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Assessment of Nutrients and Suspended Sediment Conditions in and near the Agassiz National Wildlife Refuge, Northwest Minnesota, 2008–2010






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U.S. Department of the Interior
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Cover:

Top left, A1 (Branch 1 of Judicial Ditch 11 above Mud River Pool) (photograph by Rochelle A. Nustad, U.S. Geological Survey).

Top right, A3 (Judicial Ditch 11 above Agassiz Pool) (photograph by Rochelle A. Nustad, U.S. Geological Survey).

Middle left, A4 (Branch 200 of Judicial Ditch 11 above Farmes Pool) (photograph by Rochelle A. Nustad, U.S. Geological Survey).

Middle right, SG140 (Thief River inlet to the Agassiz National Wildlife Refuge) (photograph by Rochelle A. Nustad, U.S. Geological Survey).

Bottom left, A2 (Judicial Ditch 11 below Agassiz Pool) (photograph by Rochelle A. Nustad, U.S. Geological Survey).

Bottom right, A5 (Northwest outlet of Agassiz Pool) (photograph by Rochelle A. Nustad, U.S. Geological Survey).

Assessment of Nutrients and Suspended Sediment Conditions in and near the Agassiz National Wildlife Refuge, Northwest Minnesota, 2008–2010

Rochelle A. Nustad and Joel M. Galloway

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Conversion Factors

Inch/Pound to SI

Multiply	By	To obtain
inch (in.)	25.4	millimeter (mm)
foot (ft)	0.3048	meter (m)
mile (mi)	1.609	kilometer (km)
acre (ac)	0.0041	square kilometer (km^2)
square mile (mi^2)	259.0	hectare (ha)
pint (pt)	0.4732	liter (L)
quart (qt)	0.9464	liter (L)
gallon (gal)	3.785	liter (L)
cubic foot per second (ft^3/s)	0.02832	cubic meter per second (m^3/s)
pound per day (lb/d)	453.6	gram per day
ton per day (ton/d)	0.9072	metric ton per day

Temperature in degrees Celsius (°C) may be converted to degrees Fahrenheit (°F) as follows:

$$°F=(1.8\times°C)+32$$

Vertical coordinate information is referenced to North American Vertical Datum of 1988 (NAVD 88).

Horizontal coordinate information is referenced to North American Datum of 1983 (NAD 83).

Assessment of Nutrients and Suspended Sediment Conditions in and near the Agassiz National Wildlife Refuge, Northwest Minnesota, 2008–2010

By Rochelle A. Nustad and Joel M. Galloway

Abstract

In response to concerns about water-quality impairments that may affect habitat degradation in Agassiz National Wildlife Refuge in northwest Minnesota, the U.S. Geological Survey, in cooperation with the U.S. Fish and Wildlife Service collected streamflow data, discrete nutrient and suspended-sediment samples, and continuous water-quality data from 2008 to 2010. Constituent loads were estimated for nutrients and suspended sediment using sample data and streamflow data. In addition, a potential water-quality and streamflow monitoring program design was developed for Agassiz National Wildlife Refuge. Results from this study can be used by resource managers to address identified impairments and protect wildlife habitat and public water supply, and may contribute toward developing more effective water-management plans for Agassiz National Wildlife Refuge.

Streamflow was measured by the U.S. Geological Survey at four inflow and two outflow sites located on rivers and drainage ditches in and near Agassiz National Wildlife Refuge during open-water (no ice cover) periods in 2008, 2009, and 2010. Discrete samples were collected and analyzed for nutrients (total ammonia plus organic nitrogen, dissolved nitrate plus nitrite, dissolved ammonia, total nitrogen, dissolved orthophosphorus, and total phosphorus) and suspended-sediment concentration. Continuous water-quality measurements were collected for water temperature, specific conductance, dissolved-oxygen concentration, pH, and turbidity.

In 2006, the Thief River from Thief Lake to Agassiz Pool was listed as impaired for high ammonia concentrations. Results from this study indicate that concentrations at all sites did not exceed the 0.04-mg/L water-quality standard for un-ionized ammonia. Compared with the four inflow sites, the two outflow sites generally had significantly greater dissolved ammonia concentrations, significantly smaller nitrate plus nitrite concentrations, and no major differences in total ammonia plus organic nitrogen and total nitrogen. Small differences in suspended-sediment concentration were observed among inflow sites, but outflow sites had significantly greater suspended-sediment concentrations than inflow sites. At the primary outflow site, during the scheduled drawdown of Agassiz Pool from October 2009 into 2010, suspended-sediment concentrations were high compared to concentrations prior to the scheduled drawdown of Agassiz Pool. Overall, orthophosphorus and total phosphorus concentrations were significantly greater at inflow site A1 (located on Branch 1 of Ditch 11) than any other site. In 2010, although this site accounted for only 3 percent of the total streamflow from inflow sites, this same site accounted for 31, 27, and 13 percent of the inflow load for nitrate plus nitrite, orthophosphorus, and total phosphorus, respectively.

Among the sites, for most constituents, annual (open-water period) nutrient and sediment loads generally were greatest at the site with greatest volume of streamflow (the primary outflow site) and greatest in the year with the greatest amount of streamflow (2010). Large loads at the primary outflow site in 2010, particularly for sediment, likely resulted from the combination of greater flows in 2010 and scheduled drawdown of Agassiz Pool. Of the three inflow sites to Agassiz Pool, two of the sites accounted for at least 97 percent of the total annual sediment load from 2008 to 2010. Although loads were greater in 2010, in many cases the annual flow-weighted concentrations for nutrients and suspended-sediment were greatest in 2009, which may have been related to differences in the streamflow patterns between 2009 and 2010. For most sites and constituents, mean monthly nutrient and sediment loads were greatest in April, May, June, September, and October, which corresponded with months of greater streamflow volume. For the primary outflow site, the greatest sediment load occurred in October, which is likely related to high concentrations of suspended sediment at the start of scheduled drawdown of Agassiz Pool in October 2009 and large streamflow volume in October of 2010. For sites located downstream from Thief Lake and Agassiz Pool, the seasonal pattern of most mean monthly nutrient loads and mean monthly flow-weighted nutrient concentrations were affected by releases from these water bodies and the vegetative growing season. For inflow sites not located directly downstream from impoundments, much less variability in the flow-weighted concentrations of nitrate plus nitrite and orthophosphorus was observed. Continuous water-quality monitor data from 2010 indicated instances when water-quality standards for dissolved oxygen, pH, and turbidity were not met. For all sites, spikes in turbidity occurred related to rainfall, with as little as 2 percent

of the values exceeding the 25 nephelometric turbidity units water-quality standard and at most 38 percent of the values exceeding the standard.

A recent (2011) radioisotope study indicates that Agassiz Pool has been experiencing a net gain of sediment (more inflow load than outflow load) in the last 68 years, but during the 3-year period of this study (2008 to 2010), a net loss of sediment from Agassiz Pool occurred. A net loss from 2008 to 2010 was likely related to a combination of several atypical water-management activities that occurred at the two outflow sites including: the first year of operation of the water control structure at the smaller outflow site in 2008; construction downstream from the primary outflow site in 2008 and 2009; and scheduled drawdown of Agassiz Pool in fall 2009 through 2010, which occurs only once every 10 years.

A future water-quality monitoring program for Agassiz National Wildlife Refuge could include data collection at 2 indicator sites (one inflow and one outflow site) with a total of 7 discrete samples and 7 streamflow measurements consisting of the following: 5 samples, along with a streamflow measurement, collected during the same week each month in April, May, June, July, and October combined with 2 supplementary samples and streamflow measurements during periods of storm runoff. In addition to the discrete samples, continuous water-quality monitors could be deployed at each site. Future water-quality monitoring in Agassiz National Wildlife Refuge would provide information that can be used to assess the changes in water quality with time, changes in management conditions, effects of upstream mitigation practices (for example, buffer strips, side-channel inlets) within the Thief River watershed, as well as other variables.

Introduction

Agassiz National Wildlife Refuge (Agassiz NWR) is a 61,500-acre complex of wetlands and uplands located in the Thief River Watershed, in northwest Minnesota (fig. 1). Agassiz NWR was established in 1937 as Mud Lake Refuge, later renamed in 1961, and has been managed for the primary purpose of supporting breeding and migratory waterfowl (U.S. Fish and Wildlife Service, 2005). In 2006, the Minnesota Pollution Control Agency identified two reaches on the Thief River and one reach on the Mud River as impaired (Minnesota Pollution Control Agency, 2010). Both rivers enter and exit Agassiz NWR (fig.1). The water-quality impairments included high turbidity, low dissolved oxygen (DO), and high ammonia (Red Lake Watershed District, 2007). In 2008, the U.S. Geological Survey (USGS), in cooperation with the U.S. Fish and Wildlife Service (USFWS) began collecting information on streamflow, water quality, and suspended sediment in rivers and drainage ditches entering and exiting Agassiz NWR to address concerns about water-quality impairments that may affect habitat degradation.

Description of Study Area

Historically, the area that is now (2012) Agassiz NWR consisted of a boggy wilderness, checkered with wetlands. In 1909, a large public drainage project was undertaken to make the area more conducive to farming. After a million dollars was spent without accomplishing the intended objective, the Minnesota legislature, in 1937, authorized the purchase of 61,500 acres of land that is now Agassiz NWR by the Federal Government (U.S. Fish and Wildlife Service, 2005). Approximately 390,000 acres of drainage basin are upstream from Agassiz NWR. Of those 390,000 acres, the Thief River north of Agassiz NWR drains 224,000 acres of land and the Mud River to the east of Agassiz NWR drains approximately 102,000 acres of land (U.S. Fish and Wildlife Service, 2005). The Thief River Watershed is extensively drained and managed, with more than 1,200 miles of county, state, and judicial ditches and many tile drainage systems. Most of the water upstream from Agassiz NWR is controlled by two main water bodies (fig. 1). Thief Lake (7,100 acres), approximately 4 miles to the north of Agassiz NWR, is managed by the Minnesota Department of Natural Resources (MNDNR) for wildlife habitat, and Moose River Impoundment, a 12,000 acre-foot impoundment about 15 miles east of Agassiz NWR, is managed cooperatively by the Red Lake Watershed District (RLWD) and MNDNR primarily for flood control and secondarily for wildlife habitat. The North Pool of the Moose River Impoundment flows into the Moose River (also referred to as Judicial Ditch 21) and eventually flows into the Thief River. The South Pool flows into the Mud River (also referred to as Judicial Ditch 11). As the Thief River and the Mud River enter and exit Agassiz NWR, they are channelized. Downstream from Agassiz NWR, the Thief River flows to the Red Lake River, which is a drinking water source for the cities of Thief River and East Grand Forks, Minnesota and Grand Forks, North Dakota.

Land use in the Thief River Watershed consists of wetlands (33 percent), row crops (33 percent), forest (21 percent), pasture (8 percent), and residential or commercial development (3 percent) (Minnesota Pollution Control Agency, 2012). In addition to Agassiz NWR, numerous wetlands exist within Wildlife Management Areas that are managed by the MNDNR. Within Agassiz NWR there are 26 impoundments (also referred to as pools or wetlands) and three natural lakes (U.S. Fish and Wildlife Service, 2005). Wetlands and open water comprise approximately 61 percent of Agassiz NWR's 61,500 acres. Row crops in the Thief River Watershed include wheat, soybeans, barley, alfalfa, corn, sunflowers, and canola. A few livestock operations are located along the Mud River upstream from highway 89, but in recent years livestock numbers have been low in the Thief River Watershed (RLWD, 2010).

Soils in the Thief River Watershed consist of fine loams to coarse loams, with areas of sand soils in the northern reaches in the basin (Minnesota Pollution Control Agency,

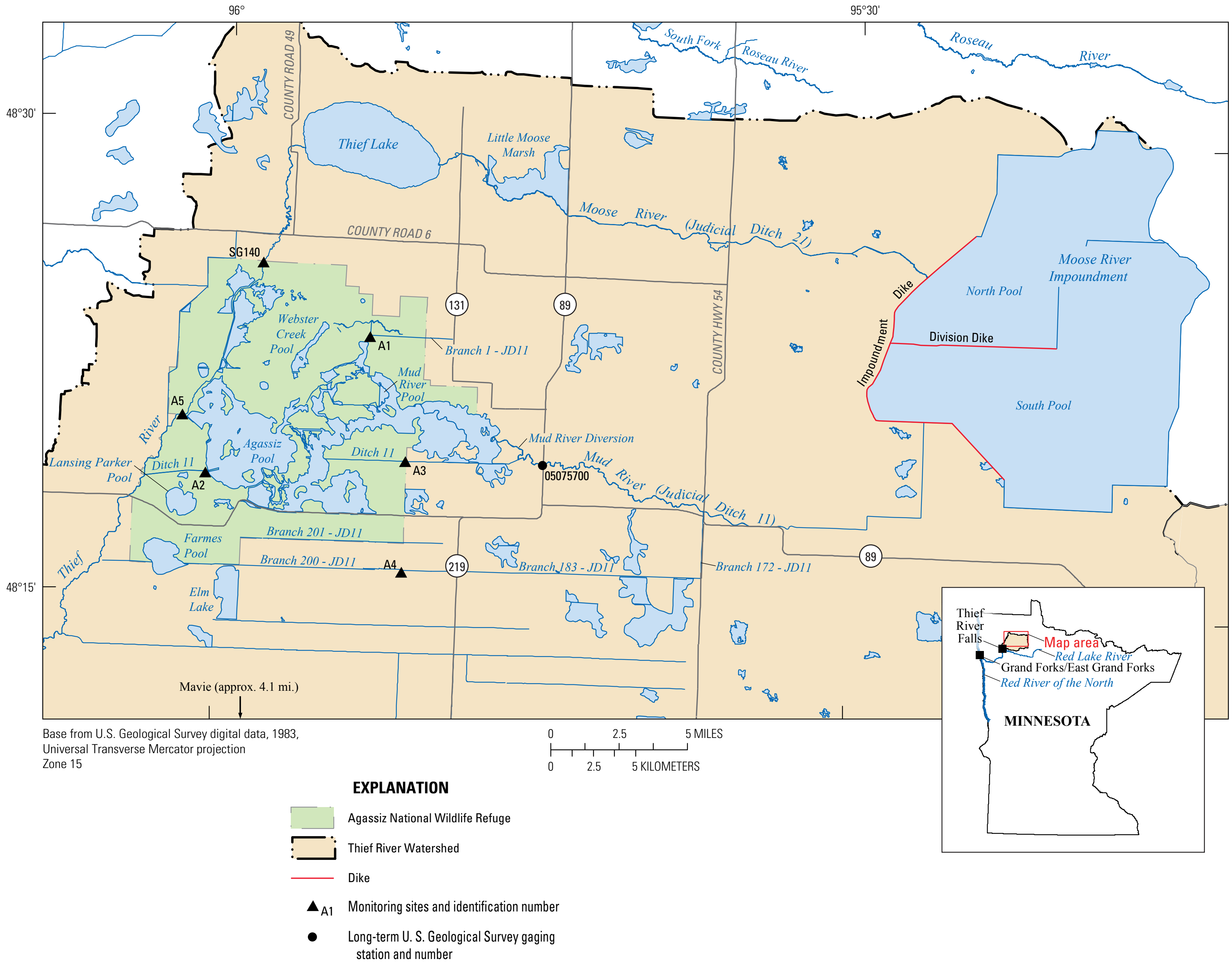


Figure 1. Location of monitoring sites in and near Agassiz National Wildlife Refuge, northwest Minnesota.

2012). Within Agassiz NWR, soils consist of peat or silty loams and clays (U.S. Fish and Wildlife Service, 2005). Beneath these surface soils, clay-dominated glacial drifts with pockets and lenses of sand are found. The glacial lake sediments and drift deposits of sand and gravel contain groundwater in quantities sufficient for domestic and stock use. The water table for most of Agassiz NWR is only 1 to 4 feet deep. This proximity to the surface has been favorable for pothole development, but conversely, makes building construction difficult and subsurface waste disposal impractical. The relative impermeability of the surface soils in Agassiz NWR impedes recharge of even its more permeable aquifers (U.S. Fish and Wildlife Service, 2005).

Although there are several smaller ditches entering and exiting Agassiz NWR, the six monitoring sites established for this study represent the majority of streamflow into and out of Agassiz NWR (fig. 1). Inflow monitoring sites to Agassiz NWR included A1, A3, SG140, and A4 (figs. 1–2). A1, located on Branch 1 of Judicial Ditch 11, receives all of its flow from field runoff and flows into Mud River Pool, which ultimately drains to Agassiz Pool. A3, located on the Mud River (Judicial Ditch 11) approximately 20 river miles downstream from the South Pool of the Moose River Impoundment, flows into Agassiz Pool. During high streamflow periods, the inflow to Agassiz Pool from Mud River is split between A3 and a diversion channel (hereafter, referred to as Mud River diversion). SG140, located on the Thief River as it enters Agassiz NWR from the north, also drains to Agassiz Pool. The three inflow sites, A1, A3, and SG140 comprise most of the inflow to Agassiz Pool; however, during high streamflow periods, the Mud River diversion and several smaller inflows contribute substantial inflow to Agassiz Pool. A4, located on Branch 200 of Judicial Ditch 11, receives all of its flow from field runoff and drains to Farmes Pool. Although A4 is not located within Agassiz NWR (fig. 1), Farmes Pool is located within Agassiz NWR and is managed cooperatively by Agassiz NWR, the MNDNR, and the RLWD. Outflows from Farmes Pool were monitored from 2007 through 2009 by the RLWD (Red Lake Watershed District, 2010).

Outflow sites from Agassiz NWR for this study included the two main Agassiz Pool outlet structures, A2 and A5 (figs. 1–2). A2 and A5 are regulated by water control structures (WCS) to maintain a desired pool elevation for Agassiz Pool that meets Agassiz NWR management objectives. At A2, the WCS consists of two 14-foot wide radial gates and a 2.63-foot screw gate with a combined maximum capacity of about 4,500 ft^3/s (Gregg Knutsen, U.S. Fish and Wildlife Service, written commun., 2011). The WCS at A5 became operational in 2008 and consists of a stop-log structure with three 4.5-foot culverts that are designed for a maximum capacity of about 350 ft^3/s (Gregg Knutsen, U.S. Fish and Wildlife Service, written commun., 2011). Before the spring of 2008 when the study started, A2 was the principal WCS used in regulating Agassiz Pool's elevation. From the time when Agassiz Pool became an operational impoundment in 1940, until A5 became operational in spring 2008, most of Agassiz Pool's outlflow went into the Thief River by way of the channel downstream from A2 (Gregg Knutsen, U.S. Fish and Wildlife Service, written commun., 2011). Between 2008 and 2010, several atypical water-management activities were occurring for the two outflow sites. From March 2008 through September 2009, the channel downstream from the WCS at A2 was in various stages of construction. The channel slope and banks were being restructured in an effort to reduce erosion downstream from the WCS. As a result, from August through October of 2008, the gates at A2 were closed completely while construction work was being done, and A5 was used to manage the elevation of Agassiz Pool. In late summer of 2009, scheduled drawdown of Agassiz Pool was initiated and continued through the open-water period in 2010. Scheduled drawdown of Agassiz Pool is initiated every 10 years as prescribed by the Agassiz NWR Habitat Management Plan (U.S. Fish and Wildlife Service, 2007).

Purpose and Scope

The primary purpose of this report is to assess the nutrient and sediment conditions of rivers and drainage ditches entering and exiting Agassiz NWR. Samples were collected at 6 sites (4 inflow, 2 outflow) and analyzed for nutrients and suspended sediment from April through October during 2008, and March through October during 2009 and 2010. Continuous water temperature, specific conductance, DO, pH, and turbidity data also were collected from 2008 through 2010. Constituent loads were estimated for nutrients and suspended sediment using water-quality and streamflow data. Regression equations between continuously measured data and suspended-sediment data also were developed. A secondary purpose of this report is to present a potential water-quality and streamflow monitoring program design for use at Agassiz NWR. Results from this study can be used by resource managers to address identified impairments and protect wildlife habitat and public water supply, and may contribute toward developing more effective water-management plans for Agassiz NWR.

Methods

The following sections describe the methods used for the collection and analysis of water-quality and streamflow data collected by the USGS and USFWS at Agassiz NWR from 2008 to 2010. Data were collected from six sites at or near Agassiz NWR included streamflow, and discrete and continuous water-quality data.

Streamflow Data Collection

Stream stage was measured continuously by the USGS at six sites during open-water periods (no ice cover) (fig. 1,

Figure 2. Inflow monitoring sites (A1, A3, A4, and SG140) and outflow monitoring sites (A2, A5) in and near Agassiz National Wildlife Refuge, northwest Minnesota.

table 1). The open-water period included April through October in 2008 and March through October in 2009 and 2010. Stage and instantaneous discharge were measured approximately eight times per year to compute continuous streamflow from stage-discharge rating curves using methods described in Turnipseed and Sauer (2010) and Sauer and Turnipseed (2010). Data for stream stage and streamflow were stored in the National Water Information System (NWIS) (U.S. Geological Survey, 2012)

Water-Quality Data Collection

Water-quality samples were collected during open-water periods by USGS personnel at six sites located on rivers and drainage ditches flowing into and out of Agassiz NWR from 2008 to 2010 (fig. 1, table 1). Water-quality measurements including water temperature, specific conductance, DO, pH, and turbidity were continuously recorded at the six sites.

Sample Collection

Discrete water-quality samples were collected approximately eight times a year at each of the six sites throughout a wide range of streamflow. The total number of samples collected at each site ranged from 20 to 25 samples. Fewer samples were collected at some sites because there were more periods of zero streamflow, which precluded sample collection. Samples were collected following equal-width increment methods using depth-integrated samplers and processed using protocols described in U.S. Geological Survey (variously dated). Samples were analyzed for nutrients (total ammonia plus organic nitrogen, dissolved nitrate plus nitrite, dissolved ammonia, total nitrogen, dissolved orthophosphorus, and total phosphorus) by the USGS National Water Quality Laboratory in Denver, Colorado, following procedures described in Fishman (1993). Samples also were analyzed for suspended-sediment concentration by the USGS Iowa Water Science Center Sediment Laboratory in Iowa City, Iowa, following procedures described in Guy (1969). There were instances when samples for nutrients arrived compromised at the laboratory, resulting in fewer valid results for some of the constituents and sites. Results from the analyses were stored in the USGS NWIS database (*http://nwis.waterdata.usgs.gov/nd/nwis/qw*). Field measurements, including water temperature, DO, pH, and specific conductance also were collected with each sample following protocols described in Wilde and Radke (1998).

To maintain proper quality assurance and quality control (QA/QC) of water-quality data, protocols for instrument calibration (Wilde and Radke, 1998) and equipment cleaning (Wilde and others, 1998) were followed during the study. Associated blank and replicate water-quality samples also were collected by USGS personnel periodically during the study. QA/QC sample data were stored in the USGS NWIS database (*http://waterdata.usgs.gov/nwis*).

