	0.4	0.3	0.2-0.9
Lime	95	18	0.2	0.2	0.1-0.3	0.4	0.3	0.2-1.4
Talcot	90	17	0.3	0.3	0.1-0.9	0.5	0.4	0.3-1.7
Fulda	79	141	0.1	0.1	0-0.5	0.2	0.2	0.1-0.6
West Graham	77	60	0.2	0.1	0-0.5	0.2	0.2	0.1-0.5
East Graham	76	60	0.2	0.2	0-0.7	0.2	0.2	0.1-0.6
South Heron	37	89	0.7	0.5	0.1-5.5	0.6	0.5	0.3-1.4
North Heron	36	41	0.5	0.4	0.1-1.8	0.6	0.4	0.1-5.1
Okamanpeedan	171	26	0.2	0.2	0.1-0.5	0.3	0.3	0.2-1.3

Figure 6-8 to Figure 6-10 show the simulated time-series of nitrite + nitrogen, total Kjeldahl nitrogen and total phosphorus, respectively. The model simulates the in-lake nitrite + nitrate and total phosphorus concentrations well. Although the average total Kjeldahl nitrogen concentration is simulated well by the model, it is apparent that the model generally under-estimates the higher concentrations. Some of the high concentrations in phosphorus concentrations observed in Heron Lake prior to 2005 may be due to sub-monthly variability in the large phosphorus loads discharged to the Heron Lake complex via Okabena Creek from Worthington Industrial WWTP. In addition, most of the lakes are wide and shallow, which can lead to intermittent fluxes of nutrients from the sediment to the water column associated with wave-induced scour and recreational boating propwash, neither of which are well-represented in HSPF due to its one-dimensional reach simulation.

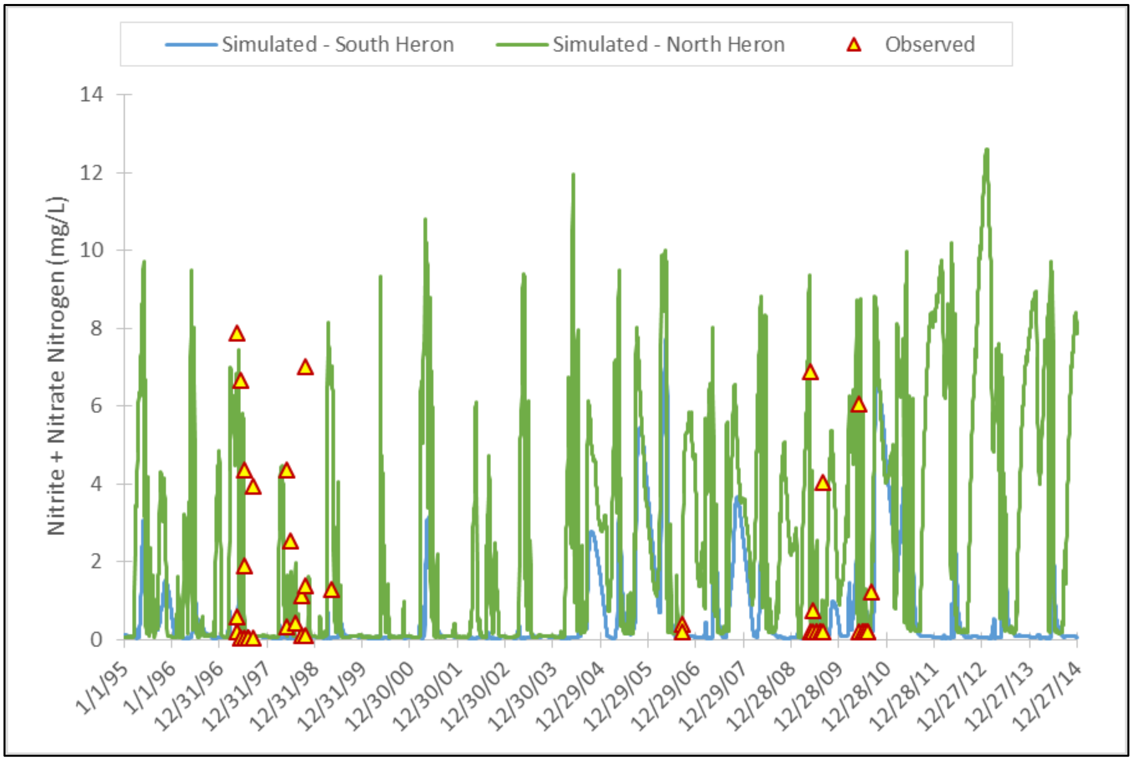


Figure 6-8. Time-series of Simulated Average Daily Nitrite + Nitrate Nitrogen Concentration Compared to Point-in-time Measurements for Heron Lakes

