Abstract

The Neal Smith National Wildlife Refuge was established as a tallgrass prairie ecosystem reconstruction in the Walnut Creek watershed (5238 ha), Jasper County, Iowa, with >1200 ha of prairie plantings initiated between 1993 and 2006. This study updates the documented decreases in watershed NO₃–N losses that accompanied this change in land cover to a 20-yr record. Annual flow-weighted NO₃–N concentrations declined by 0.15 mg NO₃–N L⁻¹ yr⁻¹, which was not significantly different from the rate of 0.07 mg NO₃–N L⁻¹ yr⁻¹ reported after the first decade of monitoring. There was also evidence \( (p < 0.1) \) that prairie reconstruction led to a declining trend in annual watershed water yield, which would have contributed to the trend of decreasing NO₃–N loads. However, variability in climate, including 2 yr with significant flooding events followed by a major drought during the second decade of monitoring, challenged any notion that a watershed water quality record will stabilize even >10 yr after a substantial change in land cover, in this naturally drained watershed underlain by fine grained glacial deposits that exhibit multidecadal groundwater transport times. Efforts to document progress toward water quality goals will need to consider dominant flow paths and associated travel times, uncertainty in the effectiveness of management changes, and a changeable climate.

Core Ideas

- Nitrate in streamflow draining a prairie reconstruction was tracked for 20 yr.
- Nitrate loss decreased with time, at a slow rate of only 0.15 mg NO₃–N L⁻¹ yr⁻¹.
- Even with substantial land use change, water quality response may take decades.
annual cropping, which led to increased NO₃⁻N losses during 1996 to 2005. This contrast in land cover change between the two watersheds showed that declines in water quality can occur quickly, but recovery of degraded water quality can be expected to occur slowly for soluble contaminants (e.g., NO₃⁻N) delivered to surface waters via groundwater baseflow.

The two Walnut Creek stream gages were reestablished by the USDA-ARS in late 2006. The objective of this study was to assess the NO₃⁻N water quality record for Walnut Creek after a second decade of prairie reconstruction and stream monitoring and thereby update any continued trends in NO₃⁻N concentrations and loads from this watershed.

**Materials and Methods**

For this second decade of monitoring, the two prior established stream gages on Walnut Creek were instrumented with noncontact bubbler stage sensors and electronic peristaltic automated samplers. The UG and LG sites were denoted WNT1 and WNT2, respectively, by Schilling and Spooner (2006). Water samples were manually collected on each visit to the gauge sites, which occurred at least weekly, using an ice axe to collect liquid samples when necessary during winter freeze conditions. The automated samplers were programmed to collect one flow-weighted composite sample for each runoff event, triggered by a rise in stage, and composed of 100-mL subsamples collected every 0.5 mm of discharge. The NO₃⁻N concentration of each composite sample was used to represent concentration during the event when calculating daily values. Walnut Creek exhibits a flashy hydrology, with most runoff events returning to baseflow conditions within 10 h. Winter runoff events were not autosampled to avoid freeze risks to equipment; snowmelt has been shown to be less important to nutrient losses in central Iowa watersheds than in more northern climes of the Midwest in recent years (Kalkhoff et al., 2016). The water samples were transported to the laboratory where NO₃⁻N concentrations were determined by autoanalyzer (Wood et al., 1967); the detection limit was 0.3 mg NO₃⁻N L⁻¹. Nitrate-N concentrations were interpolated to daily values by linear interpolation; nondetectable concentrations were assigned half the detection limit (i.e., 0.15 mg NO₃⁻N L⁻¹) for calculating loads. Flow volumes were recorded every 10 min and summed to daily (Q) values. Daily NO₃⁻N loads were calculated by multiplying daily NO₃⁻N concentrations and discharge.

The first statistical analysis was conducted on measured NO₃⁻N concentrations of manually collected samples and regressed NO₃⁻N concentrations of samples taken at the UG against those at the LG, paired by date; sample pairs with any nondetectable concentrations were excluded. Subsequent analyses were conducted for monthly and annual data, obtained by summing daily values of discharge and NO₃⁻N loads by calendar month and water year (1 October–30 September). Flow-weighted NO₃⁻N concentrations were calculated by dividing the monthly and annual NO₃⁻N loads by corresponding discharge volumes. The gages were reestablished during October 2006;
low-flow conditions occurred that month, and therefore, 2007 was used as the first year of record (despite missing nearly 3 wk of low-flow data), whereas November 2006 was the first month of record. Because the land cover above the UG was dominantly in row crop production and NSNWR covered part of the watershed between the two gages (Fig. 1), the effect of the NSNWR prairie reconstructions on water quality was first assessed by estimating the proportion of the NO₃–N loads that originated from the lower part of the watershed, then determining whether this proportion showed a temporal trend. This proportion was calculated from $P_t = 1 - \left(\frac{UG_t}{LG_t}\right)$ where $P_t$ is the proportion of the load from the lower watershed, and $UG_t$ and $LG_t$ are the monthly or annual NO₃–N loads at the upper and lower gages, respectively. The resulting 10-yr series of monthly proportions, being normally distributed, was regressed against time (in years). Data points for 11 mo were deleted because there was either no discharge (occurred at the UG during a 2012 drought), or the FWC of NO₃–N was at or below the detection limit at one or both gages. Regression residuals were checked and were not significantly autocorrelated ($p > 0.1$). When we analyzed the annual data, annual proportions and FWCSs of NO₃–N were regressed against time using all 20 yr of data, including data from 1996 to 2005 as tabulated by Schilling and Spooner (2006).

