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Ecological Restoration of a Cultivated Wetland to Enhance Nitrogen Retention in a Rapidly Developing Watershed

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Key Points:

- Nitrogen (N) from septic systems is degrading water quality in Massachusetts, but sewering is not currently a financially viable option
- Wetland restoration retained 28% of N largely during warm summer months due to temperature effects on in-stream processes of N retention
- Wetland restoration can reduce N inputs to coastal embayments, but is an unlikely alternative to centralized sewering

Supporting Information:

Supporting Information may be found in the online version of this article.

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Abstract For much of the coastal United States, water quality continues to worsen due to excess nitrogen (N) from agricultural production and urban development. Restoration of freshwater wetlands, especially those connected to streams (i.e., fluvial wetlands), is a promising management option for curtailing watershed inputs of N to coastal embayments, with regional models assuming 77% retention of N by natural wetlands. In this study, we combine modeling, field measurements, and mass-balance analysis to quantify N inputs and retention in a 56-ha, 7-year-old restored fluvial wetland in southeastern Massachusetts, United States. Rapid watershed land-use change, including development of ∼30% of forested land between 2001 and 2019, contributed to wastewater and turfgrass fertilizers accounting for the majority (80%) of watershed inputs of N to the restored wetland compared with the atmosphere (13%), a landfill (7%), and agricultural fertilizers (<1%). Field measurements showed that in-stream processes retained 28% of surface water and groundwater inputs of nitrate (NO_3^-), which was slightly lower than N retention for restored wetlands and about half the N retention in natural wetlands. NO_3^- retention of 1,029 kg N yr^{-1} was not strongly correlated with stream residence time or NO_3^- inputs, but rather to stream temperature, pointing to environmental effects on in-stream processes such as uptake and denitrification. Although NO_3^- retention may increase as the wetland matures, the current N removal rate suggests that regional models may overestimate N removal in restored wetlands and that wetland restoration is an unlikely substitute for centralized sewering or other forms of enhanced wastewater management.

1. Introduction

Along with carbon, hydrogen, and oxygen, nitrogen (N) is an element essential to life that cycles between atmospheric, terrestrial, and marine ecosystems by mostly microbially mediated processes. However, no other basic element has been altered more by the impacts of humans than N (Katz, 2020). Even atmospheric carbon, which has been dramatically impacted by combustion of fossil fuels, has increased by 50% from preindustrial concentrations, whereas reactive N has increased by 100% from fossil fuel and biomass burning and production of synthetic fertilizers (Erisman et al., 2015). The environmental consequences of excess reactive N are well documented at both regional and global scales (Galloway et al., 2008; Houlton et al., 2019; Sobota et al., 2013), especially for coastal ecosystems where N is generally limiting for phytoplankton production (Rabalais, 2002). In the United States, excess N has resulted in two-thirds of its estuaries exhibiting moderate to high levels of eutrophication (Bricker et al., 2008).

As is the case for much of the eastern United States, coastal ecosystems in southeastern Massachusetts reflect the consequences of unregulated N inputs on watersheds, including loss of eelgrass beds, persistent hypoxia, and reduction in benthic invertebrate species richness (Luk et al., 2019; Valiela et al., 1992). To help address the problems caused by excess N loading, $12 million in local, state, and federal funding was used to establish the Massachusetts Estuaries Project (MEP), which evaluated sources of N to approximately 70 coastal embayments in southeastern Massachusetts (MADEP, 2003). Overwhelmingly, septic systems were found to be the most pervasive source of N pollution in these coastal embayments (MADEP, 2022). For example, in Waquoit Bay, an impaired coastal embayment on Cape Cod, the MEP (Howes et al., 2011) and others (Valiela et al., 1997) reported that 60%–85% of controllable (non-atmospheric) N was due to septic systems, but that declines in ecosystem health were reversible over a relatively short time period (∼10 years) through broad sewer service and centralized wastewater treatment. Despite these recommendations, the number of septic systems in proximity to Waquoit Bay increased by ∼60% between 1990 and 2014 (Valiela et al., 2016, their Figure 3). Today, Waquoit Bay, which is completely eutrophic (MADEP, 2020), has become emblematic of environmental decline on Cape Cod, with 87% of its coastal embayments impaired by excess N (APCC, 2022).