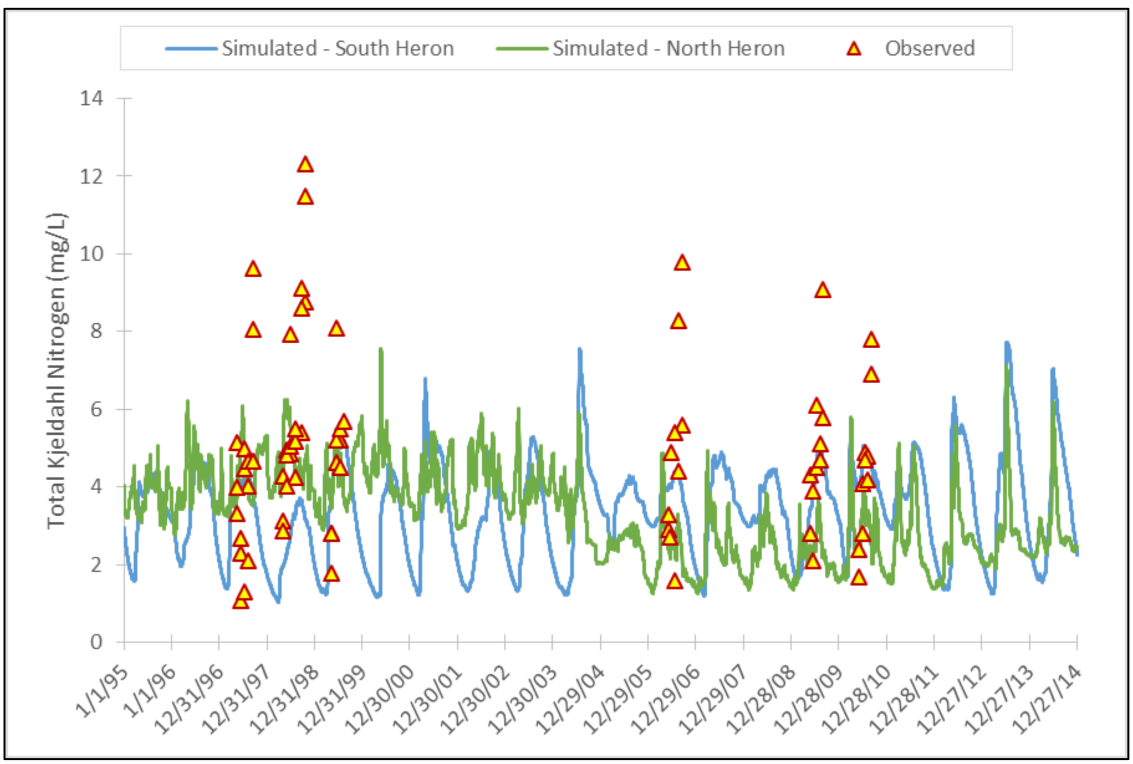


Figure 6-9. Time-series of Simulated Average Daily Total Kjeldahl Nitrogen Concentration Compared to Point-in-time Measurements for Heron Lakes

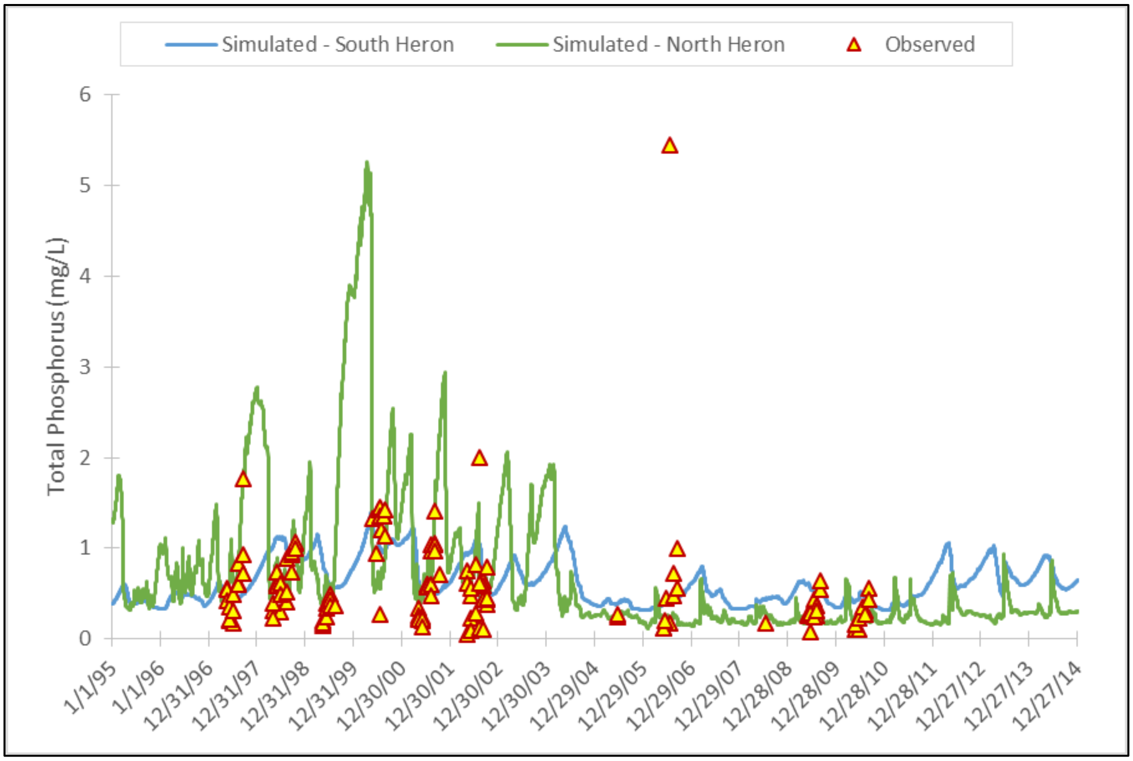


Figure 6-10. Time-series of Simulated Average Daily Total Phosphorus Concentration Compared to Point-in-time Measurements for Heron Lakes

7 Water Temperature Calibration

Water temperatures are of interest in their own right for habitat evaluation. Water temperature also has an important influence on the simulation biochemical transformations. The HSPF modules used to represent water temperature include PSTEMP (soil temperature) and HTRCH (heat exchange and water temperature).