Continuous Water-Quality Monitoring

Water-quality measurements including water temperature, specific conductance, DO, pH, and turbidity were recorded using continuous water-quality monitors during open-water periods from 2008 to 2010. The water-quality monitors were configured to collect data in 15-minute intervals. In 2008 and 2009, numerous equipment issues were encountered, resulting in as little as 11 percent (A1 in 2008) data completion to as much as 60 percent (A3 in 2008) completion. In 2010, fewer equipment issues were encountered resulting in 24 (A2) to 76 percent (SG140) data completion. The operation of the water-quality monitors and record computations were conducted according to methods described in Wagner and others (2006). With guidance from USGS personnel, USFWS personnel operated the monitors on site and serviced the monitors off site (RLWD Laboratory, Thief River Falls, Minn.). Accuracy ratings for water-quality records are presented in table 18 of Wagner (2006) and are based on combined sensor fouling and calibration drift corrections applied to the record. Accuracy

Table 1. Monitoring sites in and near Agassiz National Wildlife Refuge, northwest Minnesota, 2008 to 2010.

[NWR, National Wildlife Refuge]

Site identification number (fig. 1)	U.S. Geological Survey station number	Site name	Relation to Agassiz NWR	Latitude	Longitude
A1	05075694	Branch 1 of Judicial Ditch 11 above Mud River Pool	Inflow	48°23′22″	95°53′01″
A2	05075750	Judicial Ditch 11 below Agassiz Pool	Outflow	48°18′53″	96°00′35″
A3	05075720	Judicial Ditch 11 above Agassiz Pool	Inflow	48°19′27″	95°51′07″
A4	05075930	Branch 200 of Judicial Ditch 11 above Farmes Pool	Inflow	48°15′57″	95°51′07″
A5	05075697	Northwest outlet of Agassiz Pool	Outflow	48°20′41″	96°01′47″
SG140	05075690	Thief River inlet to the Agassiz NWR	Inflow	48°25′36″	95°58′12″

ratings for this study were downgraded because of protocol issues beyond the control of USGS. Continuous water-quality monitoring data collected and reported were subject to the qualified professional judgment of the hydrographer and followed standard USGS procedures as closely as possible. For all sites and years, the water-quality records for water temperature and specific conductance were considered good, DO and pH were considered fair, and turbidity was considered poor.

Data Analysis

The resulting streamflow and water-quality data were analyzed or summarized using several statistical and graphical techniques. Boxplots were used to compare concentrations of selected water-quality constituents. For constituents containing censored data (data that contains values less than laboratory reporting levels), the paired generalized Wilcoxon test was used to test for differences among sites (Helsel and Cohn, 1988). The paired generalized Wilcoxon test is a nonparametric test that determines the probability (p) that the distribution of the dataset is similar to the distribution of another dataset within a 95-percent confidence interval (p less than 0.05). For constituents without censored data, the Wilcoxon rank sum test (Helsel and Hirsch, 2002) was used.

Constituent load is a function of the volumetric rate of water passing a point in the stream and the constituent concentration within the water. Constituent loads for sediment and nutrients at all six sites were estimated using S-LOADEST, a software program based on the FORTRAN version developed by Runkel and others (2004). Given a time-series of streamflow data, additional data variables and constituent concentrations, the S-LOADEST program assists the user in developing a regression model for the estimation of constituent loads (calibration; Runkel and others, 2004). Explanatory variables of the predefined regression models in the S-LOADEST program include various functions of streamflow, decimal time, and additional user specified variables (table 2). The user can choose from the predefined models or allow the software to automatically choose the best-defined model. The formulated regression model then is used to estimate loads during a specified period of time (estimation).

The calibration and estimation procedures within S-LOADEST are based on three statistical estimation methods, Adjusted Maximum Likelihood Estimate (AMLE), Maximum Likelihood Estimate (MLE), and Least Absolute Deviation (LAD; Runkel and others, 2004). The first two methods, AMLE and MLE, are used when the residuals (difference between the measured and estimated value) are normally distributed. The primary estimation method within S-LOADEST is the AMLE, but for the special case when no censored data are present, the AMLE method converges to the MLE method (Runkel and others, 2004). The third method, LAD, is used when the residuals are not normally distributed, but cannot be used for censored data. The calibration and estimation procedures within S-LOADEST can account for non-normal data distributions, seasonal and long-term cycles, censored data, biases associated with using logarithmic transformations (retransformation bias), and serial correlations of the residuals (Runkel and others, 2004). A complete discussion of the theory and principles behind the calibration and estimation procedures used in S-LOADEST are given in Runkel and others (2004).

For this application, the regression model was selected by the user from one of the predefined models and the AMLE method was used for all sites and constituents (table 3). Depending on the constituent and site, streamflow (model 1) or streamflow and seasonality (model 4) were selected as explanatory variables for the regression models. These relatively simple models were used because of the small watershed area, range of flows, and small number of samples, and because data were collected over a relatively short period. The explanatory variables were considered to be statistically significant if the probability value (p-value) was less than 0.05 for the t-statistic. In using the AMLE method, normal

Table 2. Predefined regression models from S-LOADEST (Runkel and others, 2004).

[ln = natural logarithm; $\hat{}(L)$= estimated load, Q=centered streamflow; $a_0,a_1,a_2,a_3,a_4,a_5,a_6$, coefficients of the model; dtime = centered decimal time; sin, sine; cos, cosine; π,pi]

Model number	Regression model
1	$\ln \hat{}(L) = a_0 + a_1 \ln Q$
2	$\ln \hat{}(L) = a_0 + a_1 \ln Q + a_2 \ln Q^2$
3	$\ln \hat{}(L) = a_0 + a_1 \ln Q + a_2 \mathrm{dtime}$
4	$\ln \hat{}(L) = a_0 + a_1 \ln Q + a_2 \sin(2\pi \mathrm{dtime}) + a_3 \cos(2\pi \mathrm{dtime})$
5	$\ln \hat{}(L) = a_0 + a_1 \ln Q + a_2 \ln Q^2 + a_3 \mathrm{dtime}$
6	$\ln \hat{}(L) = a_0 + a_1 \ln Q + a_2 \ln Q^2 + a_3 \sin(2\pi \mathrm{dtime}) + a_4 \cos(2\pi \mathrm{dtime})$
7	$\ln \hat{}(L) = a_0 + a_1 \ln Q + a_2 \sin(2\pi \mathrm{dtime}) + a_3 \cos(2\pi \mathrm{dtime}) + a_4 \mathrm{dtime}$
8	$\ln \hat{}(L) = a_0 + a_1 \ln Q + a_2 \ln Q^2 + a_3 \sin(2\pi \mathrm{dtime}) + a_4 \cos(2\pi \mathrm{dtime}) + a_5 \mathrm{dtime}$
9	$\ln \hat{}(L) = a_0 + a_1 \ln Q + a_2 \ln Q^2 + a_3 \sin(2\pi \mathrm{dtime}) + a_4 \cos(2\pi \mathrm{dtime}) + a_5 \mathrm{dtime} + a_6 \mathrm{dtime}^2$

Table 3. Regression model characteristics for S-LOADEST models for six sites in and near Agassiz Nation Wildlife Refuge, northwest, Minnesota.

[N, nitrogen; P, phosphorus; ft³/s, cubic feet per second]

Site identification number	Regression model characteristic	Total ammonia plus organic nitrogen as N	Dissolved ammonia as N	Dissolved nitrate plus nitrite as N	Total nitrogen as N	Dissolved orthophosphorus as P	Total phosphorus as P	Suspended sediment
A1	S-LOADEST model number	1	4	4	4	1	1	4
	Coefficient of determination	98.7	87.9	96.8	98.0	85.4	90.7	88.2
	Standard error of prediction as a percent of total load from 2008 to 2010	3.30	30.0	50.5	13.5	16.7	12.1	52.9
	Range of streamflow (in ft³/s) for developing regression	0.53–161						
	Number of samples	20	20	20	20	20	20	20
A2	S-LOADEST model number	1	4	4	4	4	4	4
	Coefficient of determination	88.9	76.9	82.9	90.1	76.8	92.3	68.2
	Standard error of prediction as a percent of total load from 2008 to 2010	3.95	41.9	165	10.6	103	9.7	58.0
	Range of streamflow (in ft³/s) for developing regression	24.0–1,130						
	Number of samples	25	25	25	25	25	25	25
A3	S-LOADEST model number	1	4	1	1	1	1	1
	Coefficient of determination	96.5	77.2	39.3	80.8	78.0	90.8	92.0
	Standard error of prediction as a percent of total load from 2008 to 2010	2.2	15.3	24.4	5.7	18.4	8.5	7.9
	Range of streamflow (in ft³/s) for developing regression	11.0–679						
	Number of samples	25	24	24	24	24	25	24
A4	S-LOADEST model number	1	4	1	1	1	1	1
	Coefficient of determination	95.7	80.1	41.5	81.4	87.1	96.5	92.5
	Standard error of prediction as a percent of total load from 2008 to 2010	2.9	13.0	45.3	7.0	10.6	5.6	10.2
	Range of streamflow (in ft³/s) for developing regression	1.70–227						
	Number of samples	23	22	22	22	22	23	23
A5	S-LOADEST model number	1	4	4	4	4	4	1
	Coefficient of determination	92.0	51.6	61.1	86.3	61.8	87.4	82.4
	Standard error of prediction as a percent of total load from 2008 to 2010	2.3	33.5	56.1	6.1	21.8	5.4	9.0
	Range of streamflow (in ft³/s) for developing regression	12.0–329						
	Number of samples	21	21	22	21	21	22	22
SG140	S-LOADEST model number	1	4	4	4	1	4	1
	Coefficient of determination	98.2	67.1	48.9	95.8	68.9	95.4	95.4
	Standard error of prediction as a percent of total load from 2008 to 2010	2.4	71.4	79.8	6.0	17.3	7.9	8.9
	Range of streamflow (in ft³/s) for developing regression	2.4–543						
	Number of samples	24	24	24	24	24	24	23

distribution (normality) of the dataset is assumed. The validity of the normality assumption for the residuals was examined using the Turnbull-Weiss likelihood ratio normality statistic (Turnbull and Weiss, 1978). If the p-value from the Turnbull-Weiss statistic was less than 0.05, the residual plots were examined for homoscedasticity (equal statistical variances) and normality. There were some cases where the p-value was less than 0.05, but the AMLE method was used because the dataset contained censored data.

As a measure of how much variability in the dependent variable is explained by the independent variable and the S-LOADEST regression equation, coefficients of determination (R^2) were computed and expressed as a percentage. R^2 is a number, 0 through 1, that when multiplied by 100 is interpreted as the percentage of the variability in the dependent variable explained by the independent variable(s) and the regression equation (Helsel and Hirsch, 1995). Generally, a larger R^2 value indicates a better relation. For example, an R^2 of 100 percent indicates that all of the variability in the dependent variable is explained by the independent variable(s). However, a large R^2 value does not guarantee the relation is useful (Neter and others, 1996). For example, if estimates require extrapolation outside of the observed independent variables, the estimates may not be accurate. Unless constituent concentrations were highly variable, R^2 values were expected to be large for the S-LOADEST models because the dependent variable in the S-LOADEST models (constituent load) is a function of one of the independent variables (streamflow).

As a measure of uncertainty in the load estimates, the standard error of prediction (SEP) is provided in S-LOADEST output (Runkel and others, 2004). To compare uncertainty among sites with large differences in loads, the SEP was expressed as a percentage of the total estimated load during the 3-year period for each site and constituent.

Because loads are streamflow-dependent and streamflow varied greatly from year to year and from site to site, flow-weighted concentrations for constituents for each year at each site were calculated to better compare changes in conditions over time and between sites. Flow-weighted concentrations for each site were calculated by dividing the annual load (open-water period) by annual mean streamflow (open-water period), and applying appropriate conversion factors for dimensional units:

$$C_{FWA} = (L/Q_{\text{Annual}}) \times 5.08 \times 10^{-4}, \tag{1}$$

where

C_{FWA} represents the flow-weighted concentration, in milligrams per liter,

L represents the annual constituent load, in pounds per year, and

Q_{Annual} represents the annual mean streamflow, in cubic feet per second.

Mean monthly flow-weighted concentrations also were computed:

$$C_{FWM} = (L/Q_{\text{Mean monthly}}) \times 0.186, \tag{2}$$

where

C_{FWM} represents the flow-weighted concentration, in milligrams per liter,

L represents the mean monthly constituent load, in pounds per year, and

$Q_{\text{Mean monthly}}$ represents the mean monthly streamflow, in cubic feet per second.

Another method that can be used for estimating loads involves the development of a regression equation using continuously monitored water-quality data. Helsel and Hirsch (1995) described the regression method used to estimate water-quality constituent concentrations (dependent variable) in terms of other surrogate constituents or physical properties (independent variables). Given an estimated constituent concentration from the regression, a continuous record of streamflow can be used to estimate load. Many studies have been completed using this method to relate independent variables of continuously measured water-quality data such as turbidity, specific conductance, and water temperature to dependent variables of water-quality constituents such as alkalinity, dissolved solids, total suspended-sediment, chloride, sulfate, atrazine, and fecal coliform bacteria concentrations (Christensen and others, 2000; Christensen, 2001; Galloway and Green, 2004; Galloway and others, 2008; Ryberg, 2006 and 2007). For this study, simple regressions were developed only for suspended sediment because of large gaps in the continuous water-quality data and preliminary analysis indicated poor relations for other constituents. The concentration of suspended sediment is often strongly correlated to turbidity and streamflow, but specific conductance, water temperature, and DO also were evaluated for correlation. To determine which independent variable or variables to include in each regression equation, a stepwise procedure was used (Ott, 1993). If there was a significant (p-value less than 0.05) correlation between the independent variable and suspended-sediment concentration, then the independent variable was included in the regression equation. Graphical plots were created to determine linearity and visually examine relations and grouping of the data. When developing regressions, the independent variable, dependent variable, or both were log transformed to eliminate curvature in the data and simplify the analysis of the data (Ott, 1993).

To compare between the two estimation methods, R^2 values also were computed for the regression equations. Also, for both methods, the measured instantaneous concentrations were plotted against the estimated concentrations and the relative percent differences (RPD) were computed. The RPD is an indicator of the ability of the regression equation to estimate constituent concentrations and is computed as (Ryberg, 2006):

$$RPD = |(E\text{-}M)/M \times 100|, \quad (3)$$

where

E is the constituent concentration estimated from the regression equation, in milligrams per liter and

M is the measured constituent concentration, in milligrams per liter.

The 2010 continuous water-quality datasets for A3 and A5 were complete enough to compute daily sediment loads from the regression equations and compare them with sediment loads estimated from S-LOADEST. Estimated sediment loads were computed by multiplying the daily suspended-sediment concentrations estimated from the regression equations by the daily streamflow and by a conversion factor. Because the regressions were developed in terms of logarithm-transformed constituent concentrations, a bias correction factor (BCF) was applied to account for retransformation back to the original units (Duan, 1983). Calculation of the BCF is shown below:

$$BCF = \frac{\sum_{i=1}^{n} 10^{e_i}}{n}, \quad (4)$$

where

e_i is the regression residual, in log units, and

n is the number of samples used to develop the regression relation.

Hydrologic Characteristics

Of the 6 monitoring sites, 4 sites were inflow sites (A1, A3, A4, and SG140) and 2 were outflow sites (A2 and A5; fig. 1). The hydrologic characteristics of the six monitoring sites (figs. 3 and 4, table 4) were affected by water management activities at outflow sites A2 and A5, precipitation, and evapotranspiration.

Overall, among the four inflow sites, the annual mean streamflow and the total streamflow volume for the period from 2008 through 2010 were greatest at SG140 and least at A1 (table 4). The annual mean streamflow (open-water period) at SG140, varied from 36.7 ft^3/s in 2008 to 170 ft^3/s in 2010 (table 4). The annual mean streamflow at A1 varied from 1.77 ft^3/s in 2008 to 11.5 ft^3/s in 2010 (fig. 3, table 4). Of the four inflow sites, the greatest daily streamflow of 820 ft^3/s occurred at A3 on May 26, 2010 (fig. 3, table 4). Annually, the streamflow volume at sites SG140 and A3 accounted for about 88 to 92 percent of the measured inflow.

As previously mentioned in the Description of Study Area section, the combined streamflow from inflow sites, A1, A3, and SG140 did not include the total streamflow into Agassiz Pool. Streamflow that is diverted into the Mud River diversion and smaller inflows also contribute to Agassiz Pool. These additional streamflows account for a substantial percentage of the total inflow to Agassiz Pool during high streamflow periods, such as those that occurred in the early part of 2009 and most of 2010. Based on comparison of measurements made at A3 and measurements made at long-term USGS gaging station 05075700 (Mud River near Grygla, fig. 1) from 2008 to 2010 (measurements ranged from 123 and 1,710 ft^3/s), the Mud River diversion accounts for approximately one-half of the streamflow coming into Agassiz Pool from the Mud River. The exact volume of the smaller inflows is unknown and more difficult to estimate. However, based on mass balance calculations for Agassiz Pool from March 20 through April 1, 2009, (when the WCS at A2 and A5 were closed) and for the 2010 period (when the WCSs at A2 and A5 were open), it was estimated that doubling of the inflows at A3 (to account for the Mud River diversion) and doubling of inflows at A1 (to account for several smaller inflows) accounted for the missing inflow to Agassiz Pool during the high streamflows of 2009 and 2010. During the lower streamflow year of 2008, it is estimated that nearly all streamflow into Agassiz Pool was accounted for by the three inflow sites, A1, A3, and SG140.

Between the two outflow sites, A2 and A5, streamflow was greatest at A2 for all years (fig. 4, table 4). Annual mean streamflow varied from 86 ft^3/s in 2008 to 381 ft^3/s in 2010 at A2 compared with 23.5 ft^3/s in 2008 to 115 ft^3/s in 2010 at A5 (table 4). During the period of data collection, water management activities affected the streamflow characteristics for the two outflow sites. From 2008 to October 2009, for both sites, there were periods of streamflow when the WCSs were opened, alternated with periods of no streamflow when the WCSs were closed. As part of the management plan for Agassiz NWR, scheduled drawdown of Agassiz Pool was initiated in 2009. Between August 25 and 28, 2009, all stop logs were removed from the WCS at A5. At A2, the WCS was periodically opened and closed during the month of October, until the WCS was completely open at the end of the month (fig. 4). For the remainder of the data collection period, the WCSs at A2 and A5 were completely open, resulting in a flow-through system that resulted in nearly continuous streamflow in 2010 (fig. 4).

Differences in annual precipitation and evapotranspiration affected the annual mean streamflow and total volume of streamflow for all sites. Annual precipitation (March through October) was greatest in 2010 totaling 30.7 inches, and in 2008 and 2009 annual precipitation was roughly equivalent with total amounts equal to 18.6 inches and 17.7 inches, respectively (Gregg Knutsen, U.S. Fish and Wildlife Service, written commun., 2011). Annual potential evapotranspiration (March through October), calculated from the Penman equation for a weather station located near Mavie, Minn. (fig. 1) was least in 2009 (35 inches) compared with 39.4 inches in 2008 and 40.6 inches in 2010 (North Dakota Agricultural Weather Network, 2012). Despite similar amounts of

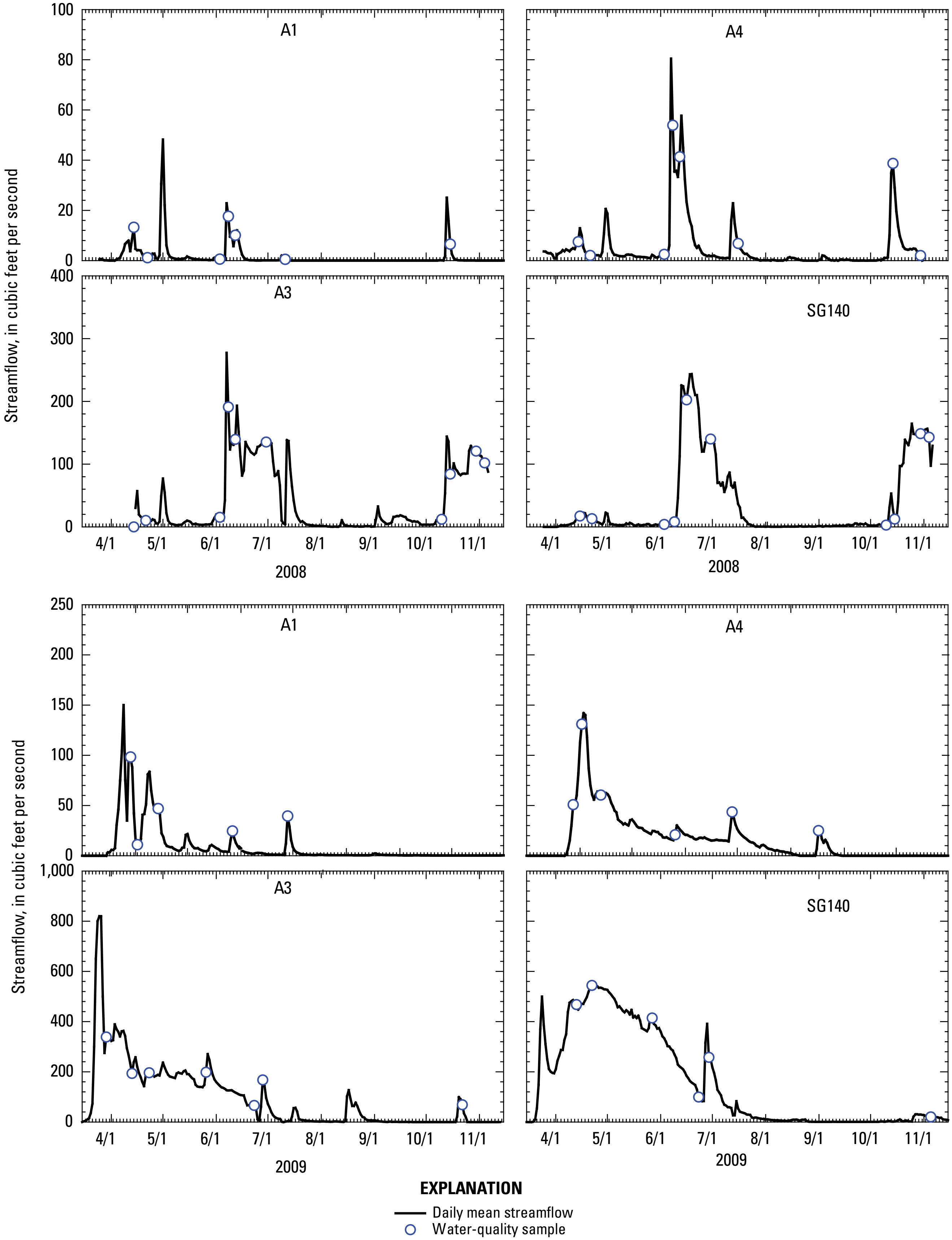
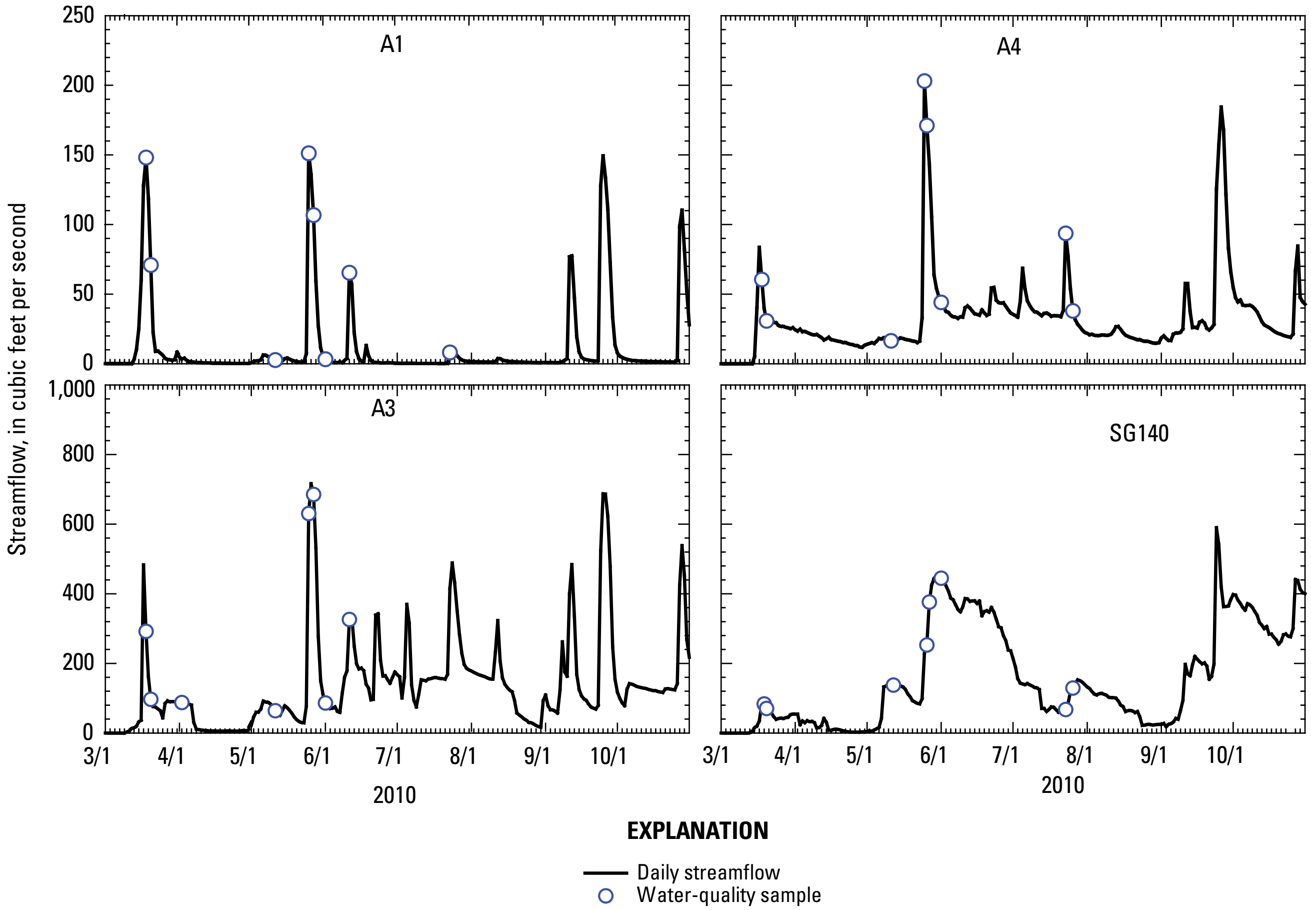


Figure 3. Daily streamflow and water-quality samples for inflow sites in and near Agassiz National Wildlife Refuge, northwest Minnesota, 2008 to 2010.