The analysis of the proportional NO₃–N load data assumes that all NO₃–N that passes the UG also passes the LG (i.e., no significant losses of NO₃–N occurred due to biological uptake or denitrification within the stream channel between the gages). This assumption was also made by Schilling and Spooner (2006). Slopes of $\log(Q) - \log(\text{NO₃–N load})$ plots were examined as a check of this assumption. An increase in the slope of this relationship from the UG to the LG would indicate substantial N loss along the channel, as previously shown for a different Walnut Creek in Iowa (Tomer et al., 2003), where in-stream NO₃–N losses along the channel were attributed by losses to, and dilution by, groundwater associated with coarse-textured glacial outwash.

A shift from annual cropping to prairie may result in increased evapotranspiration, as reported by Brye et al. (2000) and Daigh et al. (2014), but there are also comparative studies that have shown no significant difference in water uptake between row crops and native grasses (Hamilton et al., 2015). Nevertheless, prairie vegetation is known to have greater root biomass and depth than annual crops (Jarchow et al., 2014), and at NSNWR, prairie establishment has resulted in increased water table depths over time (Schilling and Jacobson, 2010), which would reduce the hydraulic gradients driving groundwater contributions to streamflow from prairie restoration sites. We sought evidence as to whether water yield was decreased by prairie establishment, which could result from greater evapotranspiration and/or lower water table elevations under prairie. Precipitation data recorded at a single rain gauge in the lower watershed were downloaded from the MesoWest database (University of Utah, 2018), which maintains data for a network of weather stations located on public lands. We calculated the fraction of annual precipitation that became discharge from the upper (UG) and lower (LG – UG) watersheds, then regressed the annual LG discharge fractions (dependent) against the annual UG discharge fractions (independent). Using a stepwise approach, we then entered year since the first prairie establishment (1992 = Year 1) as a second independent variable.

The extent of the Walnut Creek watershed and its land cover were updated using watershed data and terrain analyses from the Agricultural Conservation Planning Planning Framework (Tomer et al., 2017; Porter et al., 2018), which included land cover data from the Cropland Data Layer (USDA-NASS, 2018). The assessment includes an estimation of the extent of tile drainage in the watershed; only ~5% of the watershed has both low slopes (<5%) and dual soil hydrologic groups indicative of patterned tile drainage (Porter et al., 2018). In addition, watershed delineation results confirmed drainage areas reported by Schilling and Spooner (2006) to within a few hectares.

**Results and Discussion**

**Land Cover**

Watershed delineation showed that the drainage areas for the two gages were 1745 ha above the UG and 5238 ha above the LG, leaving by difference 3493 ha contributing to Walnut Creek between the two gages. Annual crop cover data (USDA-NASS, 2018) were evaluated from 2008 to 2016 (i.e., 9 yr of the second decade of monitoring). The upper watershed was 82 to 87% under corn (*Zea mays* L.) and soybean (*Glycine max* (L.) Merr.) cover during these years, whereas the lower watershed had 37 to 42% coverage of these crops (Fig. 1). Perennial vegetation occupied 8 to 13% of the upper watershed and 53 to 59% of the lower watershed. The perennial cover was dominated by grazed pastures in the uppersheds, and by woodlands, grazed pastures, and prairie reconstructions of the lower watershed. The prairie reconstructions alone comprised 35% of the lower watershed’s land cover. Differences in perennial cover are difficult to discern when classifying satellite remote sensing data (Tomer et al., 2017); therefore, perennial cover is shown as reconstructed prairie within the NSNWR boundary, and pasture or hay outside this boundary, in Fig. 1. Wetland areas are not known, nor apparent in the land cover data, and open water in the watershed is limited to the stream network.