Simulation of soil temperature is accomplished using three layers: the surface soil layer, upper subsurface layer, and groundwater subsurface layer. The surface layer is the portion of the land segment that determines the overland flow water temperature. The upper subsurface layer determines interflow temperature while the groundwater subsurface layer determines the temperature of discharging ground water. Surface and upper subsurface layer temperatures are estimated in HSPF by applying a regression equation relative to observed air temperature. The groundwater subsurface temperatures are supplied as slowly varying monthly time series that reflect average groundwater temperatures for the region and season. Initial parameters for the Des Moines River watershed model are based on recommendations in the Long Prairie example file provided as part of MPCA's HSPF modeling guidance (AQUA TERRA, 2012).

Once water enters a stream, temperature is impacted by processes that increase or decrease the heat content of the water. Mechanisms that can increase the heat content of the water are absorption of solar radiation, absorption of long-wave radiation, and conduction-convection exchange with the atmosphere. Mechanisms that decrease the heat content are emission of long-wave radiation, conduction-convection, and evaporation. Heat exchanges between the water and stream bed are also simulated.

Stream temperature follows diel cycles and is strongly affected by the pattern of shading over the course of the day and the local microclimate, as well as specific locations of cooler groundwater discharges to streams. Local-scale variations in hydraulics can also influence temperature readings: for instance, temperatures are likely to be different in a part of a reach impounded by a beaver dam than in a free-flowing riffle. A watershed-scale HSPF model can typically match observed daily *average* water temperature but is limited in its ability to simulate the daily cycles of water temperature at specific locations. This is because HSPF represents stream segments as one-dimensional, fully-mixed reactors. These segments are typically in the range of 3 to 15 miles in length in models built at a HUC12 scale, as is the case here, and variations within the segment are averaged out. For instance, a single average value represents shading over the whole stream segment and the model does not consider the orientation or aspect of the stream segment relative to the position of the sun. HSPF, as a one-dimensional model, also does not address vertical variation in temperature, which is especially important in deeper lakes and reservoirs. HSPF also turns off the simulation of instream heat exchange processes when water depth falls below 2 inches. In contrast, a detailed water temperature model for a stream reach (e.g., the QUAL2K model) would typically specify segments with lengths on the order of a tenth of a mile and include a detailed analysis of shading from vegetation and topography in relation to solar position throughout the day and year. For the HSPF application we used an empirical approximation fit during calibration in which the shading factor (i.e., CFSAEX, the fraction of light not shaded out) is scaled relative to the fraction of forest cover in a subwatershed as 1 – 0.73 * fraction forest.

While water temperature is reported along with most water quality observations, scattered point in time measurements are of limited use for adjusting the temperature calibration due to strong diel patterns. Modeled water temperature was compared to grab sampling at several MPCA flow gaging locations and was found generally to conform to the trends in the observed dataset. An example time-series plot for Des Moines River at Jackson is shown in Figure 7-1.