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precipitation for March through October in 2008 and 2009, runoff from snowmelt in the spring of 2009 combined with less evapotranspiration resulted in about 3 to 5 times greater total volume in 2009 than 2008 for all sites (table 4). At all sites, the greatest annual mean streamflow was in 2010 and the least annual mean streamflow was in 2008 (table 4). The total volume of streamflow in 2010 was 4.2 and 7.6 times greater than in 2008 at A3 and A1, respectively, and was 6.3 times greater, on average, for all sites (table 4).

Water-Quality Characteristics

Water-quality data were collected at all six sites from April through October of 2008, and March through October of 2009 and 2010. Discrete water-quality samples were collected throughout a wide range of streamflow and analyzed for nutrients and suspended sediment (figs. 3 and 4). Using the discrete sample data, comparisons between sites for nutrient and sediment concentrations were made. Based on continuous streamflow and discrete samples, constituent loads for nutrients and sediment were estimated using S-LOADEST. Annual and mean monthly flow-weighted concentrations were computed from estimated loads and streamflow and compared between sites and years. Sediment flux in Agassiz Pool was estimated by subtracting inflow loads from outflow loads and was evaluated for a net change in sediment. Continuous water-quality properties measured during this study were discussed and compared with water-quality standards. Finally, as another method to estimate suspended-sediment loads, regression equations were developed and compared with estimates from S-LOADEST.

Nutrients

Sources of nutrients to rivers and ditches in Agassiz NWR include runoff from agricultural areas where fertilizers are applied or livestock production occurs. The Thief River Watershed is sparsely populated with few point sources for nutrients, and the land use is primarily cultivated agricultural fields with few livestock operations (U.S. Fish and Wildlife Service, 2005). Wildlife, such as waterfowl also can be a source of nutrients to water bodies. Groundwater can be a source of nutrients, but little is known about the contribution

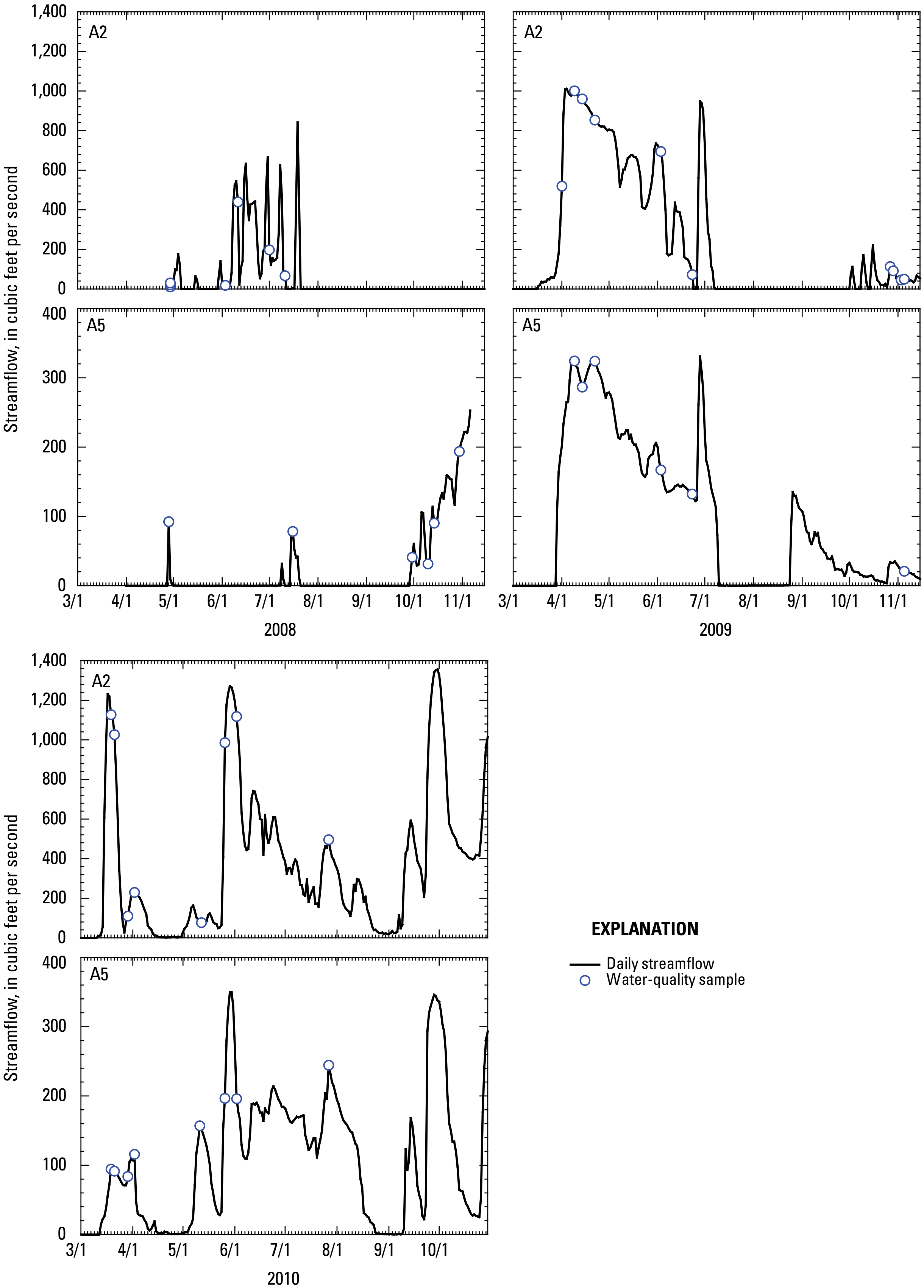


Figure 4. Daily streamflow and water-quality samples for outflow sites in Agassiz National Wildlife Refuge, northwest Minnesota, 2008 to 2010.

Table 4. Streamflow statistics for sites in and near Agassiz National Wildlife Refuge, northwest Minnesota, 2008 to 2010.

Site identification number	Year (open-water period)	Annual mean (cubic feet per second)	Maximum (cubic feet per second)	Minimum (cubic feet per second)	Total volume for period (acre-feet)
Inflow sites					
A1	2008–2010	7.60	151	0.0	11,000
	2008	1.77	48.2	-0.00	774
	2009	8.70	150	0.00	4,340
	2010	11.5	151	0.00	5,900
A3	2008–2010	94.0	820	0.00	135,000
	2008	41.3	277	0.00	16,900
	2009	90.5	820	0.00	46,300
	2010	139	716	0.00	71,600
A4	2008–2010	17.9	203	0.00	26,900
	2008	5.13	80.4	0.00	2,250
	2009	15.2	142	0.00	7,940
	2010	32.6	203	0.00	16,800
SG140	2008–2010	119	590	0.00	180,100
	2008	36.7	244	0.00	16,500
	2009	145	562	0.00	76,100
	2010	170	590	0.00	87,500
Outflow sites					
A2	2008–2010	252	1,360	0.00	346,000
	2008	86.0	841	0.00	26,100
	2009	239	1,010	0.00	125,100
	2010	381	1,360	0.00	194,900
A5	2008–2010	81.2	445	0.00	117,000
	2008	23.5	253	-0.00	10,200
	2009	96.8	331	0.00	49,900
	2010	115	445	0.00	56,800

of nutrients by way of groundwater at Agassiz NWR. Given the sparse population of the watershed (in 2001, the average population density was 6 per square mile; U.S. Fish and Wildlife Service, 2005) and uncertainty of groundwater contribution, it is likely that the primary source of nutrients to rivers and ditches in the Thief River Watershed is from nonpoint sources in the form of agricultural runoff and also may include some nutrient inputs from wildlife. Within Agassiz NWR, processes such as mineralization, denitrification, and plant uptake all affect nutrient concentrations.

Concentrations

For all nutrient constituents, comparisons were made among sites, and between concentrations and established water-quality standards. Results from this study indicate that concentrations at all sites did not exceed the water-quality standard set for un-ionized ammonia by the State of Minnesota Minnesota Office of the Revisor of Statutes, 2012). The water-quality standard (Minnesota Office of the Revisor of Statutes, 2012) for drinking water was used as a benchmark for comparison of nitrate plus nitrite concentrations and concentrations at all sites were below the standard. Compared with the four inflow sites, the two outflow sites generally had significantly greater dissolved ammonia concentrations, significantly smaller nitrate plus nitrite concentrations, and no major differences in total ammonia plus organic nitrogen and total nitrogen. The ecoregion criteria (Minnesota Office of the Revisor of Statutes, 2012) for shallow lakes was used as a benchmark for comparison of total phosphorus concentrations, and other than inflow site A1, most of the inflow sites had median concentrations less than the ecoregion criteria, but the outflow sites had median concentrations at or above the ecoregion criteria. Overall, orthophosphorus and total phosphorus concentrations were significantly greater at inflow site A1 than any other site.

The Thief River from Thief Lake to Agassiz Pool is listed as impaired for high ammonia concentrations. The Minnesota State water-quality standard for ammonia is set for un-ionized ammonia and concentrations should not exceed 0.04 milligrams per liter (mg/L) (Minnesota Office of the Revisor of Statutes, 2012). Un-ionized ammonia (NH_3) concentrations, calculated from dissolved ammonia (NH^{4+}) and as a function of temperature and pH, were determined to be less than the 0.04 mg/L standard for all samples for all sites. Among the inflow sites (A1, A3, A4, and SG140), dissolved ammonia concentrations were not significantly different (fig. 5). Median dissolved ammonia concentrations at the inflow sites were between 0.020 mg/L as nitrogen (A4 and SG140) and 0.024 mg/L as nitrogen (A3). With the exception of inflow site A1, ammonia concentrations for the outflow sites were significantly greater than the inflow sites, with median concentrations of 0.043 mg/L as nitrogen (A2) and 0.062 mg/L as nitrogen (A5) (fig. 5). Greater dissolved ammonia concentrations at the outflow sites were likely caused by organic nitrogen being converted to ammonia (mineralization) during decomposition in Agassiz Pool.

A water-quality standard specific to streams is not established for nitrate plus nitrite, but the Minnesota drinking water-quality standard of 10 mg/L was used as a benchmark for comparison (Minnesota Office of the Revisor of Statutes, 2012). At concentrations greater than 10 mg/L, nitrate plus nitrite can cause human health problems (U.S. Environmental Protection Agency, 2012). Nitrate plus nitrite concentrations for all sites and samples were well below the 10 mg/L water-quality standard (fig. 5). Nitrate plus nitrite concentrations were significantly greater at the inflow sites A1, A3, and A4 compared with the outflow sites. Inflow site SG140, and outflow sites A2 and A5 all had the same median nitrate plus nitrite concentration of 0.040 mg/L as nitrogen (for sites SG140 and A2, the concentration of 0.040 mg/L nitrate plus nitrite also corresponds to the 25th percentile). For the other inflow sites, median nitrate plus nitrite concentrations ranged from 0.485 mg/L as nitrogen (A1) to 0.595 mg/L as nitrogen (A4). A decrease in nitrate plus nitrite at the outflow sites may be related to uptake by plants and algae within Agassiz Pool and denitrification (conversion of nitrate or nitrite to nitrous oxides and nitrogen gas). The nitrate plus nitrite concentrations at SG140 are likely similar to the outflow sites because of uptake and nitrification occurring 4 miles upstream in Thief Lake.

In Minnesota, water-quality standards have not been established for total ammonia plus organic nitrogen and total nitrogen (assumed to be total ammonia plus organic nitrogen plus nitrate plus nitrite as nitrogen) for streams. Consistent differences in concentration between inflow sites and outflow sites were not observed for total ammonia plus organic nitrogen or total nitrogen. Total ammonia plus organic nitrogen concentrations for all sites ranged from 0.6 mg/L as nitrogen at A4 to 3.0 mg/L as nitrogen at A2, and median total ammonia plus organic nitrogen ranged from 1.1 mg/L as nitrogen at A3 to 1.4 mg/L as nitrogen at A2 and A5 (fig. 5). Total nitrogen concentrations ranged from 0.74 mg/L as nitrogen at A4 to 6.27 mg/L as nitrogen at A1 and median total nitrogen concentrations ranged from 1.25 mg/L as nitrogen at SG140 to 1.88 mg/L at A4 (fig. 5).

Water-quality standards specific to streams have not been established for phosphorus in Minnesota, but ecoregion criteria established for total phosphorus in lakes were used as a benchmark for comparison. Comparing the total phosphorus data to the ecoregion nutrient criteria of 0.09 mg/L for shallow lakes in the Northern Glaciated Plains, the median concentration for inflow sites A3, A4, and SG140 were less than the ecoregion criteria (Minnesota Office of the Revisor of Statutes, 2012; fig. 6). For inflow site A1 and outflow sites A2 and A5, median concentrations were at or above the ecoregion criteria. Overall, orthophosphorus and total phosphorus concentrations were significantly greater at A1 than any other site (fig. 6). Most of the phosphorus at A1 was in the dissolved form as indicated by a median orthophosphorus concentration of 0.174 mg/L as phosphorus and the median total phosphorus concentration of 0.245 mg/L as phosphorus (for SG140, the median concentration of 0.008 mg/L orthophosphorus also corresponds to the 25th percentile). The reason for significantly greater phosphorus at A1 is unknown, but it may be related to A1 functioning solely as drainage ditch (there is no natural streamflow associated with A1). Total phosphorus concentrations at A5 were significantly greater than those at SG140 (directly upstream from A5), but total phosphorus concentrations for the two outflow sites were not significantly different from one another (fig. 6).

Loads

Other than inflow site SG140 and outflow site A5, for most sites and constituents, annual (open-water period) nutrient loads estimated using S-LOADEST were greatest in 2010, which was related, in part, to larger streamflow volume in 2010 (fig. 7, table 5). Also, other than nitrate plus nitrite and orthophosphorus, annual nutrient loads generally were greatest for outflow site A2, which is most likely related to a greater volume of streamflow compared with other sites (fig. 7, table 5). For inflow site SG140 and outflow site A5, the greater loads in 2009 may have been related to the timing of releases from Thief Lake. In 2009, releases from Thief Lake were greater than 200 ft^3/s beginning in mid-April through the beginning of June, whereas in 2008 and 2010 releases from Thief Lake greater than 200 ft^3/s did not occur until late May to early June (Joel Huener, Minnesota Department of Natural Resources, written commun., 2011). Higher concentrations of nutrients may have been associated with the earlier release because of decomposition and minimal plant activity during the winter. The greatest total ammonia plus organic nitrogen annual loads were estimated for A2, ranging from 57.7 tons

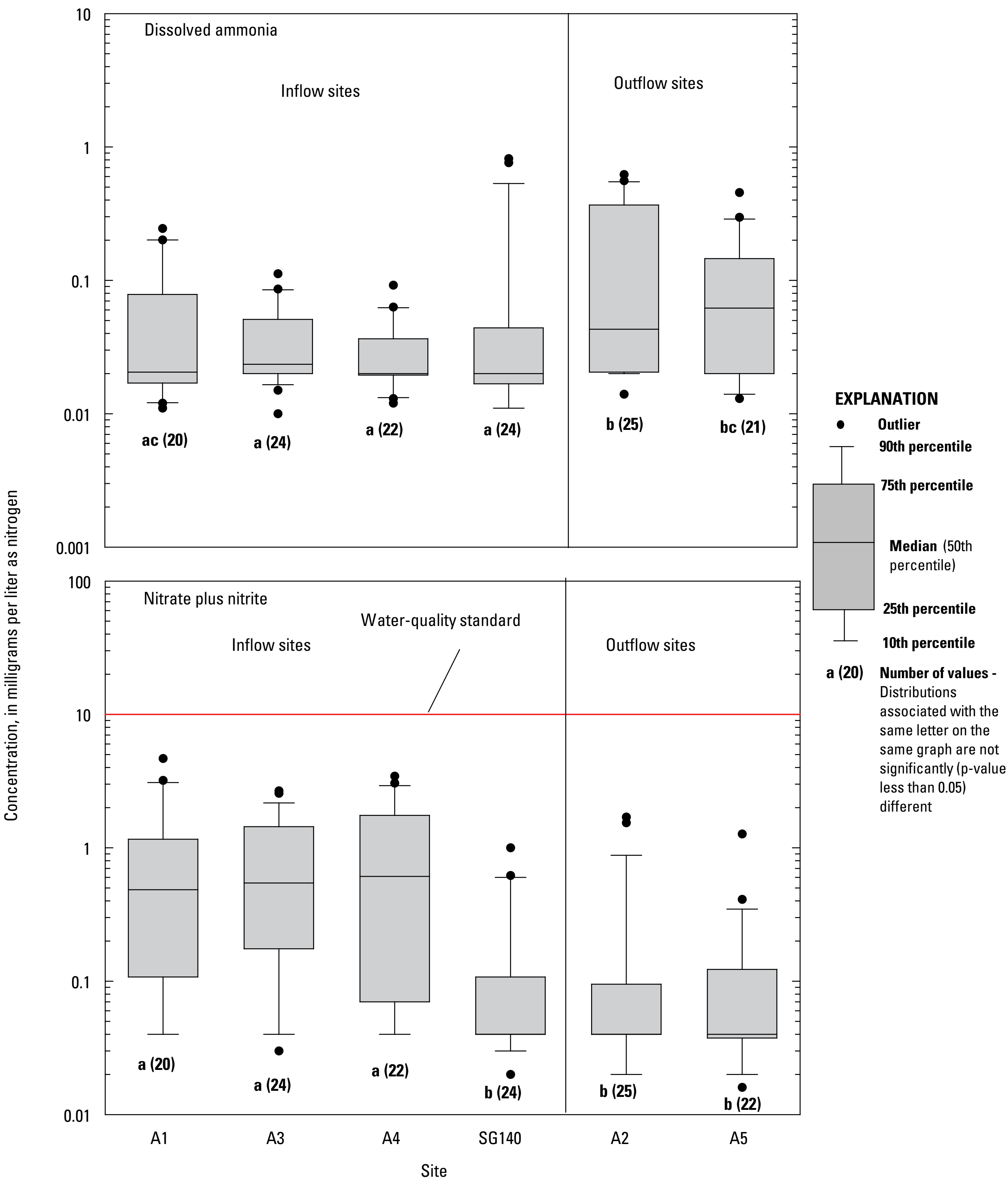
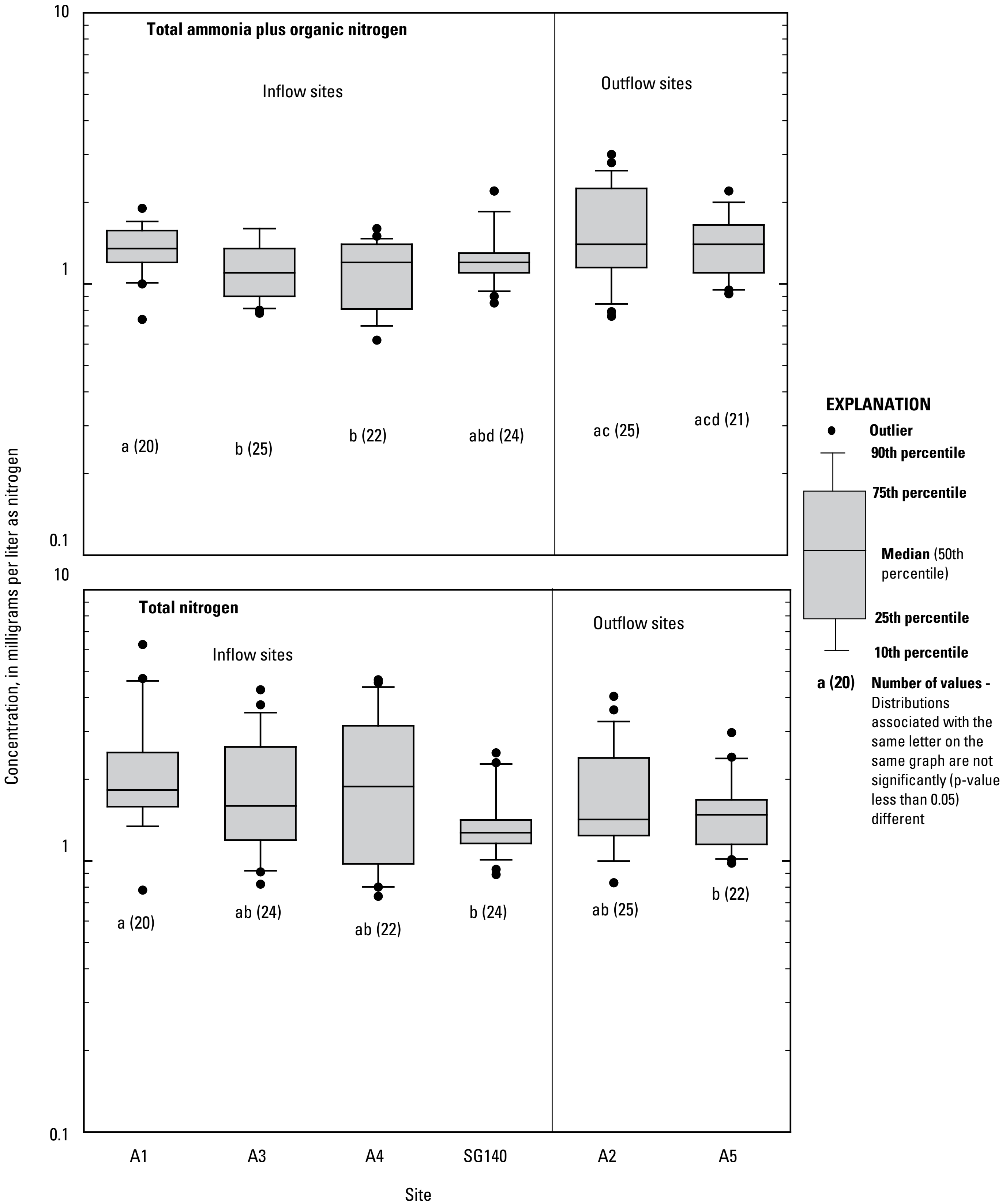


Figure 5. Distribution of nitrogen concentrations for inflow and outflow sites in and near Agassiz National Wildlife Refuge, northwest Minnesota, 2008 to 2010.