**Precipitation and Discharge, 2007–2016**

The second 10-yr period of monitoring exhibited substantial variation in precipitation and discharge (Table 1, Fig. 2). Annual precipitation averaged 907 mm and varied from 603 mm in 2012 to 1546 mm in 2010. During the first decade of monitoring, precipitation in the lower watershed averaged 836 mm and ranged between 642 and 1056 mm (Schilling and Spooner, 2006). Seasonally, precipitation in Iowa is usually greatest during May and June, but the flood of record for this watershed occurred during 8 to 11 Aug. 2010, when 263 mm of rainfall led to >90 mm of cumulative discharge across these 4 d, versus <3 mm d⁻¹ of baseflow discharge just before this storm. A significant drought then occurred during 2012–2013. No discharge was observed at the UG during the 6 mo between September 2012 and January 2014. The minimum monthly discharge at the LG was 0.05 mm during September 2012. These drought and flood events dominated streamflow variation and deterred analysis of seasonal patterns and differences, particularly because the largest flood occurred when stream discharge is typically least, in late summer 2010.

Annual discharge at the LG ranged from 66 mm in 2012 to 882 mm in 2010 and at the UG ranged from 50 mm in 2012 to 937 mm in 2012 (Table 1). During this second decade, area-weighted discharge at the UG exceeded that from the lower
part of the watershed (determined by subtracting the UG discharge from the LG discharge and dividing by the area of the lower watershed) every year except during the years with the least discharge, 2012 and 2014.

**NO₃⁻N Concentrations, 2007–2016**

Monthly FWCs for NO₃⁻N (Fig. 2) show variations related to season and climate. The largest FWCs occurred following droughts in 2006 and 2012. Soil N that is mineralized during drought accumulates and is subsequently released as stream flow recovers from drought; this can lead to large NO₃⁻N concentrations, as observed in 2007 and 2013 (Fig. 2). Relatively low NO₃⁻N concentrations occurred after high flows in 2008 and 2010, but the lowest concentrations were nondetectable and occurred during the drought that began in 2012.

Nitrate-N concentrations measured in water samples collected at the two gages were paired by date and plotted against one another (Fig. 3). Each point on this plot represents a pair of samples, manually collected at the UG and LG, on the same day and typically within 2 h of one another. A regression between the NO₃⁻N concentrations collected at the two gages shows that NO₃⁻N concentrations at the LG were, on average, ~73% of those measured at the UG (Fig. 3); the 95% confidence interval for this regression slope was 0.70 to 0.76. Paired sample data were also plotted by Schilling and Spooner (2006) for the first decade of monitoring, with a slope of 0.84 between paired UG–LG samples collected during the first decade of monitoring. These results indicate that the relative dilution of NO₃⁻N from the UG to the LG was increased during the second decade of monitoring, and that the attenuation of NO₃⁻N losses under prairie reconstruction increased as the plantings became fully established.

**Monthly Data, 2007–2016**

The relationships of monthly discharge with NO₃⁻N loads for the two gages, when plotted on log–log scales and fitted to power functions (Fig. 4), showed slopes (exponents) that were similar ($p > 0.05$) and intercepts that differed ($p < 0.05$). Larger slope values are found when low flows are subject to N loss or baseflow dilution along the stream channel (Tomer et al., 2003). The similarity in slopes for the two gages suggests that in-stream losses of NO₃⁻N (i.e., due to denitrification or plant uptake) were minimal, and that differences in per-unit-area loads between the two gages were similar, proportionately, across the range in flow conditions observed. Walnut Creek has been channelized, has steep banks, and is disconnected from its flood plain (Schilling and Drobney, 2014), which would minimize the opportunity for nutrient assimilation along the Walnut Creek channel.

The proportion of the monthly Walnut Creek NO₃⁻N loads attributed to the lower watershed showed a decline in time (Fig. 5). The rate of decline was small, with a slope of ~0.01 yr⁻¹ that accounted for only 7.5% of variation in the monthly proportions but was nevertheless significant ($p < 0.01$). When effects of changes in land cover that affect groundwater transport to streams are being evaluated using stream data in watersheds underlain by fine-grained sediments, great patience is required.

<table>
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Annual Data, 1996–2016

The slow rate of decline in proportions of lower watershed contributions to monthly NO₃⁻N loads was corroborated by the annual data, which included both decades of monitoring. The full record of annual data again estimated a decline in the proportion of NO₃⁻N load from the lower watershed at −0.01 yr⁻¹ (Fig. 6). Annual FWCs of NO₃⁻N declined at −0.15 mg NO₃⁻N L⁻¹ yr⁻¹ across the 20 yr of monitoring (Fig. 6), when the FWC was calculated by subtracting the UG load from the LG load and dividing by the difference in annual discharge. This rate of decline was slightly greater than, but not significantly different from, the rate of −0.07 mg NO₃⁻N L⁻¹ yr⁻¹ reported by Schilling and Spooner (2006) after the first decade of monitoring, using paired UG–LG comparisons.