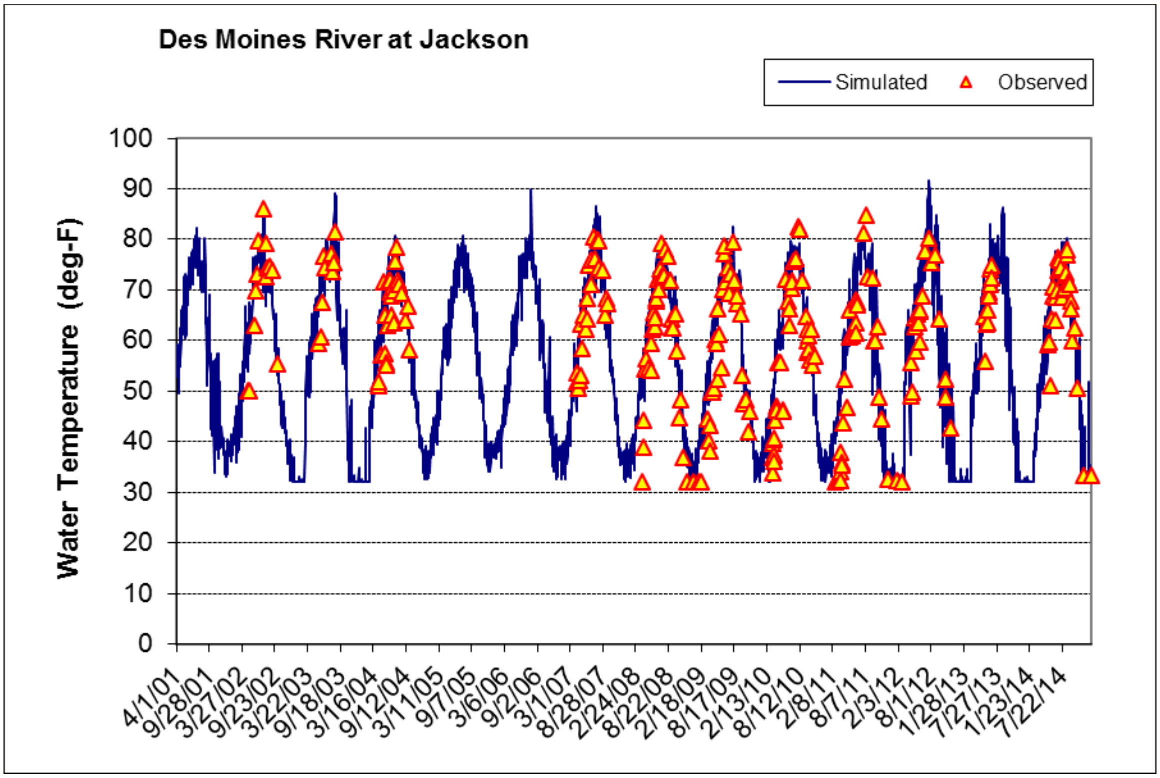


Figure 7-1. Time-series of Simulated Average Daily Water Temperature Compared to Point-in-time Measurements for Des Moines River at Jackson

8 Algae and Dissolved Oxygen Calibration

Dissolved oxygen (DO) concentration in streams results from a complex interaction of reaeration rate (a function of turbulence), the oxygen concentration of inflowing water, the saturation concentration of oxygen (which depends on temperature and salinity), consumption of oxygen by bacterial breakdown of carbonaceous and nitrogenous material in the water column (biochemical oxygen demand) and at the water-sediment interface (sediment oxygen demand), production of oxygen during photosynthesis by algae and macrophytes, and consumption of oxygen during nighttime algal/macrophytes respiration. The impact of plant photosynthesis/respiration and diel cycles of water temperature results in a situation where grab sample measures of DO are not very informative for model calibration. Further, the influence of algae/macrophytes on DO means that DO and algae must be calibrated simultaneously.

8.1 Algae

Limited data are available on algae and macrophytes in flowing streams of the Des Moines River watershed. Observations of chlorophyll *a*, the primary photosynthetic pigment in most algae, are available for many lakes and serve as an indicator of planktonic algae density - but do not provide information on benthic algae and macrophytes. However, many of the monitored lakes are of small size and not explicitly simulated in the basin-scale model. Given the relative paucity of information on algal density, model calibration focused on ensuring that planktonic chlorophyll *a* concentrations were in a reasonable range.

The Des Moines River HSPF model was calibrated to average chlorophyll *a* concentrations for several large lakes in the watershed. Although most lakes explicitly modeled have reported chlorophyll *a* data, the data are not continuous and span only a few years. As with nutrients, HSPF representation of algae is limited by the one-dimensional representation of reaches that cannot represent vertical differences in algal density during stratified conditions.

Average and median chlorophyll *a* predictions in 10 explicitly simulated lakes in the Des Moines River watershed model are compared to available observations for 1995 - 2014 in Table 8-1. Calibration for chlorophyll *a* was carried out simultaneously with nutrients at stream monitoring locations and nutrient concentrations in lakes. While the model generally represents the average concentrations reasonably, it under-predicts observed chlorophyll *a* concentration for Talcot Lake and Lime Lake. The model is generally lower than the average values reported for the South and North Heron Lakes, but the average concentrations for these lakes are inflated by a few very high observations. The observed median values are in better agreement with the simulated medians.

Table 8-1. Chlorophyll *a* Concentrations (µg/L) in Selected Explicitly Simulated Lakes of the Des Moines River Watershed

Name	Subbasin	# Samples	Monitored			Simulated		
			Average	Median	Range	Average	Median	Range
Sarah	131	21	40.5	16.2	0-118	38.5	35.7	0.1-195.7
Shetek	126	67	43.0	34.6	1.5-156	61	64.3	0.4-196
Lime	95	16	155.5	164.0	16.8-299	108.5	110.7	4.2-365.8
Talcot	90	17	155.1	154.0	8.9-357	97.2	96.6	0.3-238.7
Fulda	79	115	38.5	29.3	0-232.1	34.9	32	1.5-101.2
West Graham	77	48	50.4	37.2	1-224.4	46.2	46.1	0.3-98.1
East Graham	76	50	69.8	73.2	4.5-161.1	65.5	66.3	2.5-103.1
South Heron	37	70	162.2	144.9	29.1-427	116.6	114.5	6.3-387.5
North Heron	36	31	178.9	139.0	21.6-564	146.2	146.2	9.4-389.7
Okamanpeedan	171	26	149.2	117.0	4.7-332	95.5	101.9	1-193.6

Example plots of two major lakes in the watershed are shown below. Figure 8-1 shows observed and simulated chlorophyll *a* concentrations for Talcot Lake. The model represents the trend in observed concentrations well but is not able to predict some of the very high individual observations.