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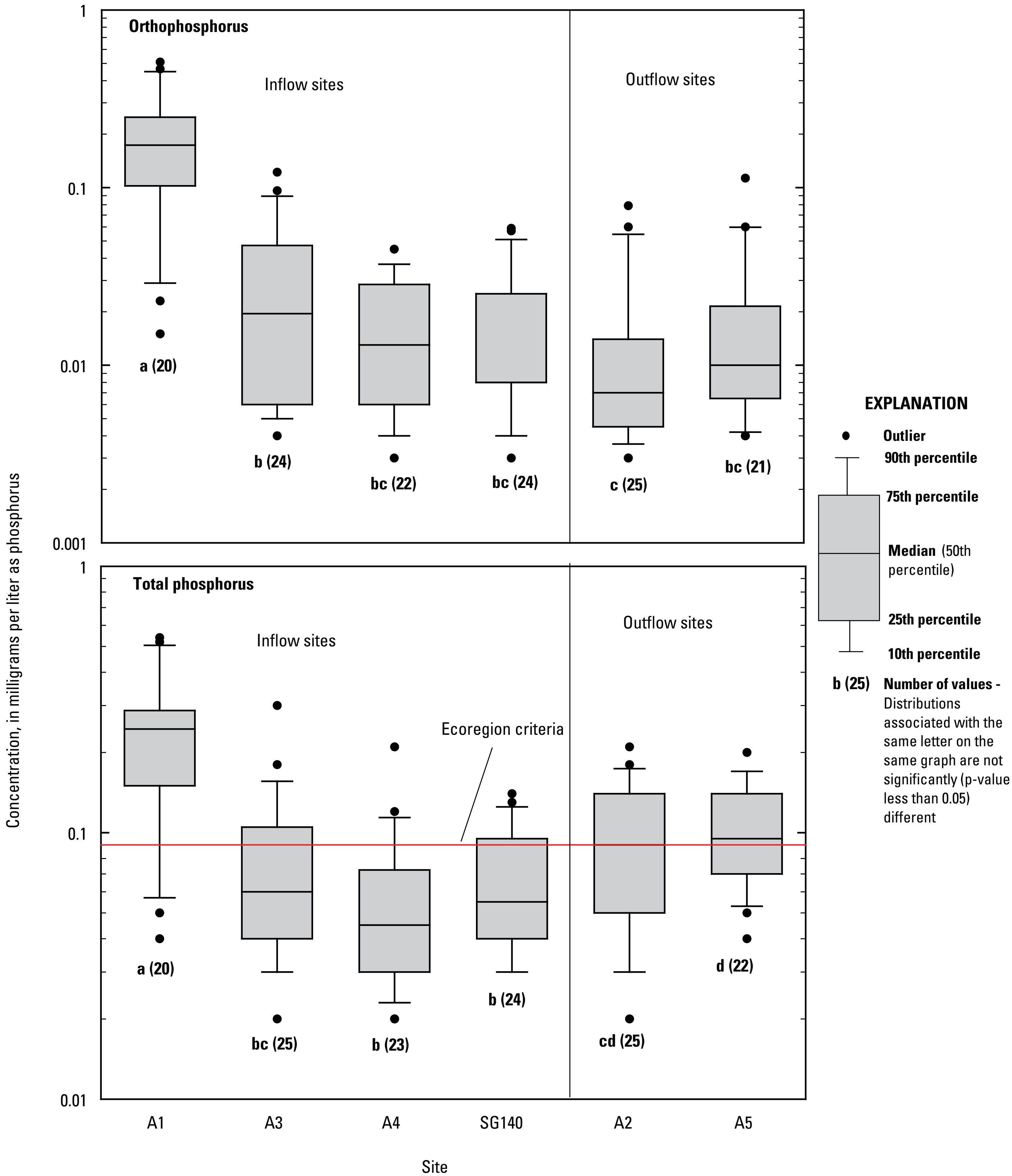

Figure 6. Distribution of phosphorus concentrations for inflow and outflow sites in and near Agassiz National Wildlife Refuge, northwest Minnesota, 2008 to 2010.

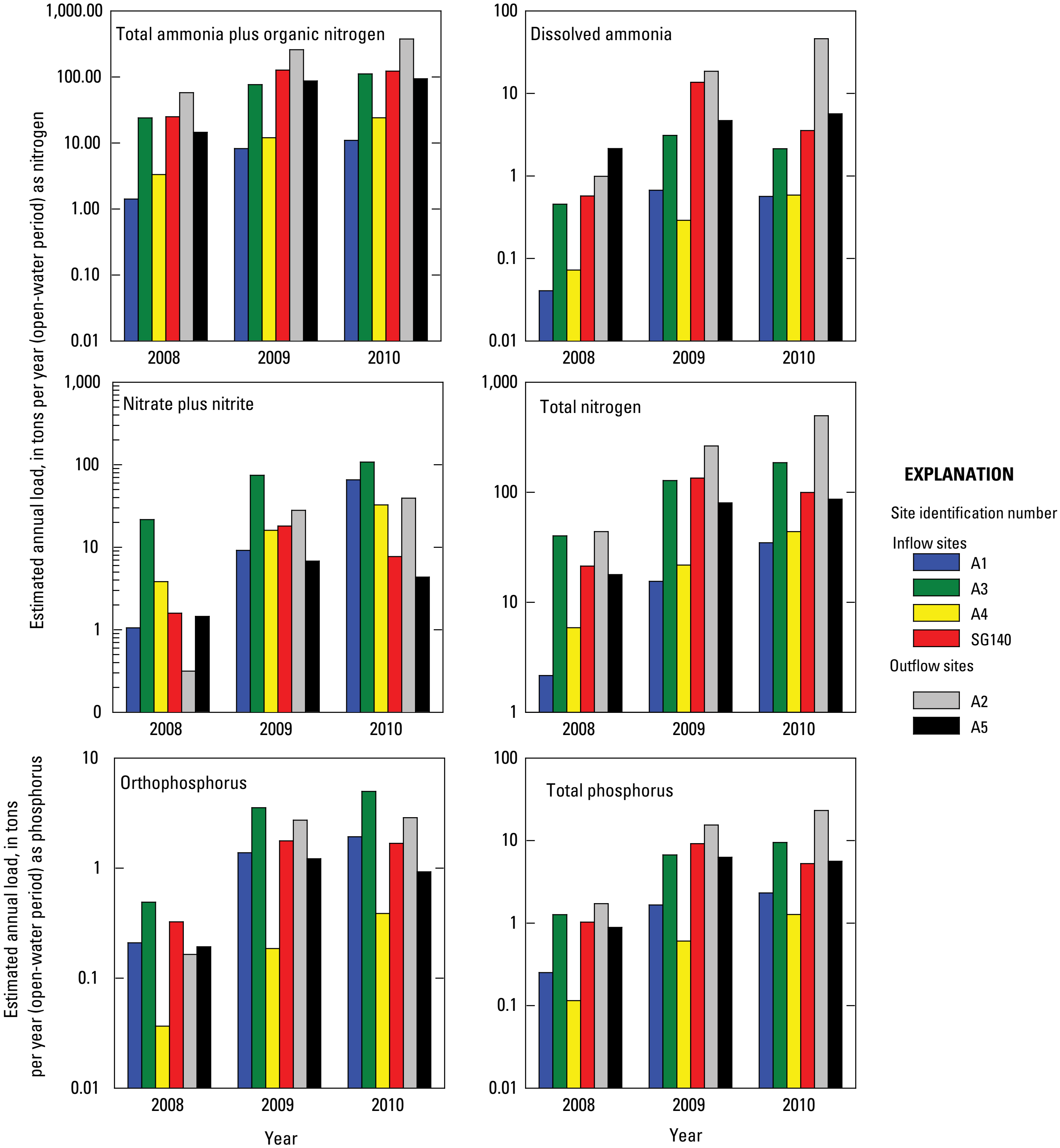

Figure 7. Estimated annual nutrient loads for inflow and outflow sites in and near Agassiz National Wildlife Refuge, northwest Minnesota, 2008 to 2010.

Table 5. Estimated annual loads (open-water period) for sites in and near Agassiz National Wildlife Refuge, northwest Minnesota, 2008 to 2010.

[yr, year; N, nitrogen; P, phosphorus]

Site identification number	Site name	Year (open-water period)	Total volume of streamflow (acre feet)	Total ammonia plus organic nitrogen as N (tons/yr)	Dissolved ammonia as N (tons/yr)	Dissolved nitrate plus nitrite (tons/yr)	Total nitrogen as N (tons/yr)	Dissolved orthophosphorus as P (tons/yr)	Total phosphorus as P (tons/yr)	Suspended sediment (tons/yr)
A1	Branch 1 of Judicial Ditch 11 above Mud River Pool	2008	1.77	1.41	0.04	1.06	2.16	0.21	0.25	9.15
		2009	8.70	8.20	0.67	9.14	15.5	1.38	1.65	53.5
		2010	11.5	10.9	0.57	65.6	34.7	1.93	2.33	142
A3	Judicial Ditch 11 above Agassiz Pool	2008	41.3	23.9	0.45	21.6	40.1	0.49	1.27	293
		2009	90.5	76.4	3.10	74.5	128	3.53	6.69	1,620
		2010	139	110	2.14	108	185	4.96	9.48	2,290
A4	Branch 200 of Judicial Ditch 11 above Farmes Pool	2008	5.13	3.32	0.07	3.83	5.87	0.04	0.11	23.5
		2009	15.2	12.0	0.29	16.0	21.8	0.19	0.60	133
		2010	32.6	24.0	0.59	32.5	43.8	0.39	1.27	286
SG140	Thief River inlet to the Agassiz NWR	2008	36.7	24.9	0.57	1.59	21.3	0.32	1.02	279
		2009	145	127	13.7	18.1	135	1.76	9.15	2,580
		2010	170	122	3.55	7.70	99.4	1.68	5.26	1,930
A2	Judicial Ditch 11 below Agassiz Pool	2008	86.0	57.7	0.99	0.32	43.9	0.16	1.72	867
		2009	239	259	18.6	28.0	263	2.72	15.4	8,950
		2010	381	375	46.0	39.3	496	2.87	23.2	29,000
A5	Northwest outlet of Agassiz Pool	2008	23.5	14.4	2.14	1.46	17.9	0.19	0.89	365
		2009	96.8	87.0	4.70	6.80	79.9	1.21	6.26	3,090
		2010	115	93.8	5.66	4.34	86.4	0.92	5.62	3,070

per year (tons/yr) in 2008 to 375 tons/yr in 2010 and the smallest annual loads were estimated for A1, ranging from 1.41 tons/yr in 2008 to 10.9 tons/yr in 2010 (fig. 7, table 5). Estimated annual loads for dissolved ammonia were greatest at the outflow sites (A2 and A5), with the loads ranging from 2.14 tons/yr in 2008 at A5 to 46.0 tons/yr in 2010 at A2. Estimated annual loads for nitrate plus nitrite were greatest for all years at inflow site A3, ranging from 21.6 tons/yr in 2008 to 108 tons/yr in 2010. The greater loads of nitrate plus nitrite for inflow site A3 are the result of relatively large streamflow volume (table 5) combined with significantly higher concentrations than other sites with larger streamflow volume (fig. 5). For total nitrogen, the estimated annual loads were greatest for A2, ranging from 43.9 tons/yr in 2008 to 496 tons/yr in 2010. The estimated annual loads for orthophosphorus were greatest for A3, ranging from 0.49 tons/yr in 2008 to 4.96 tons/yr in 2010 and the estimated annual load for total phosphorus was greatest at A2 for all years, ranging from 1.72 tons/yr in 2008 to 23.2 tons/yr in 2010. Estimated annual orthophosphorus and total phosphorus loads for A4 were smaller than for any other site (fig. 7, table 5). Of the total streamflow from inflow sites A1, A3, A4, SG140 in 2010, only 3 percent was accounted for by site A1; however, of the total load from inflow sites, 31 percent of nitrate plus nitrite, 27 percent of orthophosphorus, and 13 percent of total phosphorus was accounted for by A1 (fig. 7, table 5). Conversely, of the total streamflow from inflow sites in 2010, 39 percent was accounted for by SG140, but only 4 percent of the total load from inflow sites for nitrate plus nitrite was accounted for by SG140.

Streamflow volume (which was affected by precipitation patterns and releases from impoundments) and the growing season affected seasonal patterns of mean monthly nutrient loads (fig. 8 and figs. 3 and 4). Average monthly precipitation from 2008 to 2010 was greatest in September (4.70 inches), followed closely by June (4.43 inches) and least in March (0.97 inches) and April (1.19 inches). In response to snowmelt, rainfall, and releases from impoundments, the greatest streamflow months generally were April, May, June, September and October (figs. 3 and 4). For total ammonia plus organic nitrogen, total nitrogen, and total phosphorus, the greatest loads for most sites generally were in April, May, June, September and October, which corresponded to months of greater streamflow volume (fig. 8 and figs. 3 and 4). Mean monthly loads for dissolved ammonia tended to be the greatest in March and April, especially at outflow sites A2 and A5, and inflow site SG140, which may be related to spring releases from Agassiz Pool and Thief Lake (fig. 8). Streamflow in early spring, and especially releases from Thief Lake and Agassiz Pool may contain higher concentrations of dissolved ammonia because of the occurrence of overwinter decomposition. At outflow sites A2 and A5, substantial loads also occurred in the months of September and October (fig. 8). During the months of September and October, large volumes of water released at outflow sites A2 and A5 combined with a decrease in plant and algae growth may have resulted in more organic nitrogen being converted to ammonia. For nitrate plus nitrite, greater variability generally occurred at sites located downstream from Thief Lake and Agassiz Pool (inflow site SG140 and outflow sites A2 and A5) with the greatest loads occurring in early spring and smaller loads occurring in June, July and August (fig. 8). This pattern is possibly related to the upstream water bodies that are affected by the growing season. June, July, and August tend to be peak months in the growing season and nitrate is a readily used form of nitrogen for aquatic vegetation (Hem, 1989). At inflow site A3, the seasonal pattern of nitrate plus nitrite was less noticeable, with more consistent mean monthly loads during the year. Although inflow site A3 is downstream from an impoundment, it appears to be far enough downstream that aquatic vegetation did not noticeably reduce nitrate plus nitrite concentrations. Mean monthly orthophosphorus loads tended to be greatest in March, April, and May, although SG140 had the greatest load in June (fig. 8). Orthophosphorus is readily available for uptake by aquatic vegetation (Hem, 1989), which may explain the typically smaller loads in July and August for inflow site SG140 and outflow sites A2 and A5.

Flow-Weighted Concentrations

For most sites and constituents, estimated nutrient loads generally were greatest in 2010, but in many cases, annual flow-weighted concentrations were greatest in 2009 (fig. 9). The greater flow-weighted nutrient concentrations in 2009 may have been related to differences in the streamflow pattern between 2009 and 2010 (figs. 3 and 4). Although, the total rainfall was greater in 2010, most of the streamflow in 2009 occurred in the earlier one-half of the open-water period. Similar to annual loads, for all constituents for SG140 and for most constituents for A5, the annual flow-weighted concentrations were greatest in 2009, which may be related to the earlier pulse of streamflow (fig. 9). For A5, a substantially higher mean monthly dissolved ammonia concentration occurred in 2008 and may be a result of the new WCS at A5 being used as the primary WCS for managing Agassiz Pool (fig. 9). For outflow site A2, the annual flow-weighted concentration of all constituents other than dissolved ammonia and total nitrogen were greatest in 2009. The greater flow-weighted concentrations of dissolved ammonia and total nitrogen in 2010 at A2 may be related to scheduled drawdown of Agassiz Pool. Inflow site A1 had the greatest annual flow-weighted concentration of total nitrogen, orthophosphorus, and total phosphorus of all sites for all years, and the greatest concentrations of those constituents occurred in 2010 (fig. 9). Annual flow-weighted nitrate plus nitrite concentration at inflow site A1 in 2010 was five times larger than any other site in any other year.

Similar to the mean monthly loads, seasonal patterns of mean monthly flow-weighted concentrations were affected by releases from Thief Lake and Agassiz Pool and the growing season (fig. 8, fig. 10). Flow-weighted dissolved ammonia concentrations tended to be the greatest in March and April for all sites, which may be related to higher concentrations of dissolved ammonia because of the occurrence of overwinter decomposition. The greatest flow-weighted dissolved ammonia concentrations in March and April were at inflow site SG140 and outflow sites A2 and A5, which are affected by Thief Lake and Agassiz Pool. For these same sites, the effect of these water bodies also was evident for flow-weighted concentrations of nitrate plus nitrite, orthophosphorus, and total phosphorus as indicated by the higher concentrations early in spring followed by lower concentrations in midsummer and then higher concentrations in fall (fig. 10). For inflow sites A3 and A4, which are not affected directly by upstream impoundments, much less variability in flow-weighted concentrations of nitrate plus nitrite and orthophosphorus was observed. Flow-weighted concentrations of total nitrogen for inflow sites A3 and A4 were more variable because of greater variability in total ammonia plus organic nitrogen (fig. 10). Greater variability in total phosphorus for inflow sites A3 and A4 was likely related to variability in sediment because total phosphorus includes the phosphorus that is sorbed to particles (Hem, 1989). For inflow site A1, flow-weighted concentrations of nitrate plus nitrite, total nitrogen, orthophosphorus, and total phosphorus were often greater than any other site. Inflow site A1 is not affected by an impoundment, but seasonal patterns did not follow the pattern of inflow sites A3 and A4. The cause of greater nitrate plus nitrite, total nitrogen, orthophosphorus, and total phosphorus flow-weighted concentrations at A1 is unknown.

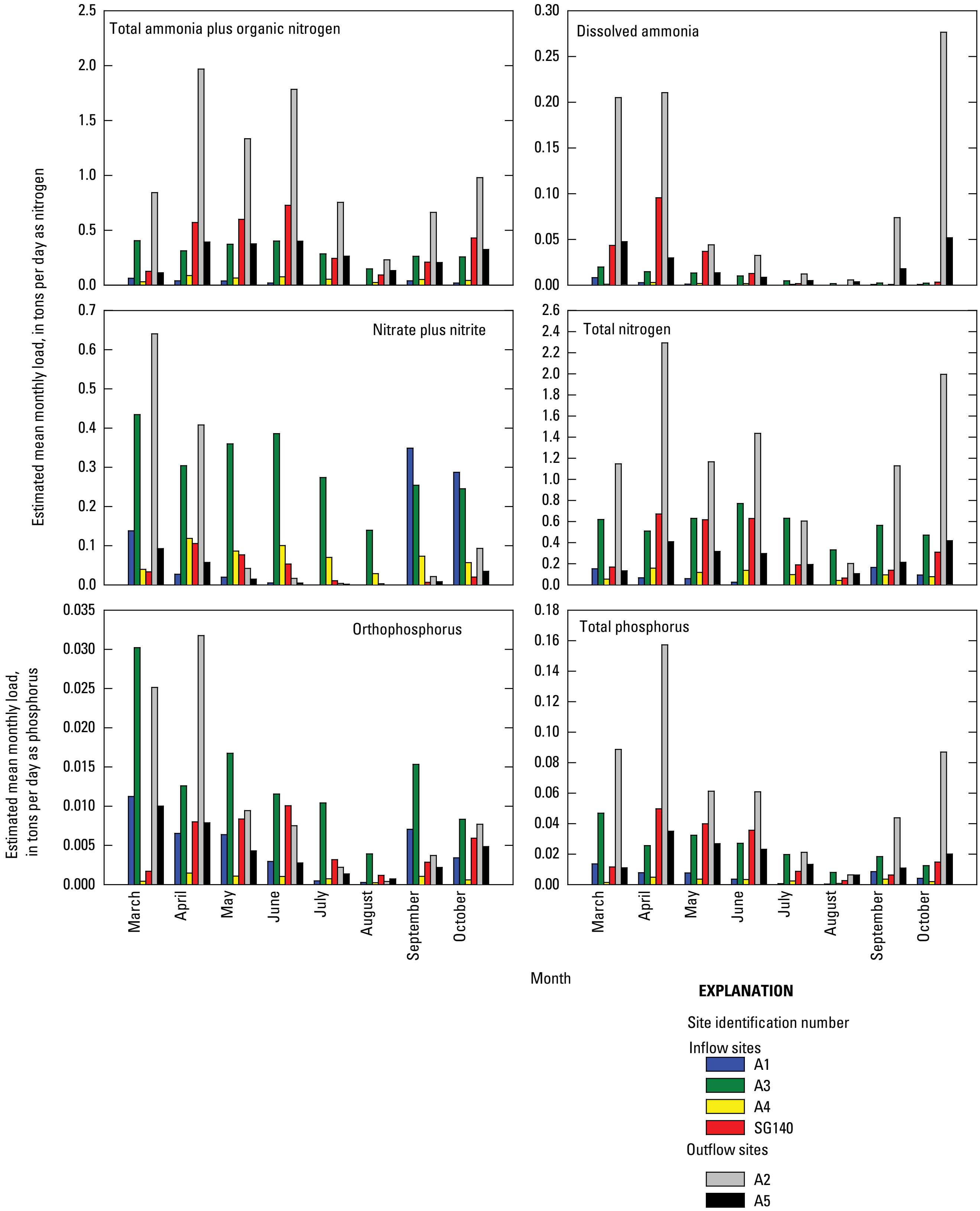

Figure 8. Mean monthly nutrient loads for inflow and outflow sites in and near Agassiz National Wildlife Refuge, northwest Minnesota, 2008 to 2010.

