Nutrient and sediment dynamics change following a major flood event on a large, grassland river

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Abstract
River ecosystems can impact the quantity of materials and nutrients flowing into receiving waters. However, these processes may change with climate change, particularly with respect to the impacts of extreme flooding events. In this study, we examined the long-term impacts of an extreme flooding event, the 2019 U.S. Midwest floods, on nutrient loads and concentrations in an unconfined, fourth-order grassland stream, the Niobrara River, Nebraska. To compare historical trends to modern conditions, archived data and contemporarily sampled data were used in conjunction. Prior to the flood (1990–2018), NO3-N concentrations showed a strong seasonal pattern with no relationship with discharge, indicating a groundwater NO3-N source. Total phosphorus (TP), dissolved inorganic phosphorus (DIP), and total suspended solids (TSS) showed slight and individual seasonal patterns relating to discharge, following expected patterns of sediment sourced P. Post-flood, NO3-N concentrations declined by 91.5% (F[2, 269] = 34.96, p < .0001). NO3-N depression in the system remained even once discharge returned to pre-flood levels. TP, DIP, and TSS (F[2, 236] = 38.14, ANOVA p < .0001; F[2, 174] = 44.06, ANOVA p < .0001; F[2, 205] = 25.98, ANOVA p < .0001) each saw elevation of 100%, 117%, and 66% respectively in 2019 with an expected drop in 2020 when discharge returned to pre-flood. It is unclear why NO3-N concentrations, unlike phosphorus, did not return to pre-flood levels in the 2 years of this study. The observation suggests extreme flooding events may cause complex, unexpected shifts in biogeochemical cycling in river ecosystems.

KEYWORDS extreme flood, Niobrara River, nitrate, nutrient cycling, phosphorus

1 | INTRODUCTION

Climate change has led to more frequent, less predictable, and more intense flood events (Sivakumar, 2011; Talbot et al., 2018). Flooding often controls ecosystem services in rivers, raising concern that changes to natural flood regimes will bring unpredictable changes to ecosystem services (Junk, Bayley, & Sparks, 1989). For example, intense rainfall that leads to flooding has been linked to increasing nutrient fluxes to surface waters. In agriculturally developed areas, sediment and water flows from nutrient-rich landscapes raise flooding-related concerns over water quality and its impacts on human and ecosystem health (Correll, Jordan, & Weller, 1999; O’Brien, Rice, Kennedy, & Bricker, 1993; Talbot et al., 2018). However, it is still unclear how flooding impacts long-term trends in nutrient and sediment transport.

Much of our knowledge of nutrient cycling in rivers is derived from baseflow or non-flood conditions rather than flood conditions,
which may disproportionately influence overall material cycling rates (Dodds, Smith, & Lohman, 2002; Peterson et al., 2001). In addition, the magnitude of flooding may impact the ecosystem response. While extreme floods tend to cause decreases of primary production, bank stability, and/or water quality, small to moderate floods tend to have positive relationships with these same characteristics (Talbot et al., 2018). Hence, identifying the conditions that are created by uncharacteristic flooding is a step toward maintaining ecosystem health (Loecke et al., 2017; Sivakumar, 2011). Still, our ability to study extreme floods is limited due to their unpredictable nature and relative rarity (Brázdil & Kundzewicz, 2006).

In the spring of 2019, a major rain-on-snow event occurred within the U.S. Midwest and led to historic levels of flooding across the Midwest (US Department of Commerce, NOAA, 2019). We took this unique opportunity to study how extreme floods impact nutrient cycles and sediment flows in the Niobrara River; a large, grassland river. The Niobrara River is a National Scenic River; hence, the freeflowing conditions and relatively undisturbed and undeveloped shorelines provide an opportunity to study the impacts of the flood with minimal confounding by land-use or geomorphic changes (Alexander, Zelt, & Schaepe, 2010). The objectives of our study were to (a) assess pre-flood trends in sediment and nutrient concentrations and (b) compare pre- and post-flood trends of these attributes. To do this, we combined historical data collected through multiple agencies over the past 20 years with our own contemporary measurements.