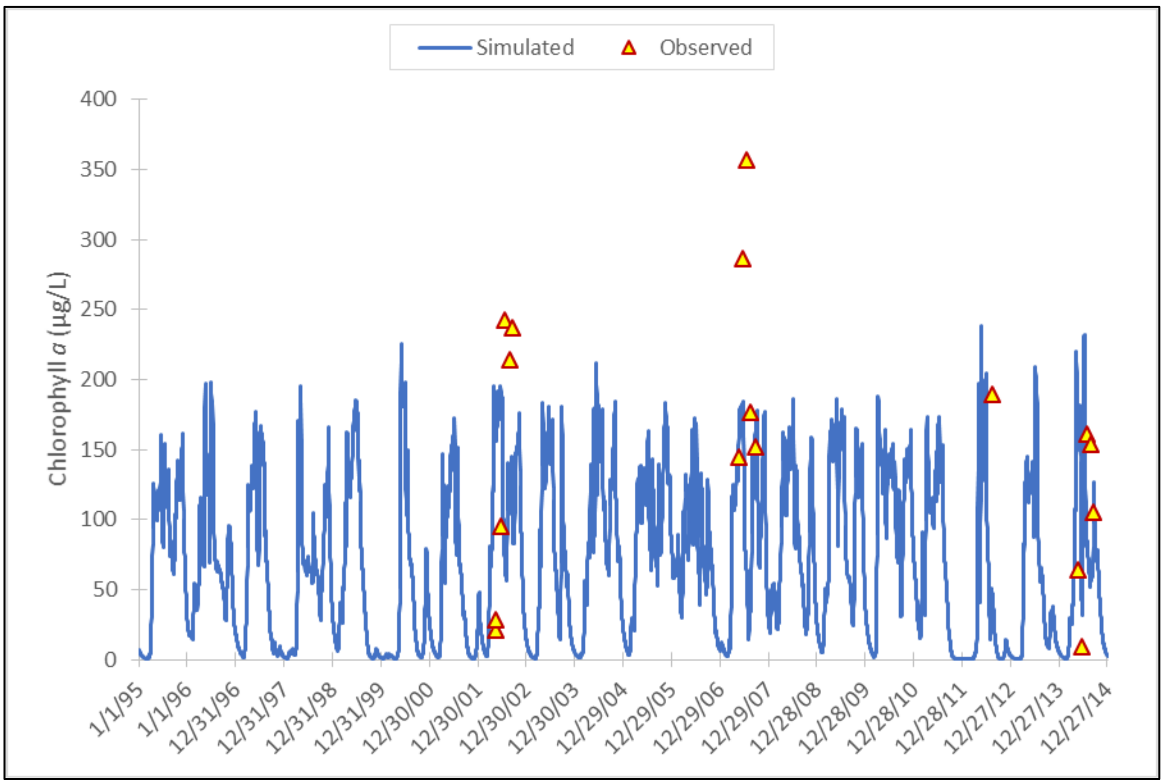


Figure 8-1. Time-series of Simulated Average Daily Chlorophyll *a* Compared to Point-in-time Measurements for Talcot Lake

Figure 8-2 shows observed and simulated chlorophyll *a* concentrations for the South and North Heron Lakes. Observed and simulated concentrations dropped after 2003 due to the reduction in phosphorus loads discharged to the Heron Lake complex via Okabena Creek from Worthington Industrial WWTP. The model generally predicts the trends well but is unable to reproduce some of the highest concentrations.

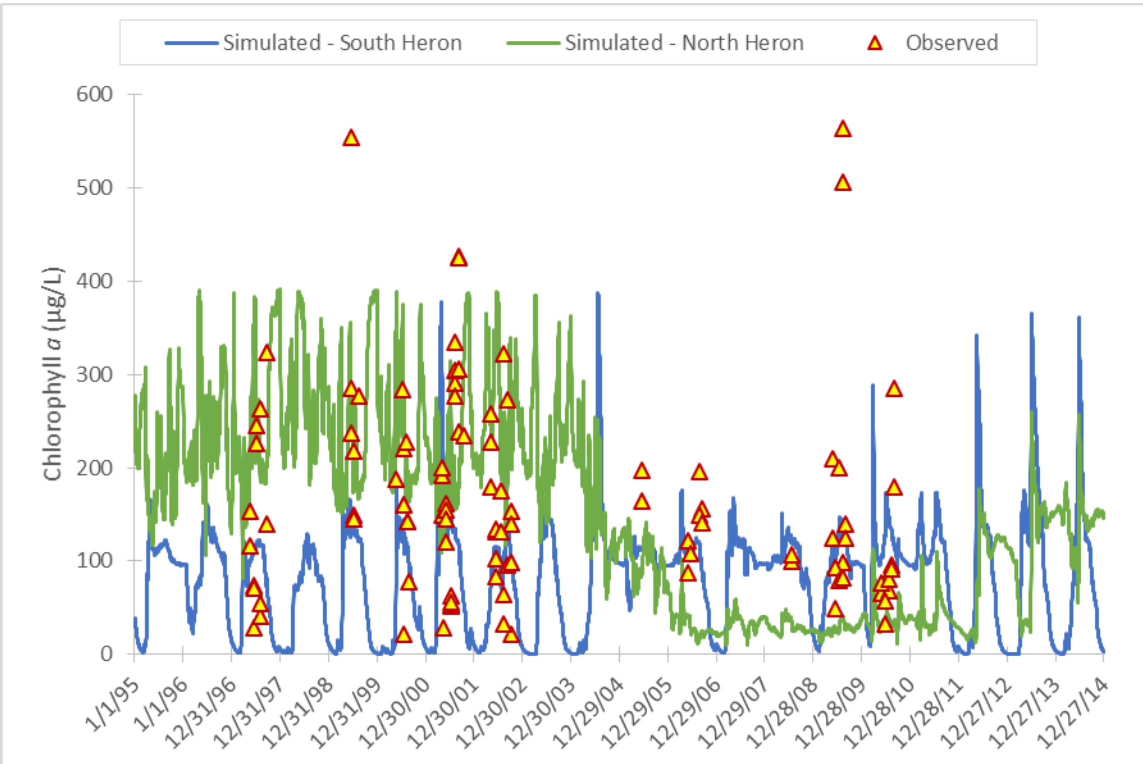


Figure 8-2. Time-series of Simulated Average Daily Chlorophyll *a* Compared to Point-in-time Measurements for Heron Lakes