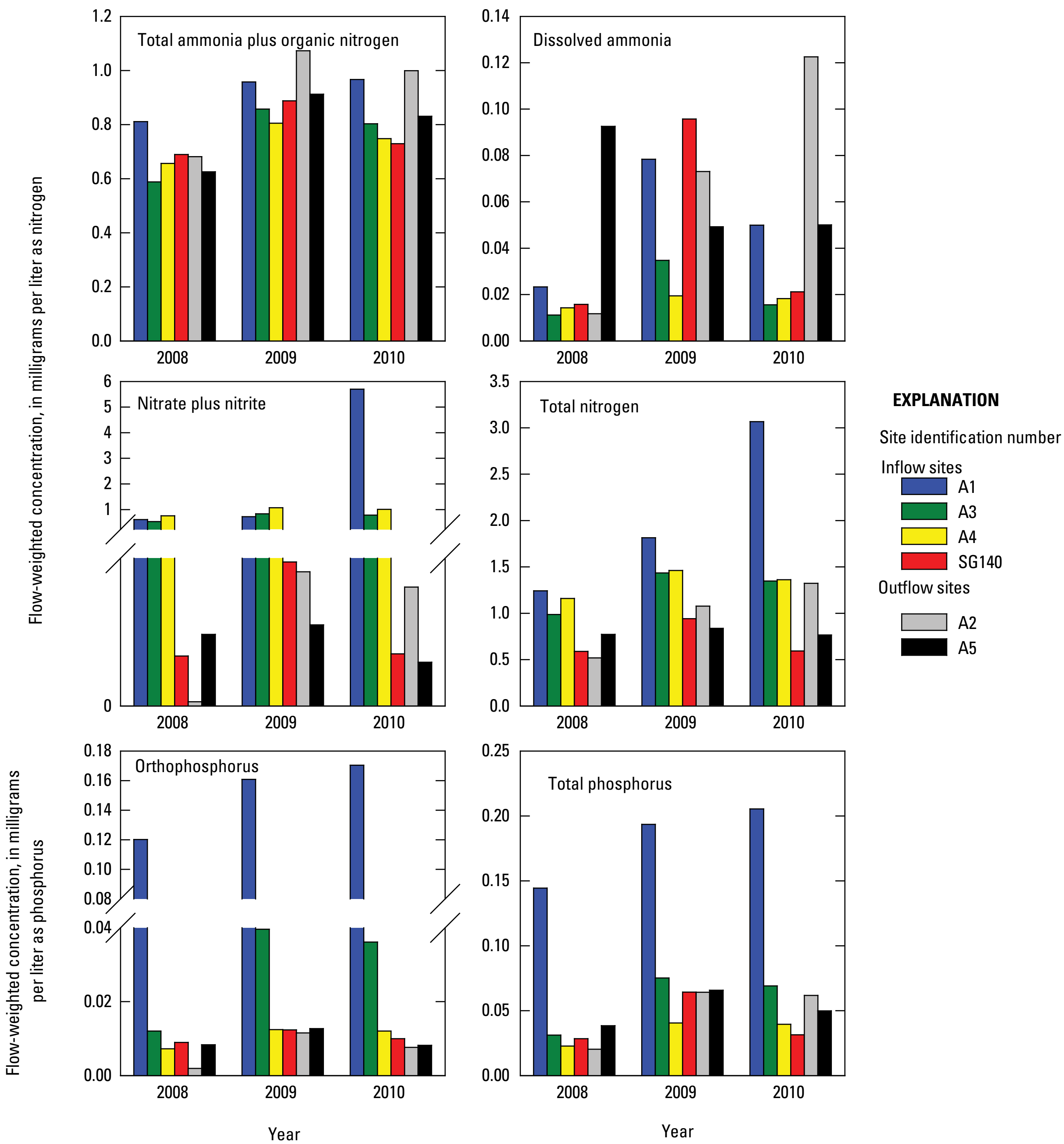

Figure 9. Annual flow-weighted nutrient concentrations for sites in and near Agassiz National Wildlife Refuge, northwest Minnesota, 2008 to 2010.

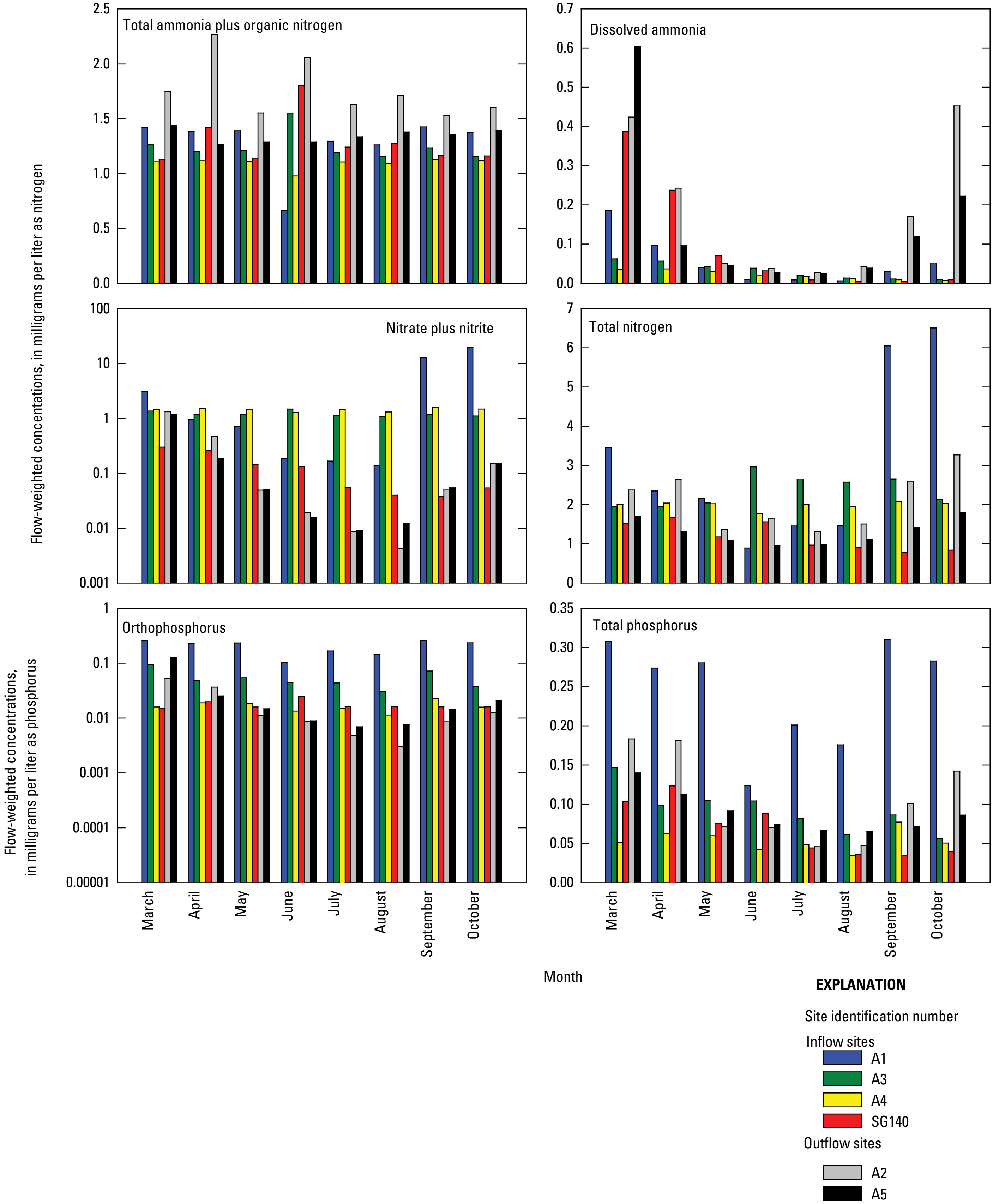

Figure 10. Mean monthly flow-weighted nutrient concentrations for sites in and near Agassiz National Wildlife Refuge, northwest Minnesota, 2008 to 2010.

Suspended Sediment

Suspended sediment is particulate matter consisting of soil and rock particles eroded from the landscape that is carried by a stream for a considerable period of time in suspension. Large concentrations of suspended sediment often are associated with storm runoff which increases streamflow, erosion, and resuspension of bed material. Activities such as row-crop agriculture, animal grazing, timber harvesting, mining, road construction and maintenance, and urbanization can cause increased suspended sediment in streams.

Concentrations

Comparison of discrete suspended-sediment concentrations for all sites indicated small differences among inflow sites, but outflow sites had significantly greater suspended-sediment concentrations than inflow sites (fig. 11). The median suspended-sediment concentration for the outflow sites ranged from 31 mg/L (A5) to 41 mg/L (A2). Among the inflow sites, suspended-sediment concentrations for inflow site A3 were significantly greater than those for A1 and A4 (fig. 11). The maximum suspended-sediment concentration of 264 mg/L for all sites was observed at outflow site A2 on October 27, 2009, as a result of an initial flush of sediment at the beginning of the scheduled drawdown of Agassiz Pool (fig. 12). During the scheduled drawdown of Agassiz Pool from October 2009 into 2010, suspended-sediment concentrations were high at outflow site A2 compared to concentrations prior to the scheduled drawdown of Agassiz Pool (fig. 12).

Loads

Annual suspended-sediment loads estimated from S-LOADEST were greatest in 2010 for all sites except inflow site SG140 and outflow site A5, with the greatest annual loads at outflow site A2, ranging from 867 tons/yr in 2008 to 29,000 tons/yr in 2010 (fig. 13, table 5). The large load at A2 in 2010 likely resulted from the combination of greater flows in 2010 (table 5) and scheduled drawdown of Agassiz Pool. Estimated annual suspended-sediment loads for inflow sites A3 and SG140 were similar, and had the largest loads of the inflow sites, ranging from 279 tons/yr to 2,580 tons/yr (fig. 13, table 5). Of the three inflow sites to Agassiz Pool (A1, A3, and SG140), A3 and SG140 accounted for at least 97 percent of the total annual sediment loads into Agassiz Pool from 2008 to 2010. Estimated annual suspended-sediment loads were smallest for A1, ranging from 9.15 tons/yr in 2008 to 142 tons/yr in 2010.

For most sites, the greatest mean monthly loads generally occurred in April, May, June, September, and October, which corresponded with months of greater streamflow (fig. 14). For inflow sites A3, A4, and SG10 and outflow site A5, the mean monthly sediment load was greatest in March or April, which are months of large runoff from snowmelt (fig. 14). For these sites, 8 (A5), 22 (A4) and 26 (A3 and SG140) percent of the annual sediment load, on average, was contributed in March or April. For inflow site A1, the greatest mean monthly sediment load occurred in September coincident with the greatest average rainfall and accounted for about 30 percent of the annual sediment load, on average (fig. 14). For outflow site A2, the greatest sediment load occurred in October, which is likely related to high concentrations of suspended sediment (fig. 12) at the start of scheduled drawdown of Agassiz Pool in October 2009, and large streamflow volume in October of 2010 (figs. 12 and 14).

Flow-Weighted Concentrations

Other than inflow site A1 and outflow site A2, the flow-weighted sediment concentrations were greatest in 2009 (fig. 15). As indicated in the previous discussion about flow-weighted concentration for nutrients, the greater concentrations in 2009 may have been related to differences in precipitation patterns between 2009 and 2010. For A2, the flow-weighted concentration in 2010 (77 mg/L) was more than double the flow-weighted concentration in 2009 (38 mg/L), which is likely related to scheduled drawdown of Agassiz Pool in 2010.

Mean monthly flow-weighted concentration of sediment followed seasonal patterns similar to mean monthly loads. For inflow sites A3 and SG140 and outflow site A5, the mean monthly flow-weighted sediment concentration was greatest in either March or April, which may be related to runoff from snowmelt (fig. 16). For inflow site A1 the greatest mean monthly flow-weighted sediment concentration occurred in September, which was likely related to rainfall runoff (fig. 16). For outflow site A2, the greatest mean monthly flow-weighted sediment concentration occurred in October, which was likely related to high concentrations of suspended sediment at the start of scheduled drawdown of Agassiz Pool in October 2009, and high streamflow in October of 2010 (figs. 12, 16). For inflow site A4, the mean monthly load was greatest in April, but the mean monthly flow-weighted concentration was greatest in September (figs. 14 and 16).

Sediment Flux in Agassiz Pool

The sediment flux from Agassiz Pool was estimated by subtracting the inflow load (A1 + A3 + SG140) from the outflow load (A2 + A5) and was evaluated for a net change of sediment. A positive sediment flux indicated a net loss of sediment from Agassiz Pool, whereas a negative sediment flux indicated a net gain of sediment to Agassiz Pool. As mentioned previously in the Description of Study Area section, some of the inflow to Agassiz Pool was not monitored. As was done for unmonitored streamflow, during the high streamflow years of 2009 and 2010, it was estimated that sediment loads from unmonitored inflow could be accounted for by doubling inputs for A1 and A3. For all years of this study

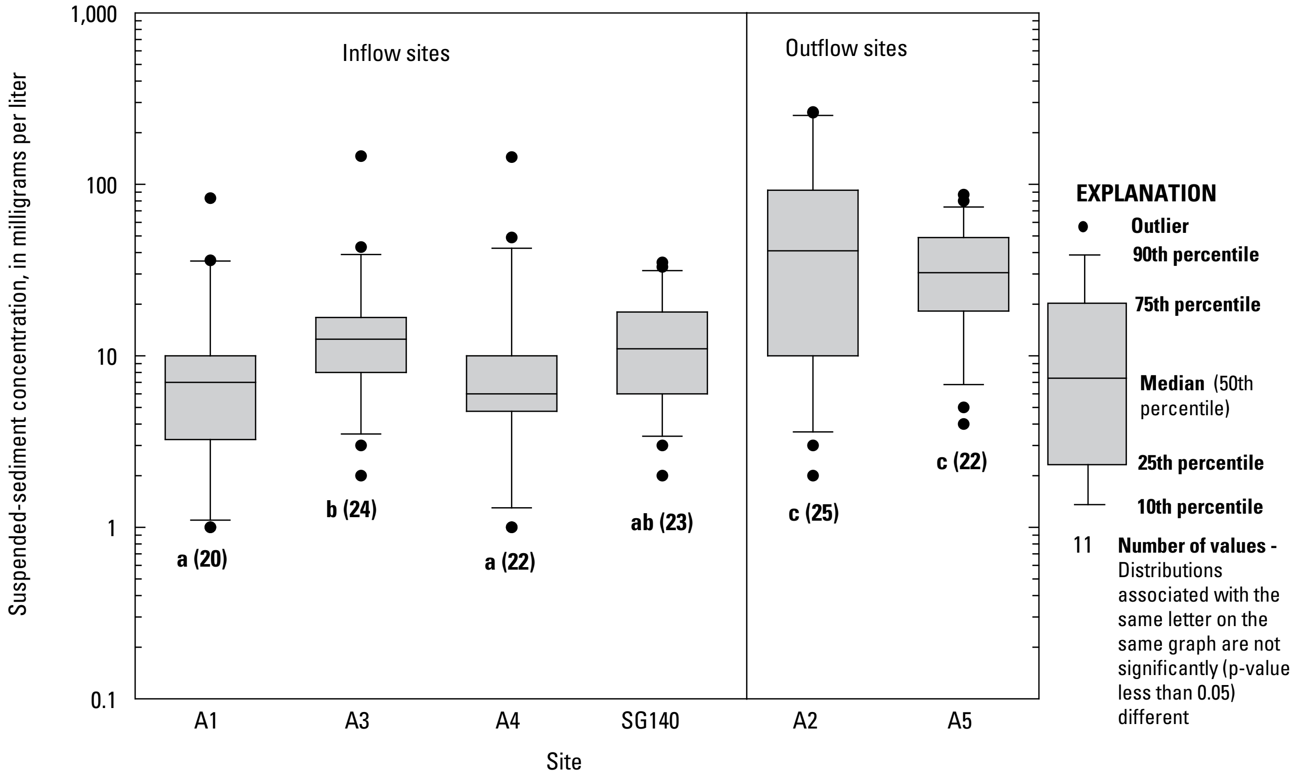

Figure 11. Distribution of suspended-sediment concentrations for inflow and outflow sites in and near Agassiz National Wildlife Refuge, northwest Minnesota, 2008 to 2010.

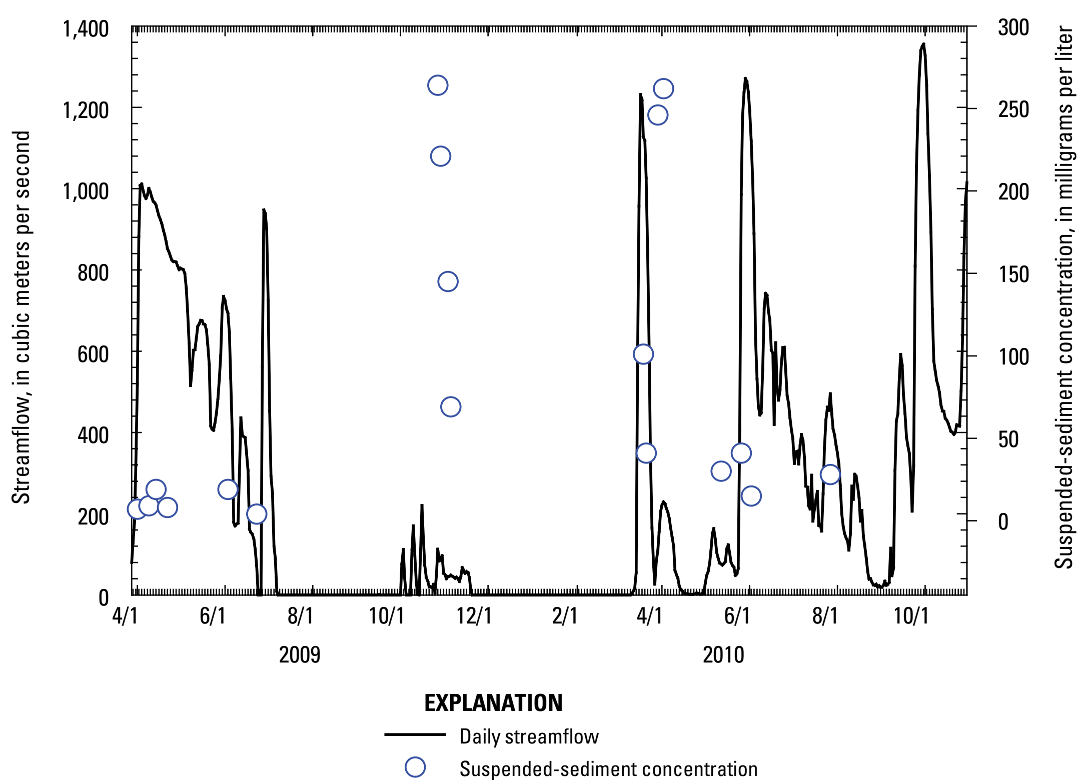

Figure 12. Daily streamflow and suspended-sediment concentrations for outflow site A2 in Agassiz National Wildlife Refuge, northwest Minnesota, 2009 to 2010.

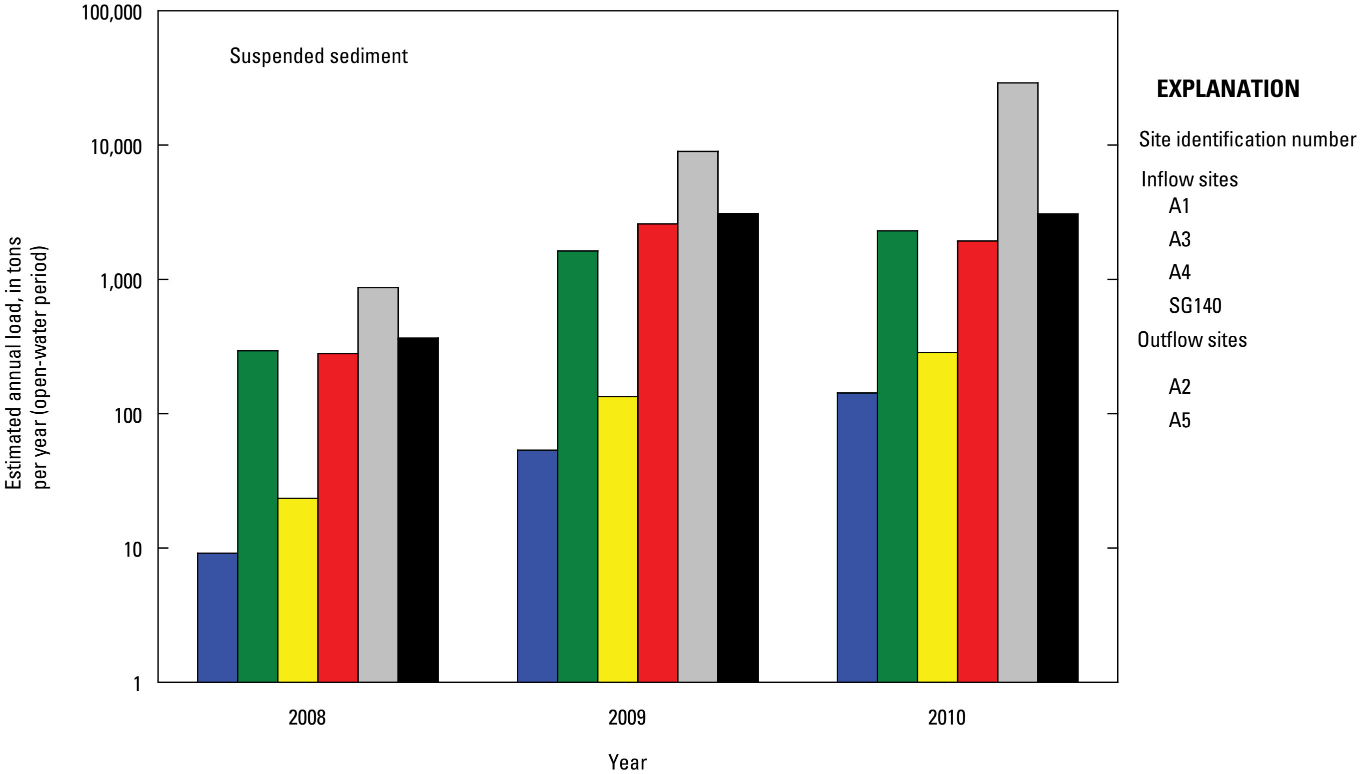

Figure 13. Estimated annual sediment loads for inflow and outflow sites in and near Agassiz National Wildlife Refuge, northwest Minnesota, 2008 to 2010.