2 | METHODS

2.1 | Site description

The Niobrara River is a fourth order, groundwater fed tributary of the Missouri River (Alexander et al., 2010). Our study reach lies within the 122 km stretch of river that is designated as a National Scenic River. The surrounding land is primarily agricultural, supporting ranching, and irrigated row crops. However, the riparian regions remain largely intact. The southern border of the river in our study area is dominated by grasslands while the northern border is largely forested and has a more diverse topography. In addition, the Niobrara River watershed has a history of wildfires, with the most recent, significant fire burning in 2012. The 2012 fire was located downstream from our study area and thus we do not take this into any further consideration during our interpretation of the results.

2.2 | Data selection and manipulation

To describe the pre-flood conditions in the Niobrara River, water chemical data were accessed through the National Water Quality Monitoring Council Water Quality Portal (WQP, https://www.waterqualitydata.us/). We downloaded all available total nitrogen (TN), total phosphorus (TP), nitrate (NO$_3$-N), ammonium (NH$_4$+), dissolved inorganic phosphorus (DIP), soluble reactive phosphorus (SRP), and total suspended solids (TSS) values. Although DIP and SRP are analyzed differently in the lab, we assume that they are the same and herein refer to both values as DIP (Haygarth & Sharples, 2000). Our initial download included sites that were both within the Niobrara watershed (HUC 101500) and within Cherry County, NE (county code 31). Eastern and western borders were defined between longitude values –100.48 and –100.39. Aquatic sites such as wells, tributaries, or ponds were removed by using keywords (removed = “upstream”, “tributary”, “between”; kept = “stream”, “niobrara”).

Data retrieved from the WQP included values reported as non-detects or at detection limits (MDL), resulting in left censored results (data below a certain value by an unknown amount). To avoid over-extrapolation of timeseries trends, we chose to remove any year with >50% of values reported as missing and/or below the MDL value from our study. This resulted in all TN and all NH$_4$+ data being removed, and thus these analytes are not included in our analyses. All years prior to 1990 were too incomplete to include in our analyses, thus a start date of 1990 was used for this study. Imputation of the remaining missing data was done using a randomly generated value between zero and the reported detection limit of the sample. More advanced methods from Shoari and Dubé (2018) were applied with poor results and therefore not used. All data manipulation was completed in R using the package tidyverse (v4.0.3; R Core Team, 2020; v1.3.1; Wickham et al., 2019).

2.3 | Contemporary sampling

As data from WQP for our region of interest was not available after the flood, we collected our own water samples. From June – November of 2019 & 2020 we collected water samples at 15 sites along a 20 km stretch between (42.859528, –100.497500) and (42.876012, –100.266671). Unfiltered water was used for TN and TP analysis, and filtered (Whatman 0.45 μm glass fiber filter) water was used for NO$_3$-N, NH$_4$+ and SRP analysis. Total suspended sediments (TSS) were collected on pre-ashed and pre-weighted Whatman 0.45 μm glass fiber filters. Samples were frozen until analysis at the University of Nebraska-Lincoln. TN and TP concentrations were analyzed following a persulfate digestion using an Astoria Pacific autoanalyzer (APHA, 2005). NO$_3$-N was analyzed using a Thermo Fisher IC-1100 ion chromatograph, NH$_4$+-N by the OPA method using a Turner Designs AquaFluor handheld fluorometer (Taylor et al., 2007), and SRP by the molybdate blue method using a spectrophotometer (APHA, 2005). TSS was measured by calculating the dry weight of material on each filter (Hauer & Lamberti 2017). We collected discharge data via the nearest USGS stream gage (#06461500, approximately 4.5 km below our most downstream sampling site). Historical records were incomplete for TN and NH$_4$+ analysis, and thus were not analyzed further in this study.