8.2 DISSOLVED OXYGEN

Simulation of DO in waterbodies depends on a complex interaction between reaeration, algal production and respiration, and BOD (Figure 8-3). Many of these processes also affect nutrient balances, so the DO calibration must be achieved consistent with the nutrient calibration. The oxygen balance is also strongly dependent on water temperature simulation, which affects reaction rates and determines the saturation DO concentration.

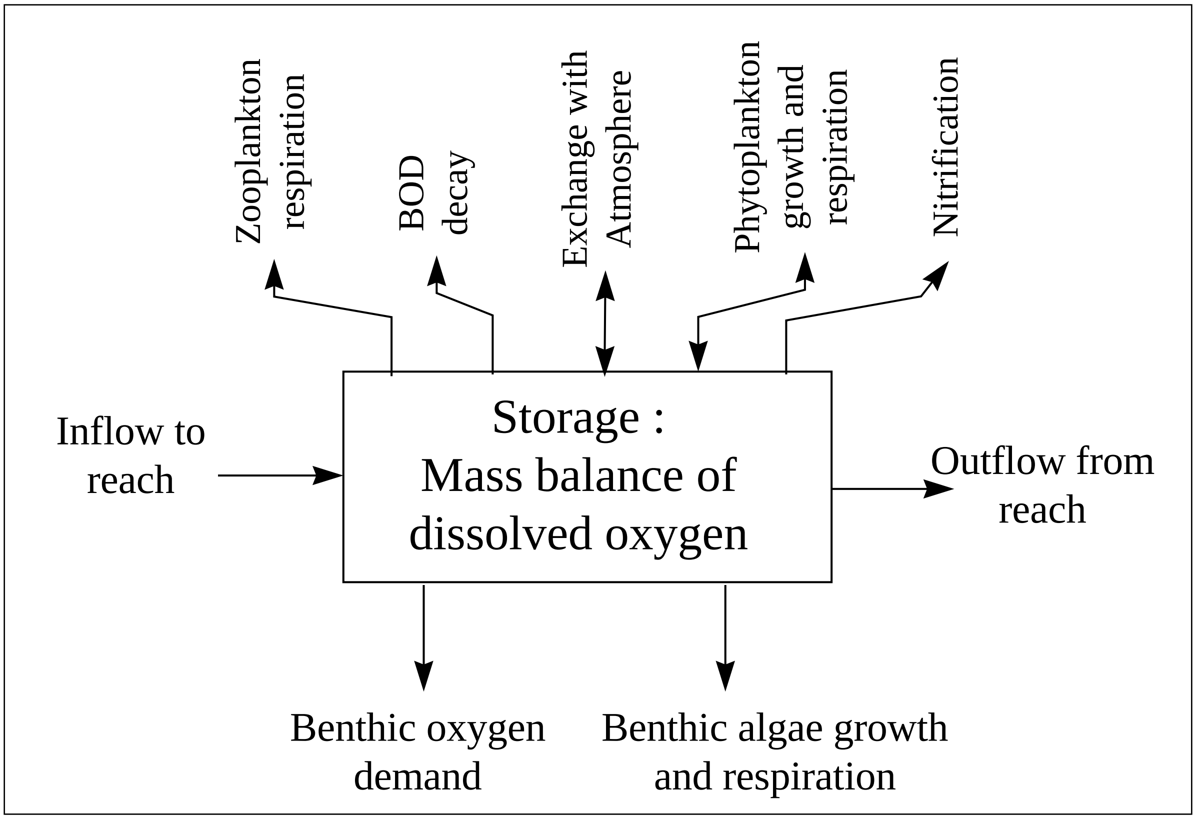


Figure 8-3. Process Diagram for Oxygen Mass Balance in HSPF

The impact of plant photosynthesis/respiration and diel cycles of water temperature on DO result in a situation where grab sample measures of DO are not very informative for model calibration. Many of the components of the oxygen mass balance in the Des Moines River watershed have little or no available monitoring data. Specifically, there are no known monitoring data for reaeration rates, benthic oxygen demand, or benthic algal or zooplankton densities. As noted in Section 8.1, monitoring for planktonic algae in streams is very limited. While biochemical oxygen demand (BOD) data exist for many locations, the majority of observations are for 5-day total BOD, whereas HSPF uses ultimate carbonaceous BOD. Total BOD includes the nitrogenous component and may also be affected by the presence of reduced iron. As a result, the model parameters must be specified based on best professional judgment and experience with other, similar sites. The model can then be tested on its ability to reproduce observed DO concentrations.