(2008 to 2010), a net loss of sediment from Agassiz Pool occurred ranging from 650 tons/yr in 2008 to 25,300 tons/yr in 2010 (table 6). However, there were errors associated with the load estimates, with the greatest errors for A2 as indicated by an SEP of 58.0 percent and an R^2 value of 68.2 percent (table 3). Comparison of the sediment load estimates with 95-percent confidence intervals, indicated for 2008 the lower confidence interval for the load estimate at A2 (243 ton/yr) compared closely to the load estimate for A3 (217 tons/yr) (fig. 17). Using the lower confidence interval of A2 for 2008 to compute the sediment flux for Agassiz Pool, the net loss of sediment would be reduced to 27 tons/yr. Similarly, in 2009 and 2010, using the lower confidence intervals for A2, the net loss of sediment would be reduced to 1,010 tons/yr in 2009 and 23 tons/yr in 2010. Taking into account unmonitored inflow load and load estimation errors, a net loss of sediment from Agassiz Pool (more outflow load than inflow load) occurred for all 3 years.

A study completed in 2011 for Agassiz NWR by St. Croix Watershed Research Station (Science Museum of Minnesota) indicated that Agassiz Pool has been experiencing a net gain of sediment during the last 68 years (Schottler and Engstrom, 2011). Their results, which used atmospherically deposited radioisotopes ^{137}Cs and ^{210}Pb to quantify sediment flux to Agassiz Pool, indicated that 1.3 million tons of inorganic sediment have been deposited and trapped in Agassiz Pool from 1940 to 2008. Of the 1.3 million tons, it was estimated that 290,000 tons are contained within Judicial Ditch 11, the main channel through Agassiz Pool, on which A3 (upstream) and A2 (downstream) were located (fig. 1). Results also suggest that erosion from agricultural fields is likely the dominant source of sediment to Agassiz Pool (Schottler and Engstrom, unpub. data, 2012).

Although Agassiz Pool has experienced a net gain of sediment over the long term, in the short 3-year period of this study (2008 to 2010) the net loss of sediment from Agassiz Pool was likely related to a combination of several atypical water-management activities that occurred at outflow sites A2 and A5 including: the first year of operation of the WCS at A5 in 2008, which likely resulted in a flush of sediment, and resulted in some erosion of the new channel immediately downstream from the WCS; construction downstream from A2 in 2008 and 2009 to reduce long-term erosion resulted in bare dirt channels; and scheduled drawdown of Agassiz Pool in fall 2009 through 2010, which occurs only once every 10 years. Scheduled drawdown coincided with a year of large amounts of precipitation, and the resultant runoff combined with large amounts of sediment available in Agassiz Pool

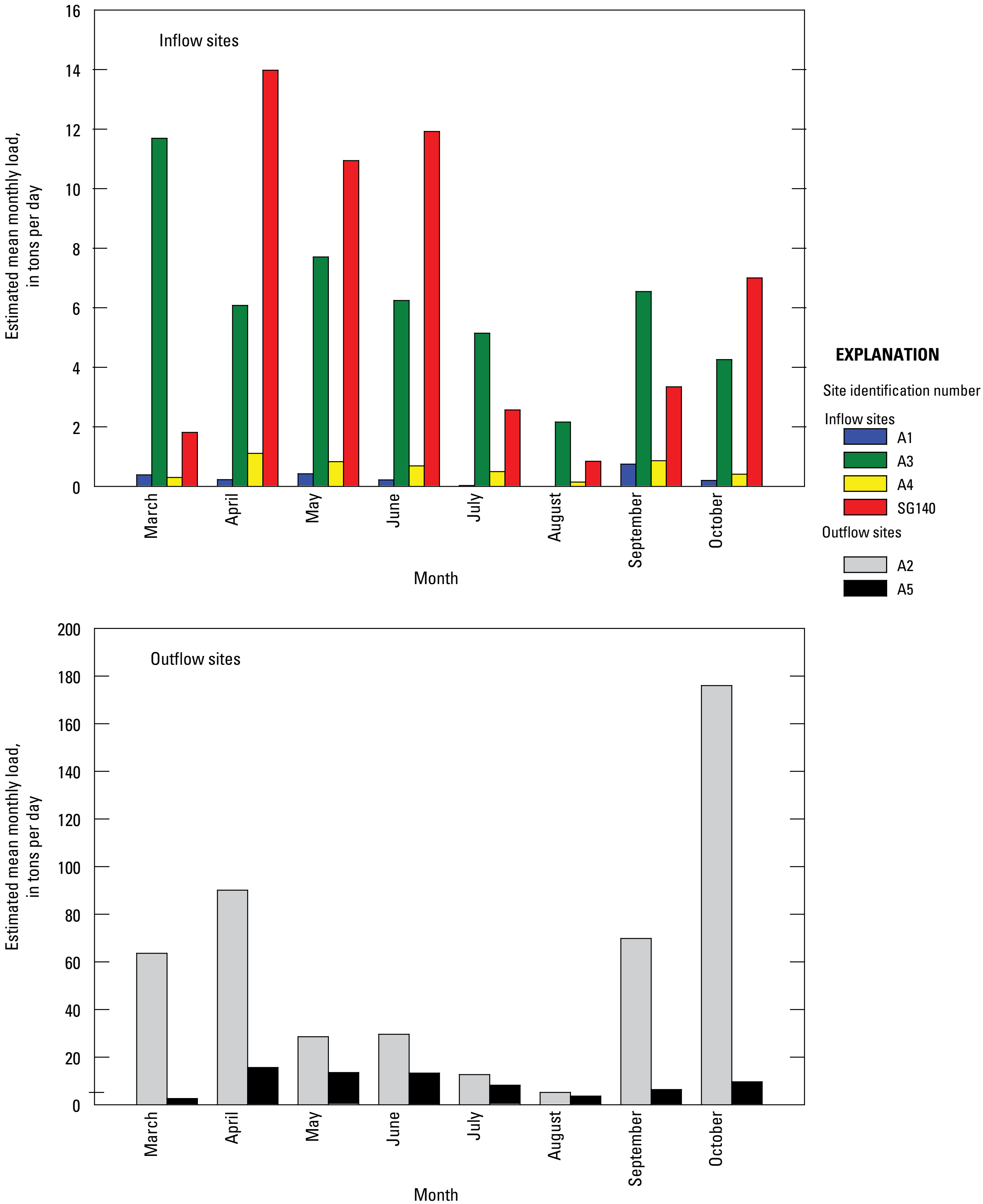

Figure 14. Estimated mean monthly sediment loads for inflow and outflow sites in and near Agassiz National Wildlife Refuge, northwest Minnesota, 2008 to 2010.

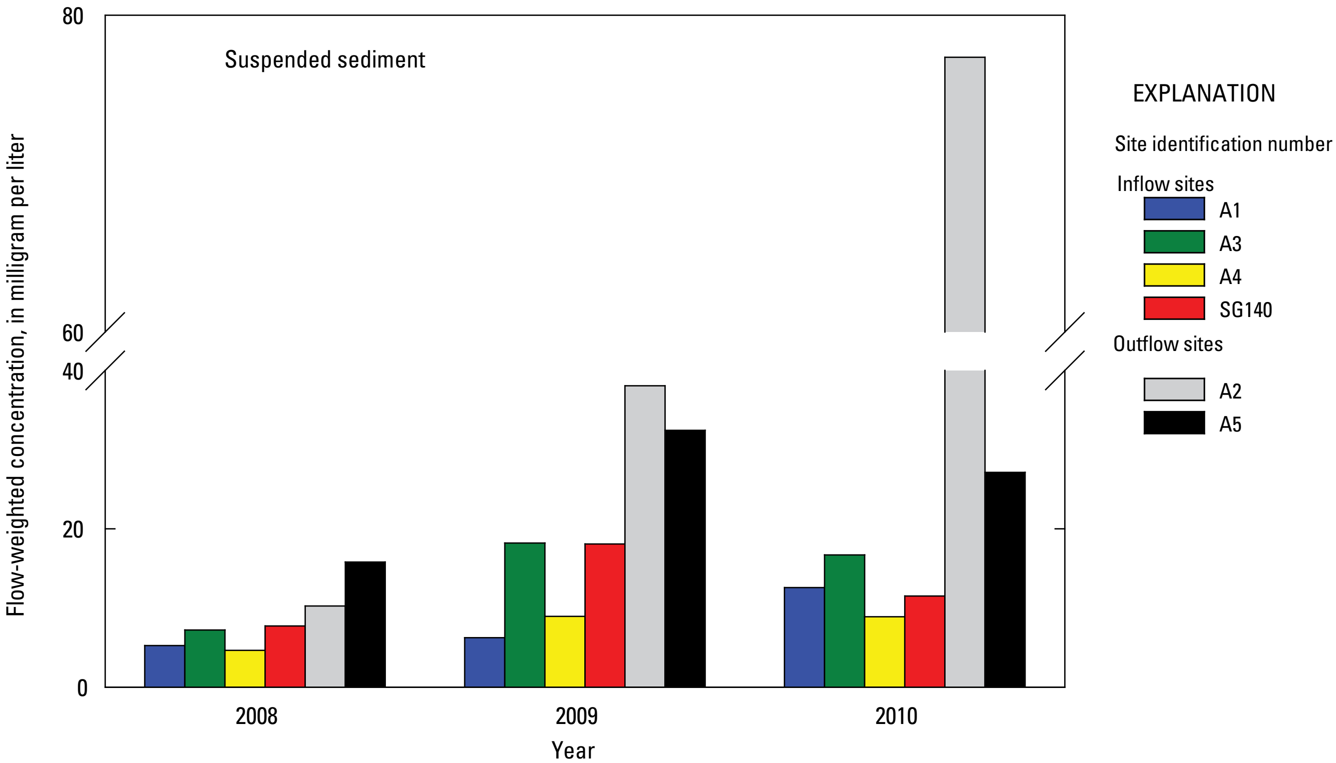

Figure 15. Annual flow-weighted sediment concentrations for sites in and near Agassiz National Wildlife Refuge, northwest Minnesota, 2008 to 2010.

likely contributed to the large sediment loads from A2 in 2010. During scheduled drawdown (October 2009 through October 2010), the average suspended-sediment concentration for discrete samples was 122 mg/L compared with an average suspended-sediment concentration of 23 mg/L for samples collected when no drawdown was occurring (April 2008 through September 2009). Consistent with discrete sample concentrations and the much larger load estimated for A2 in 2010, the flow-weighted concentration at A2 in 2010 was nearly double that in 2009 (fig. 15). In 2008 and 2009, A2 contributed approximately 70 percent of the annual outflow load, but in 2010, during scheduled drawdown, 90 percent of the outflow load was contributed by A2. During scheduled drawdown, some of the stored sediment in Agassiz Pool may have been released, however, where the sediment came from in Agassiz Pool and what type of sediment was released cannot be determined from this study. Schottler and Engstrom (2011), indicated that large amounts of inorganic sediment have been deposited in Agassiz Pool, but it is possible that organic sediment may have been flushed out as well.

Temperature, Specific Conductance, Dissolved Oxygen, pH, and Turbidity

Because of many equipment problems in 2008 and 2009, continuous water-quality monitor data for all sites other than A3 had large and numerous gaps in it. In 2010, fewer equipment problems were encountered so the data were more complete for all sites. Because the data were 50 to 70 percent complete for A3 for all 3 years, annual statistics were computed (table 7) and hourly data are presented (fig. 18). Because data were between 50 and 70 percent complete for all sites in 2010, annual statistics were computed (table 8) and the hourly data are presented (figs. 19 and 20).

For A3, the warmest mean temperature of 16.3°C occurred in 2010, but the warmest maximum temperature of 27.6°C occurred in 2009 (table 7). At inflow site A3, hourly water temperatures exhibited a similar pattern for all years, with temperatures starting out near 0°C in the spring, rising to around 25°C in late July and falling again through September and October (fig. 18). Large diurnal fluctuations were observed in August 2008 at A3, due to lower streamflow during that period (fig. 18, fig. 3). Among all the sites in 2010, the warmest mean temperature of 18.1°C and the warmest maximum temperature of 32.8°C occurred at A2 (table 8). Outflow site A2 also exhibited the largest diurnal fluctuations of all the sites (fig. 20).

Electrical conductivity is a measure of the capacity of water to conduct an electrical current and is a function of the types and quantities of dissolved substances in water (Hem, 1989). As concentrations of dissolved ions increase, conductivity of the water generally increases. Specific conductance

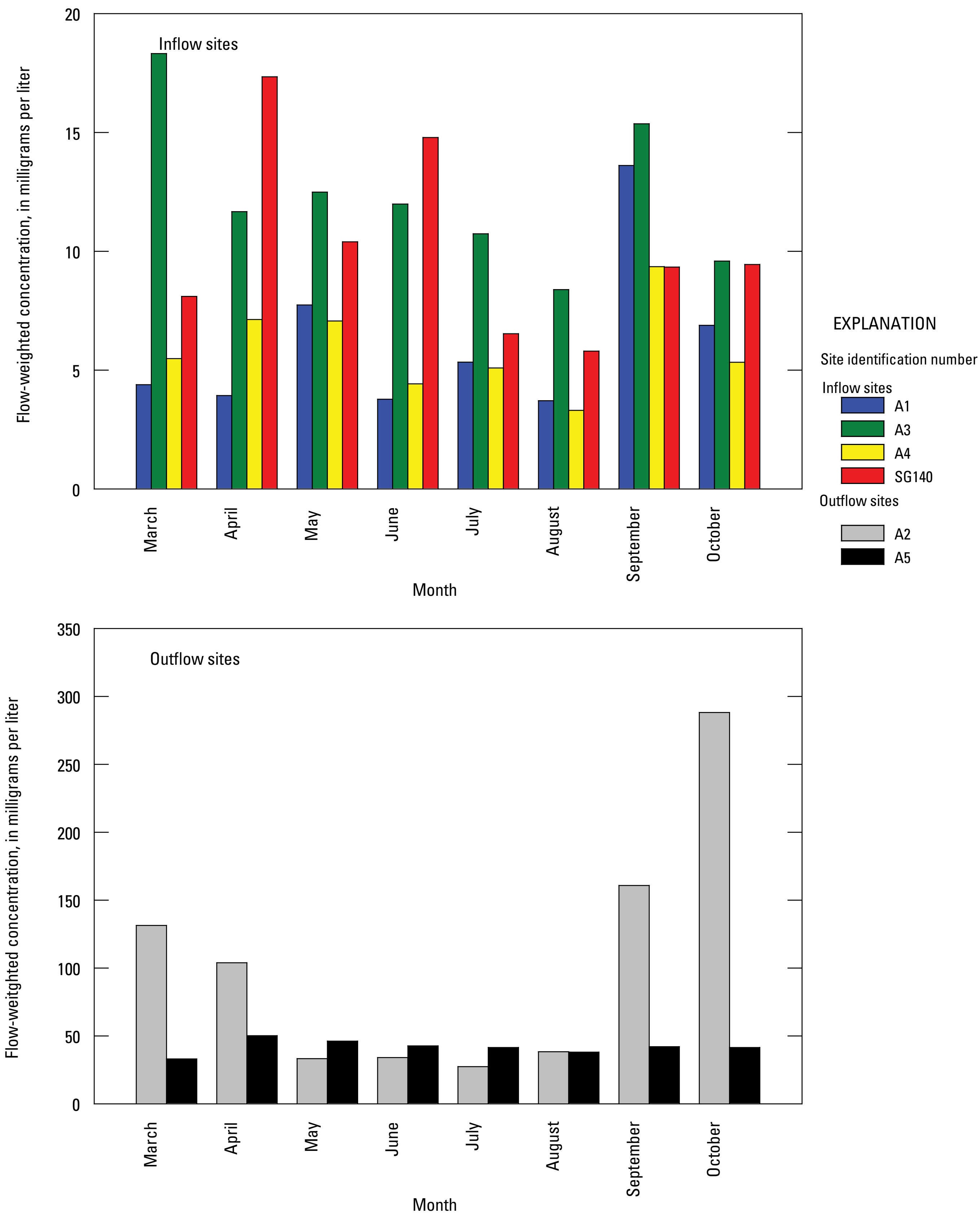

Figure 16. Mean monthly flow-weighted sediment concentrations for sites in and near Agassiz National Wildlife Refuge, northwest Minnesota, 2008 to 2010.

Table 6. Sediment flux from Agassiz Pool, Agassiz National Wildlife Refuge, northwest Minnesota, 2008 to 2010.

[yr, year]

	Year (open-water period)	Load (tons/yr)
Inflow load (A1 + A3 + SG140)	2008	581
((A1 x 2) + (A3 x 2) + SG140)[1]	2009	5,880
((A1 x 2) + (A3 x 2) + SG140)	2010	6,800
Outflow load (A2 + A5)	2008	1,232
	2009	12,000
	2010	32,100
Sediment flux (Outflow – Inflow)	**2008**	**650**
	2009	**6,120**
	2010	**25,300**

[1]Inflow load for A1 and A3 were doubled in 2009 and 2010 to account for unmonitored inflow to Agassiz Pool.

within the stream is affected by runoff from snow, rain, and groundwater. Because of low concentrations of dissolved ions in snow and rain, following snowmelt or rainfall, generally the specific conductance within the stream will decrease. If a stream is affected by groundwater, the specific conductance in the stream tends to be higher because groundwater has higher concentrations of dissolved ions. Specific conductance is the conductivity expressed in units of microsiemens per centimeter (µS/cm) at 25°C. For A3, the lowest mean specific conductance of 404 µS/cm occurred in 2009 (table 7). In 2009, specific conductance at A3 was lower in the spring than other years, which is related to a large snowmelt (fig. 18). In 2010, among all the sites, the highest mean specific conductance of 739 µS/cm occurred at A1 and the lowest mean specific conductance of 324 µS/cm occurred at SG140 (table 8). The greatest variability in specific conductance occurred at A1, ranging from 172 to 1,160 µS/cm and the least amount of variability occurred at SG140, ranging from 210 to 434 µS/cm (table 8). There was also large variability in specific conductance at A4. The higher variability in specific conductance at A1 and A4 is related to variability in streamflow. For both sites, there were long periods of little to no flow, which resulted in dissolved ions becoming more concentrated in the water (increase in specific conductance) followed by short periods of high flow due to rainfall, which resulted in dilution (decrease in specific conductance). For SG140, the smaller range in specific conductance values is caused by dilution from Thief Lake. There is also little variability in specific conductance at A5 because it is located directly downstream from SG140. The effect of rainfall on specific conductance was observed during a storm-runoff event on May 24, 2010, when a total of 3.38 inches of rain fell (about 2.44 inches of the 3.38 inches fell within a 3-hour period). Specific conductance decreased substantially at inflow sites A1, A3, A4, and A2 (figs. 19 and 20). On the same date, the specific conductance did not decrease noticeably at SG140 and A5, likely because the streamflow volume at SG140 was about 360 ft^3/s and about 85 percent of the streamflow was from Thief Lake.

Dissolved oxygen (DO) is important in chemical reactions in water and in the life cycles of aquatic organisms. Sources of DO in surface waters are primarily atmospheric reaeration and photosynthetic activity of aquatic plants. DO is consumed by the respiration of aquatic plants, ammonia nitrification, and the decomposition of organic matter in a stream. The solubility of DO is affected by water temperature and atmospheric pressure. DO solubility increases with colder water, whereas warmer water holds lesser amounts of DO, and solubility increases with increasing atmospheric pressure and decreases with decreasing atmospheric pressure (Galloway, 2008). The State of Minnesota has established a minimum water-quality standard for DO of 5 mg/L (Minnesota Office of the Revisor of Statutes, 2012). At A3, the lowest mean DO was 8.5 mg/L in 2010 (table 7). In each of the years, there were instances when hourly DO was less than the 5 mg/L standard, but for all years less than 4 percent of the hourly values were below 5 mg/L (fig. 18). In 2010, inflow site A1 had the lowest mean DO of 6.0 mg/L and A2 had the highest mean DO of 9.2 mg/L (table 8). Other than for A5, in 2010 all sites had occurrences of hourly DO less than 5 mg/L (figs. 19 and 20, table 8). For A1 and A4, 28 and 24 percent of the hourly values were less than 5 mg/L, respectively. For A2, 8 percent of the hourly values were less than 5 mg/L, compared with 0.1 percent for SG140. A2 had the greatest variability in DO with concentrations ranging from 2.2 to 17.8 mg/L (fig. 20, table 8). Diurnal fluctuations were large at A2, most likely related to large diurnal fluctuations in temperature and algal activity (fig. 20). In response to the storm-runoff event on May 24, 2010, DO decreased at A1, A3, and A4 (fig. 19). This decrease in DO was likely related to a flush of organic matter into the stream and resultant consumption of DO.

The pH of an aqueous solution is controlled by interrelated chemical reactions that produce or consume hydrogen ions (Hem, 1989). Many reactions that occur in natural water among solutes (solid or gaseous) or other liquid species involve hydrogen ions, and, therefore, affect the pH. For example, the reaction of carbon dioxide with water is one of the most important in controlling the pH in natural water systems (Hem, 1989). When algae are respiring, DO is consumed and carbon dioxide is generated resulting in a decrease in pH and DO. Based on the Minnesota water-quality standard for pH, pH should be between 6.5 and 8.5 standard units (Minnesota Office of the Revisor of Statutes, 2012). For A3, the median pH for all 3 years varied little, ranging from 7.8 to 7.9 standard units (table 7). For all 3 years, the pH was greater than 6.5 standard units, but in 2008 and 2009 pH exceeded 8.5 standard units for no more than 5 percent of the hourly values (fig. 18). In 2010, the lowest median pH of 7.5 was recorded at A1 and the highest median pH of 8.5 standard

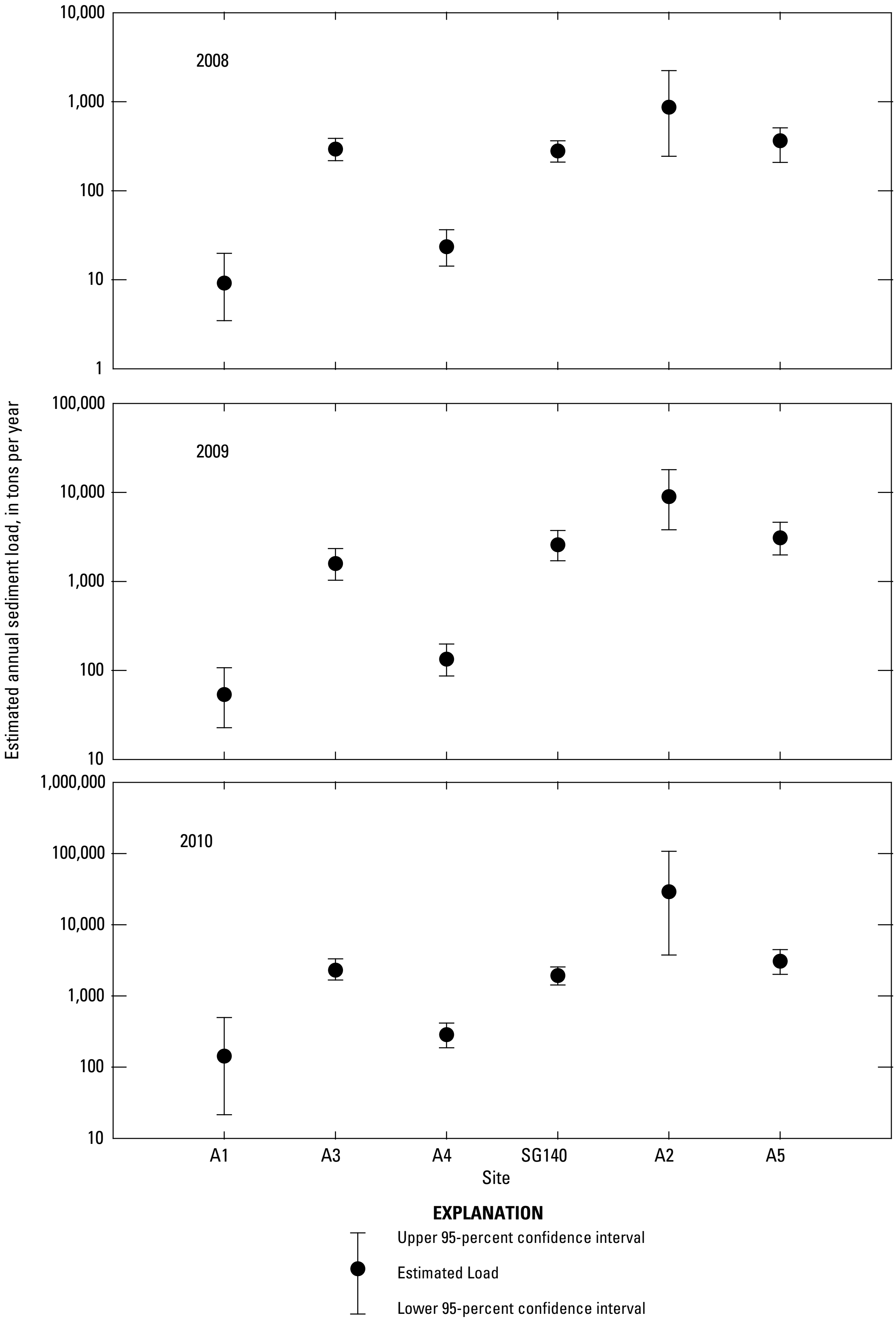

Figure 17. Confidence intervals for suspended-sediment loads for six sites in and near Agassiz National Wildlife Refuge, northwest Minnesota, from 2008 to 2010.