2.4 | Analytical methods

To compare between pre- and post- flood values, we only included months from which both pre-flood and contemporary data were
available (June–October). Simple linear regression models were used to understand concentration-discharge relationships. One-way analysis of variance was used to test significant changes between nutrient concentrations across study groups pre-flood, 2019, and 2020. All statistical analysis and data visualization were completed in R using the packages tidyverse and ggpubr (v0.4.0: Kassambara, 2020; v3.0.3: R Core Team, 2020; v1.3.1: Wickham et al., 2019).

3 | RESULTS

3.1 | Seasonality

There was strong seasonality in discharge with peak flows between March–May and minimum flows in July–August (Figure 1). While the pattern held true following the flood, discharge remained above historically observed levels between March 2019 and May 2020. A strong seasonal pattern was observed in pre-flood NO₃-N concentrations (Figure 2). Annual lows were observed between April–August whereas slightly higher concentrations were observed throughout the rest of the year. TP appeared to remain consistent throughout the year while DIP showed slight seasonality as concentrations dipped in summer months. TSS also showed a strong seasonal trend where concentrations dipped from July to November before they rose in December. Although we do not have data for every month, seasonal trends observed in pre-flood data appeared to be disrupted in 2019 before reappearing in 2020 (Figure 3).

3.2 | Nutrient-discharge interactions

From 1990–2018 we found that NO₃-N did not exhibit a linear relationship with discharge ($R^2 = 0.000028$, $p > .05$) (Figure 4). Both analytes including particulate matter (TP, TSS), and DIP showed a strong, positive relationship to discharge (TP: $R^2 = 0.028$, $p < .05$; DIP: $R^2 = 0.058$, $p < .05$; TSS: $R^2 = 0.33$, $p < .05$). Post-flood results differed slightly (NO₃-N: $R^2 = 0.078$, $p < .05$; TP: $R^2 = 0.12$, $p < .05$; DIP: $R^2 = 0.53$, $p < .05$; TSS: $R^2 = 0.17$, $p < .05$) (Figure 4). Summaries of our entire dataset and linear models are provided as Data S1 (Tables S1, S2, and S3).

3.3 | Flood effects

Flooding impacted all analytes. Post-flood NO₃-N concentrations decreased by 91.5%, while TP, DIP, and TSS all increased by 100%, 117%, and 66%, respectively (NO₃-N: $F_{(2, 269)} = 34.96$, $p < .0001$; TP: $F_{(2, 236)} = 38.14$, ANOVA $p < .0001$; DIP: $F_{(2, 176)} = 44.06$, ANOVA $p < .0001$; TSS: $F_{(2, 205)} = 25.98$, ANOVA $p < .0001$; Figure 5). In 2020, the concentrations of all four analytes began to return toward pre-flood concentrations. However, mean concentrations of NO₃-N remained particularly high, 113% below pre-flood concentrations, and did not differ from 2019 concentrations. While concentrations of TP, DIP and TSS were only 35.5%, 51%, and 11.6%, respectively, higher that pre-flood concentrations, they were all significantly less than 2019 concentrations.

![Average monthly discharge from 1980-2020](image_url)

**Figure 1** Monthly discharge averages from 1990–2020 observed at USGS Stream Gage #06461500.
FIGURE 2  Boxplots showing pre-flood (1990–2018) NO₃-N, TP, DIP, and TSS concentrations. Note variable scales for each analyte

FIGURE 3  Boxplots showing 2019 and 2020 NO₃-N, TP, DIP, and TSS concentrations. Note variable scales for each analyte
Our analysis of nutrient and material transport in the Niobrara River suggests that extreme flooding events may have longer term impacts on nitrogen biogeochemical fluxes than on phosphorus. Specifically, our results show a depression in NO$_3$-N concentrations following the flood (>1 year) (Figure 5). We also found positive correlations between discharge and TP, DIP, and TSS, but not with NO$_3$-N, suggesting differential processes are acting on nitrogen and phosphorus cycles (Figure 4). Targeted monitoring efforts are needed to disentangle the mechanism of this ecosystem response.