Reaeration: When oxygen concentrations are reduced below saturation, oxygen tends to move from the atmosphere to the water, a process known as reaeration. The rapidity of reaeration depends on how well the water is mixed and the turbulence present at the water surface. HSPF provides several options for simulating stream reaeration. For the watershed models the Tsivoglou energy dissipation method (Tsivoglou and Wallace, 1972) is used (with default parameters) for stream segments, while reaeration in lake segments is a function of wind speed and surface area (Bicknell et al., 2014).

Biochemical Oxygen Demand: HSPF simulates nitrogenous and carbonaceous components of biochemical oxygen demand separately, with the nitrogenous component being determined by concentrations of reduced inorganic nitrogen species (ammonium and nitrite). Carbonaceous biochemical oxygen demand (CBOD) loading from the watershed is simulated as the labile fraction of total organic carbon, as described in Section 6.1. As the decay of CBOD results in the conversion of labile organic matter to inorganic nutrients, the representation of CBOD is largely constrained by the nutrient calibration.

The CBOD decay rate (k_d) is expected to be relatively low due both to the nature of organic carbon derived from forest and wetland vegetation, except immediately downstream of point sources. A k_d value of 0.0035 per hour (0.084 per day) appears to provide reasonable results. This is near the low end of the range of values reported nationally for streams without untreated waste input (USEPA, 1997).

Benthic Interactions. Organic soils and sediment associated with northern wetlands affect the oxygen balance. These may both release BOD into the stream and exert a sediment oxygen demand (SOD) at the sediment-water interface. No direct measurements of SOD were identified, and these components are at this time a calibration adjustment factor. Note that in parts of the watershed the oxidation of reduced iron or sulfide could exert a significant oxygen demand. As HSPF does not explicitly address these components in the oxygen balance they are treated as part of the SOD.

Algal Dynamics: The activities of floating (planktonic) and attached (benthic) algae also affect the oxygen balance in streams. Algae produce oxygen as a byproduct of photosynthesis during sunlight hours, but are net consumers of oxygen through respiration at night. Algae can also die off, contributing to the biochemical oxygen demand.

Calibration for dissolved oxygen presents some of the same challenges as the temperature calibration as there is likely to be significant diel variability due to the influence of algal photosynthesis and respiration that limits the information value of scattered grab samples. There may also be significant spatial variability at scales smaller than the reaches in the basin-scale model due to local changes in light availability, substrate composition, and reaeration capacity.

Continuous time-series of DO observations coincident with the modeling time period were not identified for streams in the Des Moines River watershed. As a result, calibration checks for DO consisted of ensuring that simulated time-series followed the trends in the observed grab sampling data. An example time-series plot for Des Moines River at Jackson is shown in Figure 8-4. As with temperature, summer grab samples show DO concentrations higher than simulated daily averages. This reflects the influence of daytime photosynthesis, which can result in supersaturation of DO in the water column.

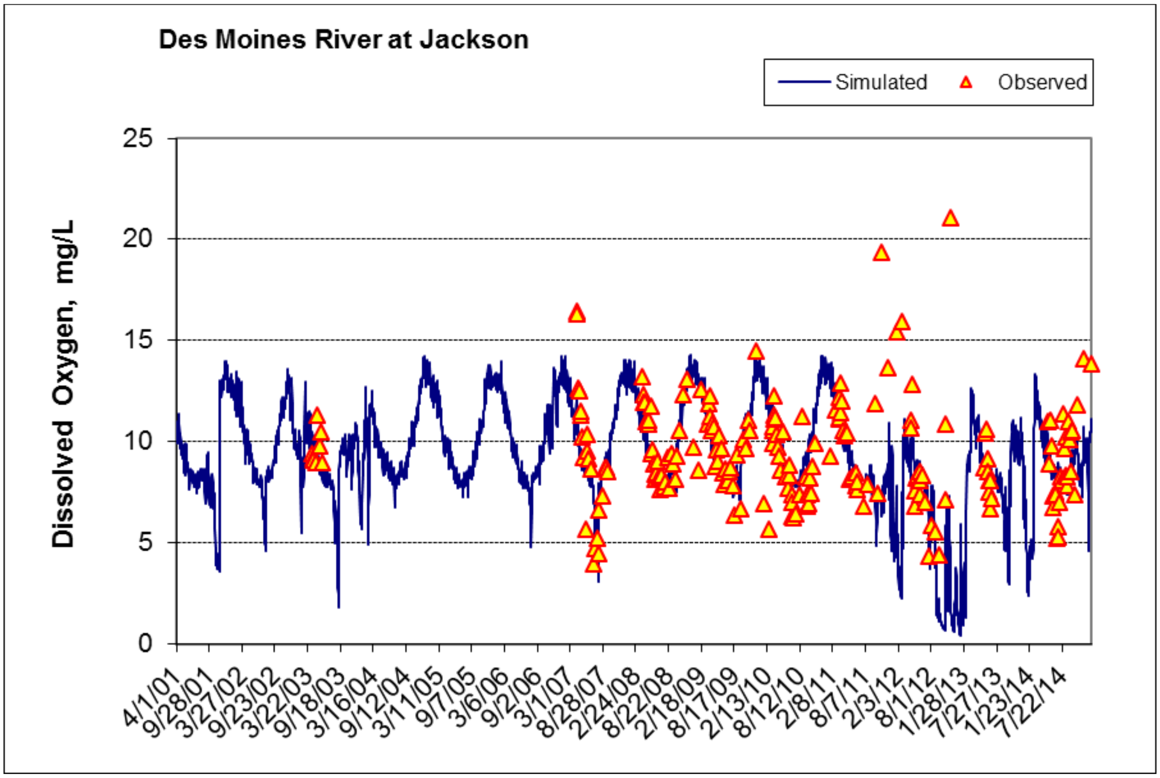


Figure 8-4. Time-series of Simulated versus Observed Average Daily Dissolved Oxygen Concentration for Des Moines River at Jackson

9 Potential Model Enhancements

The model calibration results presented in this report are based on simulations and comparisons to observed data through the end of 2014. As additional data are collected and other locations are monitored these new data are likely to prompt refinements and enhancements to the model. It is MPCA's intention to update HUC8-scale watershed models on an ongoing basis.