Table 7. Annual water temperature, specific conductance, dissolved oxygen, pH, and turbidity statistics for site A3 in Agassiz National Wildlife Refuge, northwest Minnesota, 2010.

[<, less than]

Year	Statistic	Temperature (degrees Celcius)	Specific conductance (microsiemens per centimeter at 25 degrees Celcius)	Dissolved oxygen (milligrams per liter)	pH (standard units)	Turbidity (formazin nephelometic turbidity units)
2008	Mean[1]	15.5	446	9.7	7.9	16
	Maximum[2]	27.7	775	19.1	8.9	240
	Minimum	-0.1	224	4.7	6.5	2
	Number of daily values	124	120	124	90	145
2009	Mean	14.9	404	10.1	7.8	11
	Maximum	27.6	756	17.6	9.0	160
	Minimum	1.4	218	3.2	7.3	<1
	Number of daily values	119	115	119	109	107
2010	Mean	16.3	454	8.5	7.8	16
	Maximum	27.0	831	13.7	8.5	690
	Minimum	0.0	157	3.1	7.0	<1
	Number of daily values	161	161	161	160	162

[1]Mean values are computed from daily values.

[2]Maximum and minimum values are from instantaneous values.

units was recorded at A2 (table 8). For all sites in 2010, pH was greater than 6.5 standard units, but pH at SG140, A2, and A5 exceeded 8.5 standard units in 14, 29, and 5 percent of the hourly values, respectively (figs. 19 and 20). Similar to temperature and DO data, diurnal fluctuations of pH in 2010 were more pronounced at A2 than any other site (figs. 19 and 20). Also, similar to DO, a decrease in pH was observed at A1, A3, and A4 two days after the storm-runoff event on May 24, 2010 (figs. 19 and 20).

Turbidity is a measure of the optical properties of a sample that cause light rays to be scattered and absorbed (Gray and Glysson, 2003). Turbidity of water is caused by the presence of suspended and dissolved inorganic matter such as clay and silt; suspended and dissolved organic matter such as plankton, microscopic organisms, small terrestrial organic material, and organic acids; and water color. Generally turbidity increases with storm runoff, because of sediment and other materials being washed off the landscape into the stream. For turbidity, a minimum water-quality standard of 25 nephelometric turbidity units (NTU) has been established by the State of Minnesota (Minnesota Office of the Revisor of Statutes, 2012). Turbidity for this study was measured in formazin nephelometric turbidity units (FNU) which can be compared directly to the 25 NTU water-quality standard. For A3, the mean turbidity for all years was less than 25 NTU (table 7); however, turbidity frequently spiked from storm runoff, resulting in between 5 percent (2008) and 9 percent of the hourly values exceeding 25 NTU (fig. 18). In 2010, mean turbidity ranged from 2 FNU at site A1 to 27 FNU at site A5 (table 8). Among all sites in 2010, the greatest variability in turbidity occurred at inflow site A1. Inflow site A1 is typically a clear ditch, as indicated by the mean turbidity of 2 FNU, but can become turbid from storm runoff as indicated by the maximum turbidity of 1,110 FNU (table 8). For all sites, spikes in turbidity occurred from storm runoff, with as little as 2 percent of the hourly values exceeding 25 NTU at A4 and at most 38 percent of the hourly values exceeding 25 NTU at site A5. For A2, 35 percent of the hourly values exceed 25 NTU, but in 2010, sensor fouling caused by storm runoff was a chronic problem, which resulted in only 24 percent of the turbidity data being available. During the storm-runoff event on May 24, 2010, other than at A1, the turbidity sensors at all other sites fouled, resulting in data that had to be deleted. If the sensors had not fouled, it is possible that other sites would have had higher turbidity readings than A1 during the May 24 storm-runoff event.

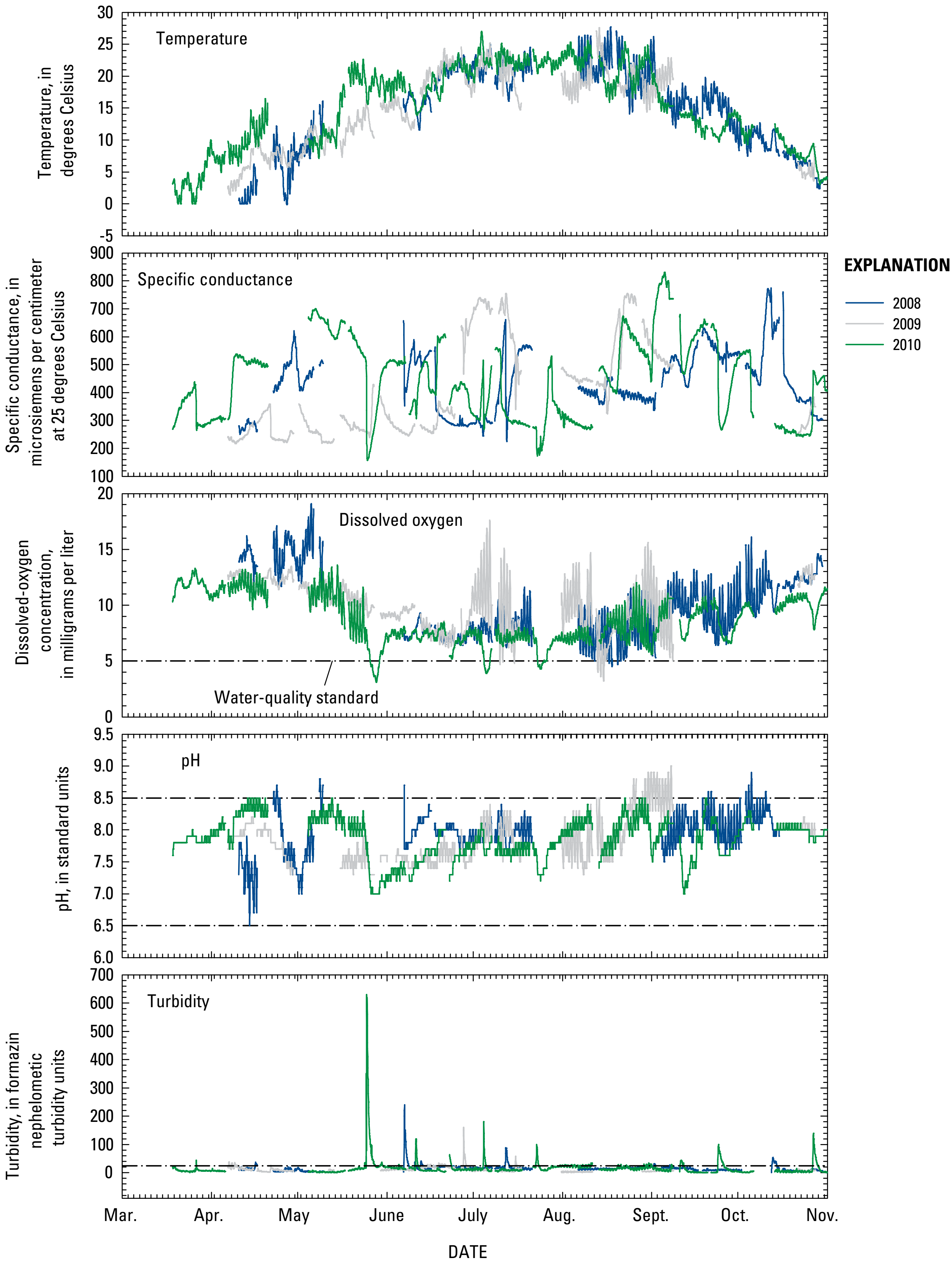

Figure 18. Hourly water temperature, specific conductance, dissolved oxygen, pH, and turbidity for site A3 in Agassiz National Wildlife Refuge, northwest Minnesota, 2008 to 2010.

Table 8. Annual water temperature, specific conductance, dissolved oxygen, pH, and turbidity statistics for six sites in Agassiz National Wildlife Refuge, northwest Minnesota, 2010.

[<, less than]

Statistic	Site					
	A1	A2	A3	A4	A5	SG140
Temperature, degrees Celsius						
Mean[1]	15.2	18.1	16.3	16.3	16.4	16.3
Maximum[2]	26.3	32.8	27.0	27.8	27.4	27.5
Minimum	0.0	-0.1	0.0	0.0	0.0	0.0
Number of daily values	118	140	161	158	162	167
Specific conductance, in microsiemens per centimeter at 25 degrees Celsius						
Mean	739	430	454	596	364	324
Maximum	1,160	763	831	1,120	524	434
Minimum	172	230	157	209	253	210
Number of daily values	116	140	161	158	161	112
Dissolved oxygen, in milligrams per liter						
Mean	6.0	9.2	8.5	6.8	8.5	7.9
Maximum	14.4	17.8	13.7	16.0	12.5	11.5
Minimum	0.0	2.2	3.1	0.7	5.3	4.2
Number of daily values	113	113	161	141	162	121
pH, in standard units						
Median	7.5	8.5	7.8	7.8	8.1	8.2
Maximum	8.4	9.6	8.5	8.6	8.9	9.0
Minimum	7.1	7.6	7.0	7.2	7.2	7.6
Number of daily values	117	139	160	154	160	148
Turbidity, in formazin nephelometric units						
Mean	2	10	16	6	27	12
Maximum	1,110	620	690	550	230	300
Minimum	<1	<1	<1	<1	6	<1
Number of daily values	118	58	162	134	176	145

[1]Mean values are computed from daily values.

[2]Maximum and minimum values are from instantaneous values.

Comparison of Suspended-Sediment Load Estimation Methods

As another method to estimate suspended-sediment concentrations and ultimately loads, regression equations were developed and compared with estimates from S-LOADEST. Turbidity was determined to be significantly correlated to suspended-sediment concentration for all sites (table 9). For sites A3 and A5, streamflow also was determined to be significantly correlated to suspended sediment and for all sites, both the independent and dependent variable(s) were log transformed (table 9). R^2 values for the regression equations were less than those for the S-LOADEST models (tables 3 and 9). For the regression equations, R^2 values were greatest for A4 and least for A2, which was consistent with the S-LOADEST models. Factors contributing to lesser R^2 values for the regression equations compared to the S-LOADEST models likely included the use of concentration as the dependent variable as opposed to load as the dependent variable; fewer paired data available to develop the regressions because of missing values of turbidity from the continuous water-quality monitors; and smaller range of values for the independent variables used for the regression equations.

Measured instantaneous suspended-sediment concentrations were plotted against estimated concentrations for the same day from both the regression method and the S-LOADEST method to visually assess the models (fig. 21). Points that plotted above the line of equal value indicate that the estimated concentrations were greater than measured concentrations (overestimated); points that plotted below

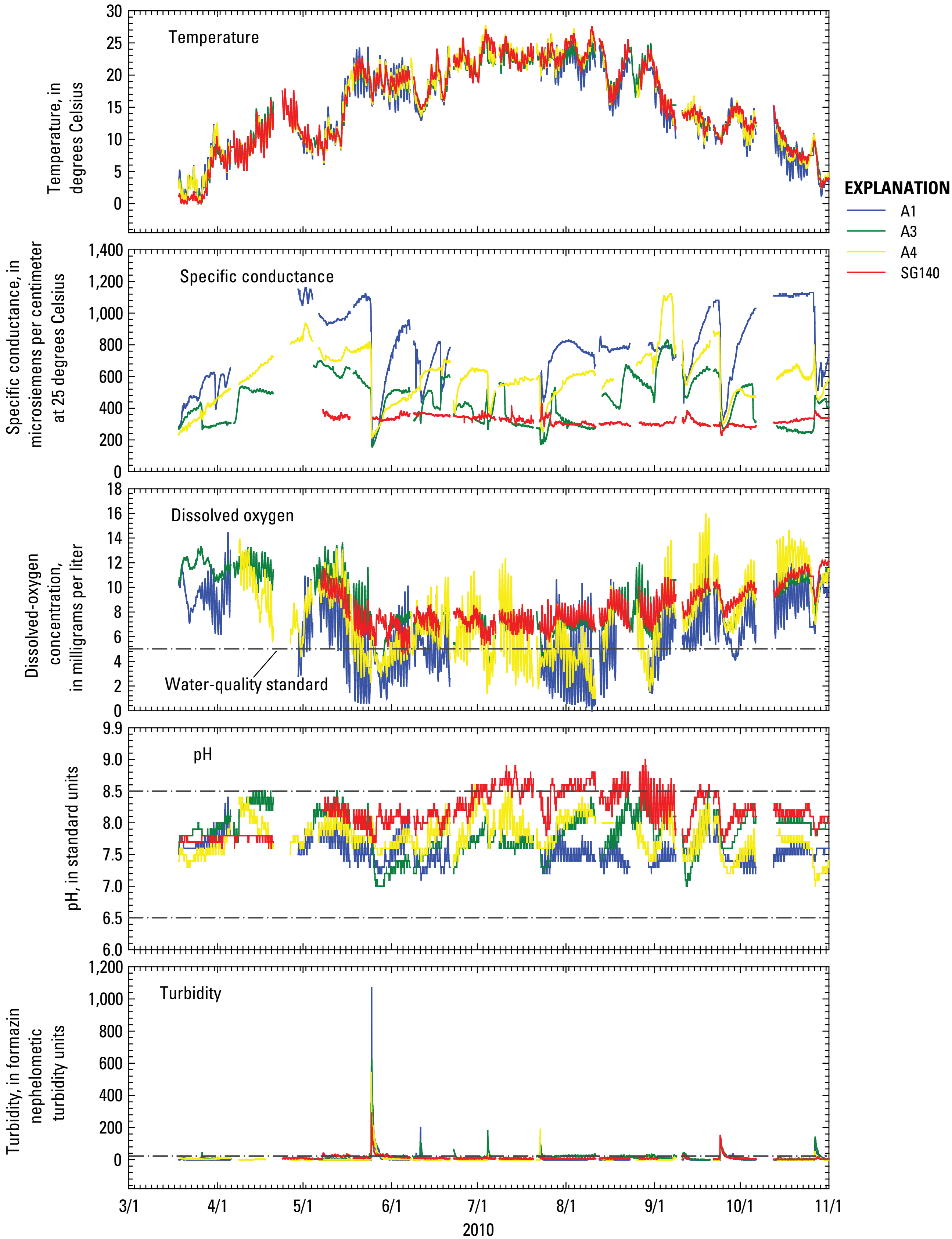

Figure 19. Hourly water temperature, specific conductance, dissolved oxygen, pH, and turbidity for inflow sites A1, A3, A4, and SG140 in and near Agassiz National Wildlife Refuge, northwest Minnesota, from March 2010 through October 2010.

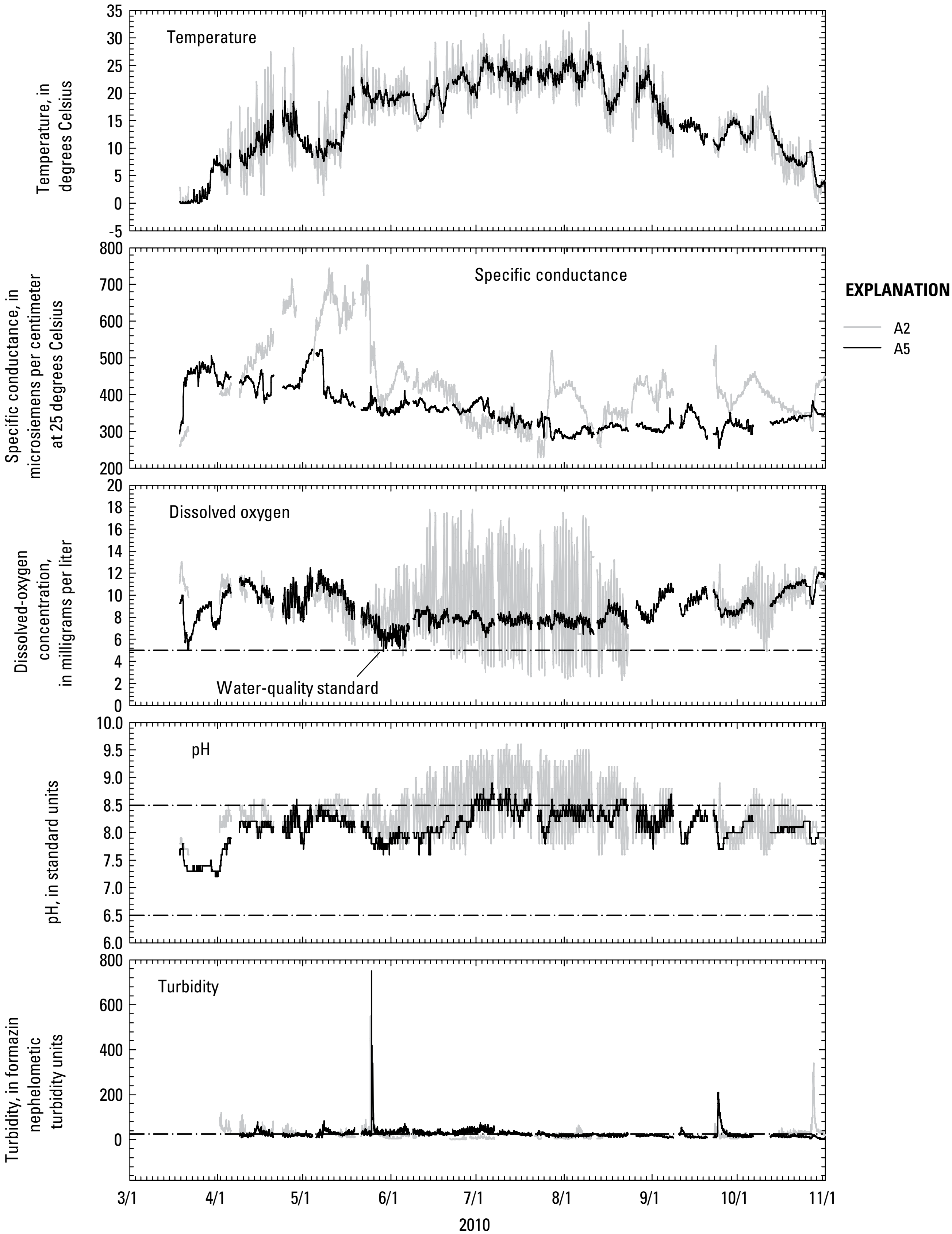

Figure 20. Hourly water temperature, specific conductance, dissolved oxygen, pH, and turbidity for outflow sites, A2 and A5 in Agassiz National Wildlife Refuge, northwest Minnesota, from March 2010 through October 2010.

Table 9. Regression equations for estimates of suspended-sediment concentrations for sites in and near Agassiz National Wildlife Refuges in northwest, Minnesota, 2008 to 2010.

[*n*, number of samples used to develop regression equation; p-value, probability value; R^2, coefficient of determination; RPD, median relative percent difference; BCF, bias correction factor; *SSC*, suspended-sediment concentration; *turb*, turbidity in formazin nephelometric units; *Q*, streamflow in cubic feet per second; <, less than]

Site identification number	*n*	Equation	Range of independent variables	p-value	R^2	Median RPD	BCF
A1	13	$\log_{10}SSC$=0.503 $\log_{10}(turb)$ +0.506	*turb*: 0–130	<0.001	0.646	60.5	1.24
A2	14	$\log_{10}SSC$=0.641 $\log_{10}(turb)$ +0.635	*turb*: 3–260	0.001	0.568	49	1.55
A3	19	$\log_{10}SSC$=(0.460 $\log_{10}(turb)$ + (0.419 x $\log_{10}Q$) -0.237	*turb*: 1.1–210 *Q*: 50.0–144	<0.001 0.01	0.824	27.4	1.05
A4	10	$\log_{10}SSC$=0.713 $\log_{10}(turb)$ +0.298	*turb*: 0–190	<0.001	0.837	37.5	1.09
A5	13	$\log_{10}SSC$=(0.72 $\log_{10}(turb)$ + (0.346 x $\log_{10}Q$) -0.290	*turb*: 7.6–130 *Q*: 11.7–329	0.005 0.036	0.801	29.2	1.08
SG140	16	$\log_{10}SSC$=0.760 $\log_{10}(turb)$ +0.218	*turb*: 2.8–33	<0.001	0.630	20.3	1.09

the line of equal value indicate that estimated concentrations were less than the measured concentrations (underestimated). To compare between methods, RPD was computed for both methods (fig. 21). For A1, points generally plotted above and below the line for both methods indicated no systematic overestimation or underestimation, but the scatter was larger at the lower end of the line indicating a poorer fit at lower concentrations. For A1, the RPD for S-LOADEST was considerably less than the RPD for regression indicating that S-LOADEST had a better ability to estimate suspended-sediment concentrations (fig. 21). For A2, the scatter was large for both methods and the plotted points indicated some overestimation. For A3 and SG140, the scatter was small and both methods had comparable RPDs (fig. 21). For A4 and A5, for both methods the scatter was small, but the RPDs for the regression method were considerably better.

Because A3 and A5 had the longest period of continuous record for turbidity in 2010, estimated suspended-sediment concentrations from the regressions developed for sites A3 and A5 were used to compute loads, and the loads were compared with loads computed from S-LOADEST (fig. 22). Overall, the two methods compared well with measured values, with periods of overestimation and underestimation. On average, the monthly load estimates from the two methods differed from one another by about 19 percent for both sites. For the total 2010 load, the difference between load estimates using the two different methods was about 3 percent for A3 and 19 percent for A5. For A3 in September, a large decrease in the estimated load from the regression method was caused by turbidity values near zero. The S-LOADEST method did not capture this abrupt decrease in load. Generally, the estimated loads from S-LOADEST method were lower than measured peak loads, but possibly slightly overestimated during periods of lower streamflow.