There was also evidence that a slight decline in annual water yield from the lower Walnut Creek watershed contributed to decreased NO₃⁻N loads. The annual discharge fractions of precipitation from the lower watershed were strongly related to those from the upper watershed (R² = 0.81). However, the count of years since the first prairie establishment (1992 = 1) was entered into a multiple regression equation with a p value of 0.08, which suggested a fractional discharge decline of −0.004 yr⁻¹. This is obviously a slow rate of decline but suggests that changes in groundwater hydrology and groundwater nutrient concentrations may both be contributing to reductions in annual NO₃⁻N losses from the lower watershed. We believe the hydrologic changes would be attributable to lower hydraulic gradients in the groundwater flow system in the lower watershed resulting from declines in water table elevation under the prairie reconstructions (Schilling and Jacobson, 2010).
Synthesis and Conclusion

After the first decade of monitoring, Schilling and Spooner (2006) reported two NO₃–N reduction rates for the LG gauge based on which control was used. Using the paired control watershed for comparison (Squaw Creek), the NO₃–N reduction was 0.12 mg NO₃–N mg⁻¹ L⁻¹, whereas using the upstream site for comparison, the NO₃–N reduction was 0.07 mg NO₃–N mg⁻¹ L⁻¹. The latter reduction rate is deemed more relevant to this updated analysis. Including a second decade of monitoring led to an estimated annual rate of decline of ~0.15 mg NO₃–N mg⁻¹ L⁻¹. We expected that a larger rate of decline would be observed with time as prairie reconstructions became fully established and effects of historical N fertilizer applications on groundwater baseflow faded (Schilling and Wolter, 2007). The longer 20-yr record did show a greater rate of NO₃–N decline, but the two rates were not statistically different. Soils and underlying sediments in this watershed are fine textured, and buried soils are present that would presumably provide C to increase the potential for denitrifying conditions in groundwater. Indeed, groundwater monitoring beneath reconstructed prairie field sites showed faster rates of decline in NO₃–N concentrations than we observed at watershed scale in this study. Using a chronosequence of prairie plantings, Schilling and Jacobson (2010) reported an average decline of ~0.6 mg NO₃–N L⁻¹ yr⁻¹, whereas Tomer et al. (2010) observed a reduction rate of ~1.9 mg NO₃–N L⁻¹ yr⁻¹ in groundwater beneath the uplands of a 7-ha NSNWR catchment, in the first years after prairie seeding. Therefore, our watershed observations show a much slower rate of NO₃–N concentration decline than has been observed when tracking groundwater beneath recent prairie reconstructions at the field scale. However, both field-based studies (Schilling and Jacobson, 2010; Tomer et al., 2010) showed that groundwater NO₃–N concentrations beneath prairie reconstructions typically declined to <5.0 mg NO₃–N L⁻¹ within 5 yr after planting, comparable with recent annual FWC results from the lower watershed (Fig. 6). The 20-yr stream record suggests (p < 0.1) that discharge also declined under prairie reconstruction. This was not necessarily expected, but both the discharge and NO₃–N trends are corroborated by a prairie chronosequence study that showed NO₃–N concentrations and groundwater levels both declined beneath prairie reconstructions at NSNWR. Declining water levels would reduce hydraulic gradients in the lower watershed that drive baseflow contributions to streamflow. This helps explain the relative decrease in Q indicated for the lower watershed and increases the time for groundwater N delivery to the stream to stabilize. Groundwater travel times in the glacial deposits of NSNWR are slow, as residence times on the order of decades have been estimated for several prairie sites in the Walnut Creek watershed (Schilling and Wolter, 2007).

Detection of a declining trend in the proportion of monthly NO₃–N loads from the lower watershed (Fig. 5) was statistically significant (p < 0.01) but accounted for little of the variation found in the data (R² = 0.075). This approach using load proportions normalized the data distribution, and although the treatment impact was detected, results show that discerning conservation impacts in landscape- and watershed-scale studies under varying cropland management histories and climates can be difficult even with long-term observations.

Opportunities to measure watershed-scale water quality responses to large-scale land use change are rare and require a long-term commitment. This study has shown that >20 yr of monitoring will be required in Walnut Creek to fully document reduced NO₃–N loads resulting from NSNWR prairie reconstructions, which occupy more than a third of the watershed. Public agencies charged with improving surface water quality often set water quality goals, and timetables to achieve those goals, in watersheds where land use conversion on a third of the land is not considered socially or economically feasible. As progress toward water quality goals is assessed, multidecadal time lags, climatic shifts, and uncertainties in the effectiveness of measures undertaken will all need to be considered. Although our results in Walnut Creek caution against expectations of early and substantial progress toward water quality improvement, the watershed context included herein consists of fine-grained substrates, an incised terrain, and limited-extent artificial drainage, where longer lag times should be expected compared with more permeable and/or artificially drained landscapes.

Conflict of Interest

The authors declare no conflict of interest.

Acknowledgments

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References