4.1 | Nitrate

The flooding experienced in 2019 led to a persistent (>1 year) decrease in NO$_3$-N concentrations (Figure 5). Similar phenomena have been observed in various studies following extreme rainfall events (Burt & Arkell, 1987; Correll et al., 1999; O’Brien et al., 1993), however, the persistent nature of the NO$_3$-N reduction found in this study is relatively unique. There are a couple possible explanations for the observed NO$_3$-N reductions post-flood. Within river biogeochemical processes may be driving the seasonal pattern observed in pre-flood NO$_3$-N. Extreme scouring of the streambed may have caused consequential changes to organic matter processing, leading to decreased rates of nitrogen mineralization (Richardson, Bartsch, Bartsch, Kiesling, & Lafrancois, 2019), or increased rates of NO$_3$-N uptake. Alternatively, floodplain reconnection may have led to an increase in denitrification rates across the watershed, leading to lowered NO$_3$-N levels in groundwater feeding the river (Knowlton & Jones, 1997; Noe & Hupp, 2007; Pinay, Ruffinoni, Wondzell, & Gazelle, 1998). These mechanisms are not mutually exclusive, and both may be contributing to the observed decreases in NO$_3$-N concentrations. It is imperative to continue our monitoring efforts to understand if a fundamental and permanent change occurred, or if there is simply a lag in the system and NO$_3$-N levels will return to pre-flood levels soon.

While the scope of our study did not allow for it, full-year monitoring of NO$_3$-N levels would have been useful to identify the conditions causing declining NO$_3$-N levels and to examine potential changes to the seasonal pattern. Broader adoption of in situ NO$_3$-N sensors could help in this regard (e.g., Jones et al., 2018). In addition, targeted measurements of denitrification rates and biological production across the entire floodplain may be key to understanding the relationship between flooding and N-cycling.
4.2 | TP, DIP, TSS

It is important to note that in contrast to NO\textsubscript{3}-N, TP, DIP, and TSS exhibited a relatively expected response to historic flooding. Our results suggest that runoff is a source of phosphorus and sediment to the Niobrara River, aligning with commonly observed phenomena in riverine ecosystems (e.g., Bowes, House, & Hodgkinson, 2003; Bowes, House, Hodgkinson, & Leach, 2005; Junk et al., 1989). Phosphorus and sediment concentrations significantly rose during high flow and quickly returned once stable flow conditions were reached, which is congruent to conceptual models of P cycling in streams and rivers (Bowes et al., 2003; Junk et al., 1989). While seasonal patterns in TP, DIP, and TSS were disrupted in 2019, they showed a far more apparent return toward pre-flood conditions in 2020 than nitrate, highlighting the potential resilience of P-cycling in the Niobrara River to extreme flooding events (Figures 2 and 3).

4.3 | Conclusions and recommendations

Historical data and continuous long-term monitoring efforts can effectively be combined to study the impacts of extreme flooding events on river ecosystems. However, the approach has its limitations, particularly when unexpected patterns are observed, as with the post-flood NO\textsubscript{3}-N concentrations in this study. Additional work is needed to fully understand the mechanism causing NO\textsubscript{3}-N losses and potential disruptions to seasonal nitrogen cycling patterns. Rather than employing an opportunistic approach, future studies must be developed to specifically target the impacts of flooding (Talbot et al., 2018). Based on our work, we suggest that, when possible, future studies should directly examine biogeochemical cycling during and after an extreme flood. This approach, coupled with long-term monitoring efforts, would better our fundamental understanding of how entire river systems respond to increasingly frequent and intense flooding (Ratajczak et al., 2018).

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DATA AVAILABILITY STATEMENT

Historical data (1990-2018) are available for download from the National Water Quality Monitoring Council (https://www.waterqualitydata.us/portal/). 2019 and 2020 data, collected at the University of Nebraska - Lincoln, are available from the corresponding author, Jessica Corman, upon reasonable request.

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