Both flow and water quality conditions in the Des Moines River watershed are affected by several large, eutrophic lakes with growing season chlorophyll *a* concentrations frequently exceeding 100 µg/L. Conditions in several lakes fail to achieve water quality standards and will need to be addressed. Detailed evaluation of model performance in selected lakes of high interest would likely result in enhanced model performance. There are, however, limitations to the ability of HSPF to simulate lake processes, as HSPF represents waterbody as one-dimensional, fully mixed reactors. This makes HSPF a useful tool for representation of nutrient balances in fully mixed lakes; however, the effects of stratification of the water column cannot be directly represented in the model. A promising approach for evaluating eutrophic lakes in the Des Moines River watershed would be to link the HSPF watershed model to lake models that are better able to represent these processes. A wide variety of lake models exist, at varying levels of complexity. For some stratified lakes of high importance, use of a complex two-dimensional lake model with a short time step (such as the USACE-supported CE-QUAL-W2) model may be appropriate. Use of a complex modeling approach is likely to be infeasible for all of the lakes present in the watershed – yet some representation of seasonal mixing in each stratifying lake is desirable to represent seasonal mixing processes. One more parsimonious and less expensive alternative would be to use a simple model such as BATHTUB (Walker, 1996) to estimate the seasonal average distribution of nutrients and algae in surface and bottom waters, driven by the cumulative loads estimated by HSPF. In addition to providing a tool to evaluate conditions within individual lakes, the BATHTUB results could be used to constrain the reasonable range of concentrations of nutrients and algae in the outflow from each lake in the basin-scale HSPF model.

Sediment routines in HSPF are designed to predict the inorganic sediment load, whereas total suspended solids observations include both organic and inorganic solids. The sediment calibration for the watershed model demonstrated that it was important to consider the sum of both inorganic sediment and algal biomass for comparison to observed total suspended solids as the algal biomass may equal or exceed inorganic sediment concentrations downstream of eutrophic lakes during summer low flow conditions. Future refinements to sediment calibration would benefit from sampling that distinguishes inorganic and volatile (organic) suspended sediment. Development of implementation strategies to address excess solids or turbidity in these streams will also need to consider the important role played by organic solids in the total sediment/solids balance.

An important feature of the hydrology and water quality of the Des Moines River watershed is interaction of streams with the alluvial aquifer. The HSPF model has been set up to use a set of parallel groundwater reaches to approximate the interactions between the surface streams and the alluvial aquifer. While this approach yields satisfactory results, model performance would benefit from linkage to a detailed groundwater flow model. A MODFLOW model exists for a small region of the West Fork Des Moines River in the Cottonwood County (Cowdery, 2005) but extending the model to the entire Des Moines River watershed would likely provide more insight into the exchanges between surface reaches and the aquifer and help improve the watershed model performance further. Creating such a model would, however, require the collection of a substantial body of water level data from monitoring wells.

Another area of for future enhancement of the model is the use of stream cross-section information for the development of stage-storage-discharge relationships for modeled reaches in the watershed. At this time, FTables in the model are based on regional regression equations due to a lack of availability of cross-section data. Incorporating such data into site-specific FTables would be expected to improve representation of the details of storm event hydrographs, but would have only minor impacts on the overall flow balance. The storm hydrographs play an important role in determining channel scour and deposition process and improving these would also likely improve the model performance for suspended sediment simulation.

Summer water temperatures and dissolved oxygen conditions are an important concern for aquatic habitat. The current model development includes limited calibration to water temperature, primarily to assure that temperatures are in the correct range; this calibration could be extended and improved. In addition, there were no available continuous DO measurement time series available. The basin scale watershed models aggregate stream reaches into segments that are several miles in length, and variations within the segment are averaged out. In contrast, continuous temperature and DO monitoring addresses conditions at a single, discrete location that is affected by local riparian cover, topographic shading, and the orientation or aspect of the stream segment relative to the position of the sun, all of which have strong impacts on energy inputs and exchanges over the course of a day, so the HSPF model is best suited to produce daily averages over a whole stream length, not hourly patterns at a specific cross section. A detailed examination of temperature and dissolved oxygen in reaches of interest would best be served through the development of finer-scale models for reaches of interest, using a tool such as the QUAL2K model. The basin-scale HSPF model can be used to provide boundary conditions for a detailed model of this type.

Finally, while most permitted point source discharges and major land use types are represented in the model, the current configuration uses monthly discharge records. Where available, daily discharge records should be incorporated. The representation of some specific source types can also be enhanced. For instance, a more detailed representation of loading from roads, feedlots, and un-sewered communities should be considered for explicit representation in the watershed model.

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