Quality Assurance and Quality Control

Blank and replicate water-quality samples were collected during the data collection period to estimate the variability in the laboratory analysis and reproducibility in the collection of the samples. Six blank samples (five field blanks and one equipment blank) were analyzed for nutrients. For all blank samples and all constituents, values were less than the laboratory reporting level. Fourteen replicate samples were collected and analyzed for nutrients and suspended sediment (table 10). For one of the replicate samples, the dissolved nutrient sample bottle arrived damaged at the lab, so the results were not used. The analytical variability of replicate nutrient and sediment samples was minimal with differences ranging from 0.9 percent to 6.9 percent (table 10). Results from QA/QC data indicate that cleaning procedures were adequate in preventing cross-contamination of samples and that the laboratory results were reproducible.

Water-Quality and Streamflow Monitoring Program Design

Water-quality monitoring programs are developed to meet many different objectives and may include; documenting water-quality conditions, assessing variability, evaluating effects of management strategies and changing land use and climate, identifying threats or impairments, improving understanding of processes, and supporting regulatory requirements. A water-quality program designed to monitor changes in water quality with time (trend analysis) and estimating constituent loads addresses many if not all of the aforementioned

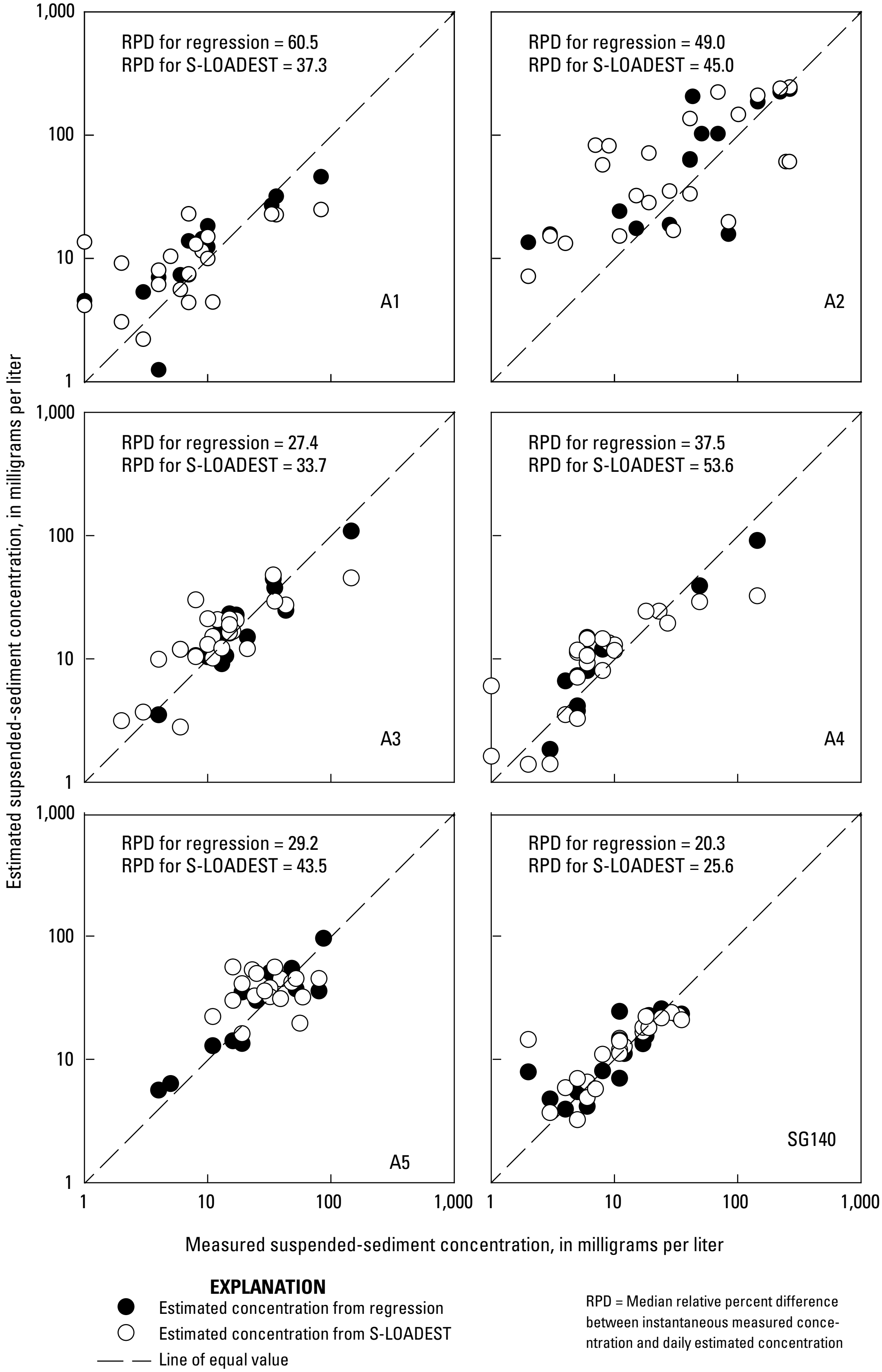

Figure 21. Comparison of measured and estimated suspended-sediment concentrations for two different estimation methods for sites in Agassiz National Wildlife Refuge northwest, Minnesota, from March 2008 to October 2010.

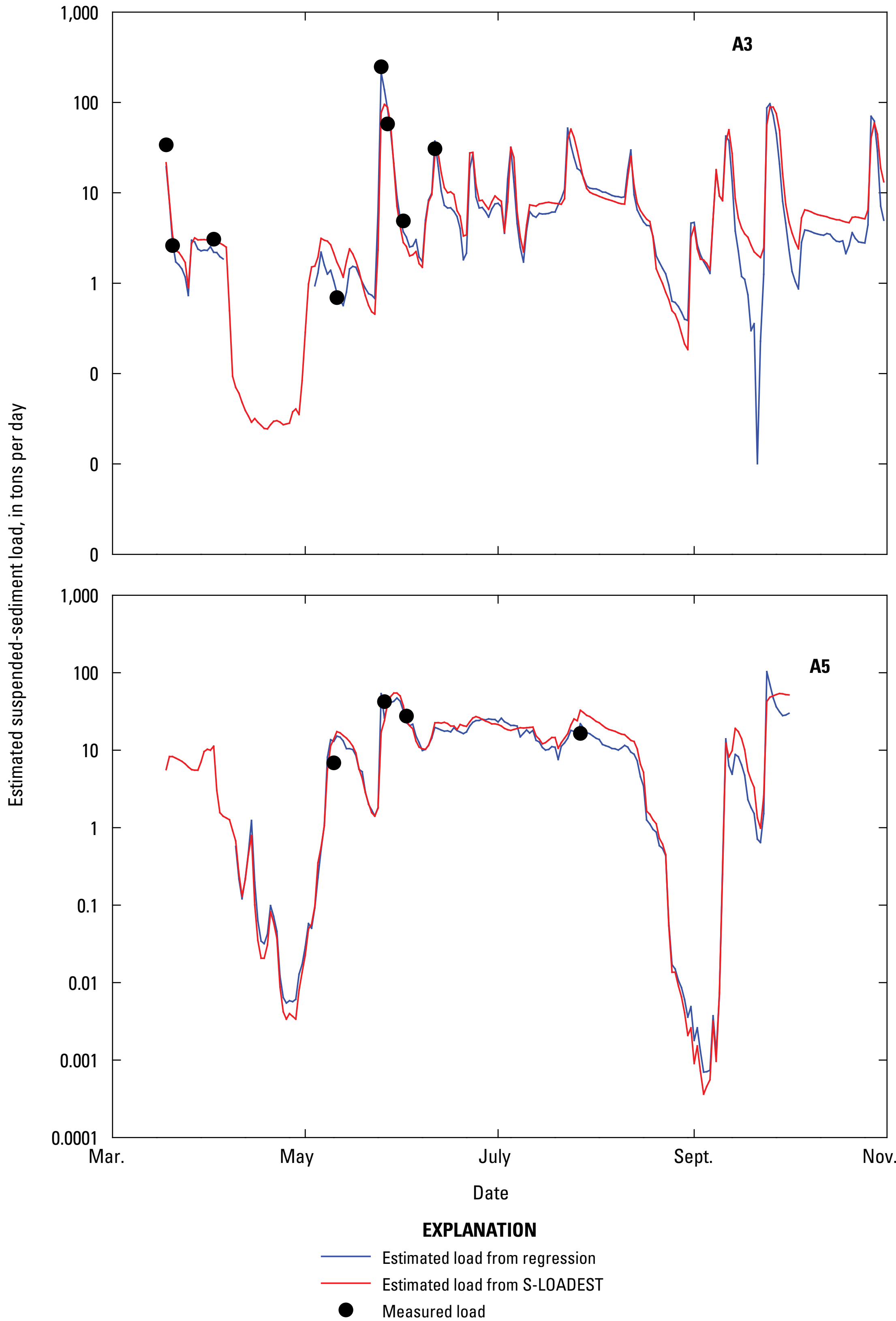

Figure 22. Comparison of measured and estimated suspended-sediment loads for two different estimation methods for select sites in Agassiz National Wildlife Refuge, northwest Minnesota, from March 2010 to October 2010.

Table 10. Results of quality-assurance samples for nutrient and suspended-sediment concentrations for samples collected in and near Agassiz National Wildlife Refuge, northwest Minnesota, 2008 through 2010.

[Calculation of percent difference is: $|x_1-x_2/(x_1+x_2)/2|(100)$, where x_1 = sample, x_2 = sequential replicate; *n*, number of samples; N, nitrogen; P, orthophosphorus]

Constituent	*n*	Average percent difference
Total ammonia plus organic nitrogen as N	14	0.9
Dissolved ammonia as N	13	4.1
Dissolved nitrate plus nitrite	13	0.6
Dissolved orthophosphorus as P	13	1.3
Total phosphorus as P	14	2.9
Suspended sediment	14	6.9

objectives. For example, an objective of determining the effects of changes in water quality through time on wildlife could be addressed with trend analysis. Similarly, an objective determining sources of constituents can be addressed with load estimation. Trend analysis and load estimation methods require, at minimum, the collection of continuous streamflow data and discrete water-quality samples. If only streamflow and discrete samples are collected, loads can be estimated using S-LOADEST. Continuous water-quality monitoring also can provide additional explanatory variables for regression models to improve estimates of constituent loads. Once regressions are developed, fewer discrete samples may be needed. Continuous monitoring also provides direct measurement of important properties such as DO and turbidity, which allows for the characterization of conditions and comparison with water-quality standards. Also, studies with more specific objectives can be met through the use of continuous water-quality monitors, such as metabolism studies that use DO, specific conductance, and water temperature to evaluate ecosystem function. Other sensors such as chlorophyll can be included to assess algal blooms or dissolved organic matter to determine whether these sources are contributing to sediment issues. This report provides an initial characterization of water quality (nutrients and sediment) and estimation of constituent loads for rivers and ditches entering and exiting Agassiz NWR during a 3-year period, during which water-management activities within Agassiz NWR were changing. Future water-quality and streamflow monitoring is needed to be able to document and characterize water quality under different hydrologic and management conditions and to evaluate long-term trends. For Agassiz NWR, a future monitoring program that includes streamgaging and discrete sampling, combined with continuous water-quality monitoring will allow for trend analysis and load estimation, which will support these objectives.

Trend analysis is a statistical technique aimed at detecting annual and seasonal variation with time, and from the analysis, a site-specific monitoring design can be developed (Vecchia, 2000, 2003, and 2005). Based on other studies (Vecchia, 2000, 2003, and 2005), a reasonable sampling design for detecting seasonal and annual trends in nutrients and suspended sediment consists of a five-sample design with samples collected during the same week each month in April, May, June, July and October. Efficient sampling designs can be determined for individual sites using statistical procedures when at least 5 years of data are available. Therefore, if enough data are collected in the future, the sampling design can be reevaluated to determine if it adequately defines water-quality characteristics through time, at specific sites, using a statistical analysis of data for the site. Continuous streamflow also is required; for trend analysis, streamflow is used to determine the variability in water quality attributed to the variability in the hydrologic conditions.

Program designs for the estimation of constituent loads should focus on the main contributing factors that can affect the annual load, such as changes in concentration with streamflow and time. Loads can be estimated using S-LOADEST or through regression analysis with continuous water-quality properties as discussed in this report. In estimating loads, defining the constituent concentration at the full range of streamflows at a site is important; if the high streamflow conditions are not adequately included in the sampling, then annual loads will be estimated lower than what is actually occurring at the site. To better estimate the annual load, or the mass of the constituent that passes the site in a year, program designs must include sampling at the site during the periods of greatest streamflow, typically during snowmelt and storms. The number of samples needed to yield a good estimate of load may depend on the site characteristics (drainage area size, base-flow characteristics) and budgetary constraints. A sampling design for sites similar to the ones sampled for Agassiz NWR that have relatively small drainage areas, quick response times to storm runoff, and only intermittent substantial streamflow, should include as many samples as possible during snowmelt and storms. Sampling at such sites could be implemented using automated pumping samplers, that can collect frequent samples during periods of storm runoff and can be programmed to begin sampling at selected times or at given conditions of stream stage, flow, or turbidity (Edwards and Glysson, 1999).

Based on design requirements for trend analysis and load estimation, a future monitoring program for Agassiz NWR could include the following elements. Indicator sites such as A3 and A2 could be selected to represent inflow and outflow. Inflow site SG140 had the largest volume of all inflow sites and generally contributed the largest loads during the study, however, A3 combined with the Mud River diversion, had a larger volume of inflow and contributed the larger loads during periods of high streamflow. A new site upstream from the diversion to capture all inflow from the Mud River may be beneficial. At these 2 sites, a total of 7 discrete samples and

7 streamflow measurements could be collected consisting of: 5 samples, along with a streamflow measurement, collected during the same week each month in April, May, June, July and October combined with 2 supplementary samples and streamflow measurements during periods of storm runoff. In addition to the discrete samples, continuous water-quality monitors could be deployed at each site.

Streamflow and water-quality monitoring provide valuable information that can be used to meet many different objectives. For this study, the water-quality data collected allow for the characterization and occurrence of nutrients and suspended sediment entering and exiting Agassiz NWR. The water-quality data that were collected indicated that the relatively clear water Branch 1 of Judicial Ditch 11 (site A1) contained high concentrations of phosphorus and nitrogen relative to other sites in and near Agassiz NWR. Also, changes in management conditions, such as releases from Thief Lake and scheduled drawdown of Agassiz Pool, affect water quality. Future monitoring will provide information that can be used to assess the changes in water quality with time, changes in management conditions, effects of upstream mitigation practices (for example, buffer strips, side-channel inlets) within the Thief River Watershed, as well as other variables.

Summary

In response to concerns about water-quality impairments that may affect habitat degradation, the Agassiz National Wildlife Refuge in northwest Minnesota, the U.S. Geological Survey, in cooperation with the U.S. Fish and Wildlife Service collected streamflow data, discrete nutrient and suspended-sediment samples, and continuous water-quality data from 2008 to 2010. Constituent loads were estimated for nutrients and suspended sediment using sample data and streamflow data. In addition, a potential water-quality and streamflow monitoring program design was developed for Agassiz National Wildlife Refuge. Results from this study can be used by resource managers to address identified impairments, protect wildlife habitat and public water supply, and may contribute toward developing more effective water management plans for Agassiz National Wildlife Refuge.

Streamflow was measured by the U.S. Geological Survey at four inflow and two outflow sites located on rivers and drainage ditches in and near Agassiz National Wildlife Refuge during the open-water (no ice cover) periods during 2008, 2009, and 2010. Discrete samples were collected and analyzed for nutrients (total ammonia plus organic nitrogen, dissolved nitrate plus nitrite, dissolved ammonia, total nitrogen, dissolved orthophosphorus, and total phosphorus) and suspended-sediment concentration. Continuous water-quality measurements were collected for water temperature, specific conductance, dissolved-oxygen concentration, pH, and turbidity.

Among the inflow sites, the streamflow volume at A3 and SG140 accounted for about 88 to 92 percent of the measured inflow. The inflow monitored for this study did not include the total streamflow into Agassiz Pool. During the higher streamflow years of 2009 and 2010, it was estimated that doubling both one of the largest and one of the smallest inflow sites would account for unmonitored inflow to Agassiz Pool. During the lower streamflow year of 2008, it was estimated that nearly all inflow to Agassiz Pool was accounted for from the inflow sites monitored in this study. For the outflow sites, water-management activities during the period of data collection affected the streamflow characteristics for the two outflow sites, particularly scheduled drawdown of Agassiz Pool in fall 2009. At all sites, the greatest annual mean streamflow was in 2010 and the least annual mean streamflow was in 2008. The total volume of streamflow in 2010 was 6.3 times greater, on average, for all sites than in 2008.

In 2006, the Thief River from Thief Lake to Agassiz Pool was listed as impaired for high ammonia concentrations. Results from this study indicate that concentrations at all sites did not exceed the 0.04 mg/L water-quality standard for un-ionized ammonia. All sites had concentrations below the drinking water standard of 10 mg/L nitrate plus nitrite as nitrogen. Three of the inflow sites had median concentrations less than the Northern Glaciated Plains shallow lakes ecoregion criteria of 0.9 mg/L total phosphorus, but both outflow sites and inflow site A1 had median concentrations at or above the ecoregion criteria. Compared with the four inflow sites, the two outflow sites generally had significantly greater dissolved ammonia concentrations, significantly smaller nitrate plus nitrite concentrations, and no major differences in total ammonia plus organic nitrogen and total nitrogen. Overall, orthophosphorus and total phosphorus concentrations were significantly greater at inflow site A1 than any other site.

For most sites and constituents, annual (open-water period) nutrient loads were greatest in 2010, which was related in part to larger streamflow volume in 2010. Also, other than nitrate plus nitrite and orthophosphorus, annual nutrient loads were generally greatest at outflow site A2, the site with the greatest volume of streamflow. Of the total streamflow from inflow sites only 3 percent was accounted for by A1; however of the total load from inflow sites, 31 percent of nitrate plus nitrite, 27 percent of orthophosphorus, and 13 percent of total phosphorus was accounted for by A1. The seasonal pattern of mean monthly nutrient loads was affected by streamflow volume (which was affected by precipitation patterns and releases from impoundments) and the growing season. For total ammonia plus organic nitrogen, total nitrogen, and total phosphorus, the greatest loads for most sites generally were in April, May, June, September and October, which corresponded to months of greater streamflow volume. Mean monthly loads for dissolved ammonia tended to be the greatest in March and April, especially at sites downstream from Thief Lake and Agassiz Pool. At the outflow sites, dissolved ammonia loads were substantial in the months of September and October, which may be related to a decrease in plant and algae growth. Sites located downstream from Thief Lake and Agassiz Pool exhibited similar seasonal patterns of nitrate plus nitrite and orthophosphorus loads, with greater loads in the spring and fall and smaller loads in the summer. Although estimated annual nutrient loads generally were greatest in 2010, in many cases, annual flow-weighted concentrations were greatest in 2009. The greater flow-weighted nutrient concentrations in 2009 may have been related to differences in the streamflow pattern between 2009 and 2010. For outflow site A2, the annual flow-weighted concentration of all constituents other than dissolved ammonia and total nitrogen were greatest in 2009, which may have been related to scheduled drawdown of Agassiz Pool. Similar to the mean monthly loads, seasonal patterns of mean monthly flow-weighted concentrations were affected by releases from Thief Lake and Agassiz Pool and the growing season. For inflow sites not affected directly by upstream impoundments, much less variability in flow-weighted concentrations of nitrate plus nitrite and orthophosphorus was observed.

Comparison of discrete suspended-sediment concentrations for all sites indicated small differences among inflow sites, but outflow sites had significantly greater suspended-sediment concentrations than inflow sites. At outflow site A2, during the scheduled drawdown of Agassiz Pool from October 2009 into 2010, suspended-sediment concentrations were high compared to concentrations prior to the scheduled drawdown of Agassiz Pool. Annual suspended-sediment loads were greatest in 2010 for all sites except inflow site SG140 and outflow site A5, with the greatest annual loads at outflow site A2, ranging from 867 tons/yr in 2008 to 29,000 tons/yr in 2010. The large load at outflow site A2 likely resulted from the combination of greater flows in 2010 and scheduled drawdown of Agassiz Pool. Of the three inflow sites to Agassiz Pool, two of the sites accounted for at least 97 percent of the total annual sediment loads from 2008 to 2010. For most sites, the greatest mean monthly loads generally occurred in April, May, June, September, and October, which corresponded with months of greater streamflow. For A3, A4, SG140, and A5, the greatest mean monthly load occurred in March or April. For outflow site A2, the greatest sediment load occurred in October, which is likely related to high concentrations of suspended sediment at the start of scheduled drawdown of Agassiz Pool in October 2009 and large streamflow volume in October of 2010. Flow-weighted sediment concentrations generally were greatest in 2009, which may have been related to differences in precipitation patterns between 2009 and 2010. Mean monthly flow-weighted concentration of sediment follow seasonal patterns similar to mean monthly loads.

A recent (2011) radioisotope study indicates that Agassiz Pool has been experiencing a net gain of sediment from 1940 to 2008, but during the 3-year period of this study (2008 to 2010), a net loss of sediment from Agassiz Pool occurred. A net loss from 2008 to 2010 was likely related to a combination of several atypical water-management activities that occurred at outflow sites A2 and A5 including the following: the first year of operation of the WCS at A5 in 2008, which likely resulted in a flush of sediment, and resulted in some erosion of the new channel immediately downstream from the WCS; construction downstream from A2 in 2008 and 2009 to reduce long-term erosion resulted in bare dirt channels; and scheduled drawdown of Agassiz Pool in fall 2009 through 2010, which occurs only once every 10 years.

Continuous water-quality monitor data from 2010 were compared among sites. Outflow site A2 exhibited the largest diurnal fluctuations in water temperature, dissolved oxygen, and pH of all the sites. Other than outflow site A5, in 2010 all sites had occurrences of hourly dissolved oxygen falling below the water-quality standard of 5 mg/L, with as little as 0.1 to as much as 28 percent of the hourly values below 5 mg/L. For all sites in 2010, pH was above the 6.5 standard units water-quality standard, but pH at SG140, A2, and A5 pH exceeded the 8.5 standard units water-quality standard in 14, 29, and 5 percent of the hourly values, respectively. For all sites in 2010, spikes in turbidity occurred from storm runoff, with as little as 2 percent (A4) of the hourly values exceeding the 25 nephelometric turbidity units water-quality standard and at most 38 percent (A5) of the values exceeding the standard.

A future monitoring program for Agassiz NWR could include data collection at 2 sites (1 inflow and 1 outflow site) with a total of 7 discrete samples and 7 streamflow measurements consisting of: 5 samples, along with a streamflow measurement, collected during the same week each month in April, May, June, July, and October combined with 2 supplementary samples and streamflow measurements during periods of storm runoff. In addition to the discrete samples, continuous water-quality monitors could be deployed at each site. Future monitoring will provide information that can be used to assess the changes in water-quality with time, changes in management conditions, effects of upstream mitigation practices (for example, buffer strips, side-channel inlets) within the Thief River Watershed, as well as other variables.

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