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Scientific consensus exists regarding septic systems as the primary source of N pollution in southeastern Massachusetts and that there is potential to improve water quality through sewering (e.g., Howes et al., 2011; Valiela et al., 1997; Williamson et al., 2017). However, questions remain about how to finance the estimated $4.2 to $6.2 billion in capital costs to sewer Cape Cod (Cape Cod Commission, 2013). Proposed N removal alternatives to sewering include fertilizer management, tidal flushing, stormwater control and treatment, oyster aquaculture, and restoration of freshwater ponds and wetlands (Buzzards Bay National Estuary Program, 2013; Hansen, 2017). Of these, restoration of wetlands on retired farmland has received perhaps the strongest support from state and federal agencies (MADER, 2022; USDA-NRCS, 2020) and, notably, the cranberry industry (MDAR, 2016). Although ecological restoration of wetlands formerly cultivated for cranberry production is somewhat unique to Massachusetts, restoring wetlands on retired farmland has been proposed as part of comprehensive watershed management plans to curtail nutrient inputs to the Gulf of Mexico (Cheng et al., 2020) and Chesapeake Bay (Goldman & Needelman, 2015).

Wetland restoration enhances the soil processes and properties that promote denitrification, including microbial biomass, soil moisture, and organic matter (Ballantine et al., 2017). Specifically, ecological restoration of retired cranberry farms often includes plugging ditches, removing water control structures, adding microtopography, and naturalizing stream channels (Ballantine et al., 2020). Creating microtopography (e.g., varying soil surface roughness and relief) during wetland restoration facilitates adjacent oxic and anoxic zones, which may promote coupled nitrification-denitrification (Bruland et al., 2006; Wolf et al., 2011). Further, naturalizing stream channels leads to greater geomorphic complexity, which promotes N uptake by slowing water velocity and enhancing transient storage, allowing time for biogeochemical transformations of N to occur (Bukaveckas, 2007). In fluvial wetlands, primary drivers of N retention include N concentration in the inflow, hydrologic connectivity between the main channel and carbon-rich floodplain sediments, and hydraulic residence time (Arheimer & Wittgren, 2002; Land et al., 2016; Powers et al., 2012; Wollheim et al., 2014). Beyond morphology and hydrology, seasonality can also impact N retention in wetlands; macrophyte biomass, vegetative N uptake, and microbially-mediated N removal processes are often greater at higher temperatures, particularly during the summer (Clarke, 2002; Lin et al., 2021; McGuire et al., 1992; Vymazal, 2007).

The amount of N retained by restoring wetlands on cranberry farms has not been quantified, but a 77% N retention rate was assumed for natural wetlands in a model of N loading to Waquoit Bay on Cape Cod, Massachusetts (Valiela et al., 1997). This retention rate is greater than the average N retention for many natural wetlands. Others have reported mean N retention of 67% (54 wetlands; Fisher & Acreman, 2004) and median N retention of 64% (23 wetlands; Saunders & Kalff, 2001). Retention rates are often lower in restored and constructed wetlands, ranging from 29% to 44% in a synthesis of 203 wetlands in Europe and North America (Land et al., 2016) and from 40% to 55% for 287 wetland measurements in Asia, North America, Europe, and Australia (Vymazal, 2007). Given the wide range of N retention that has been observed for natural and created wetlands, it is important to investigate N retention in wetlands restored on retired farmland to quantify the impact of this emerging management strategy on N removal.

In this study, we quantified N retention in a large, restored wetland in Plymouth, Massachusetts. We used geospatial analysis, water quality modeling, and field measurements to quantify watershed land use change, sources and magnitude of N loads, and the amounts and mechanisms of N attenuation for the restored wetland. This information will improve the accuracy of watershed N loading models and aid in evaluating wetland restoration as an N management strategy.

2. Study Site

The study site is Tidmarsh Farms East ("Tidmarsh"), an extensive (56 ha) cultivated peatland in the Manomet section of the town of Plymouth, Massachusetts (Figure 1a). Tidmarsh was a profitable cranberry operation until 2010, when the farm was retired from production after an easement for conservation and restoration was completed with the US Department of Agriculture's Natural Resources Conservation Service (Ballantine et al., 2020). With over $3 million in federal, state, and private funding (O'Connor, 2017; Truslow, 2018), 91 ha of Tidmarsh and its surrounding area were ecologically restored in 2016, making it the largest freshwater ecological restoration project to date in Massachusetts (Ballantine et al., 2020). Active restoration included: (a) removal of eight dams and water control structures; (b) addition of ~40 ha of microtopography, 2 flow-through ponds, and 20,000 (native) plants; and (c) naturalization of 5.6 km of stream channel that included reforming the channel with

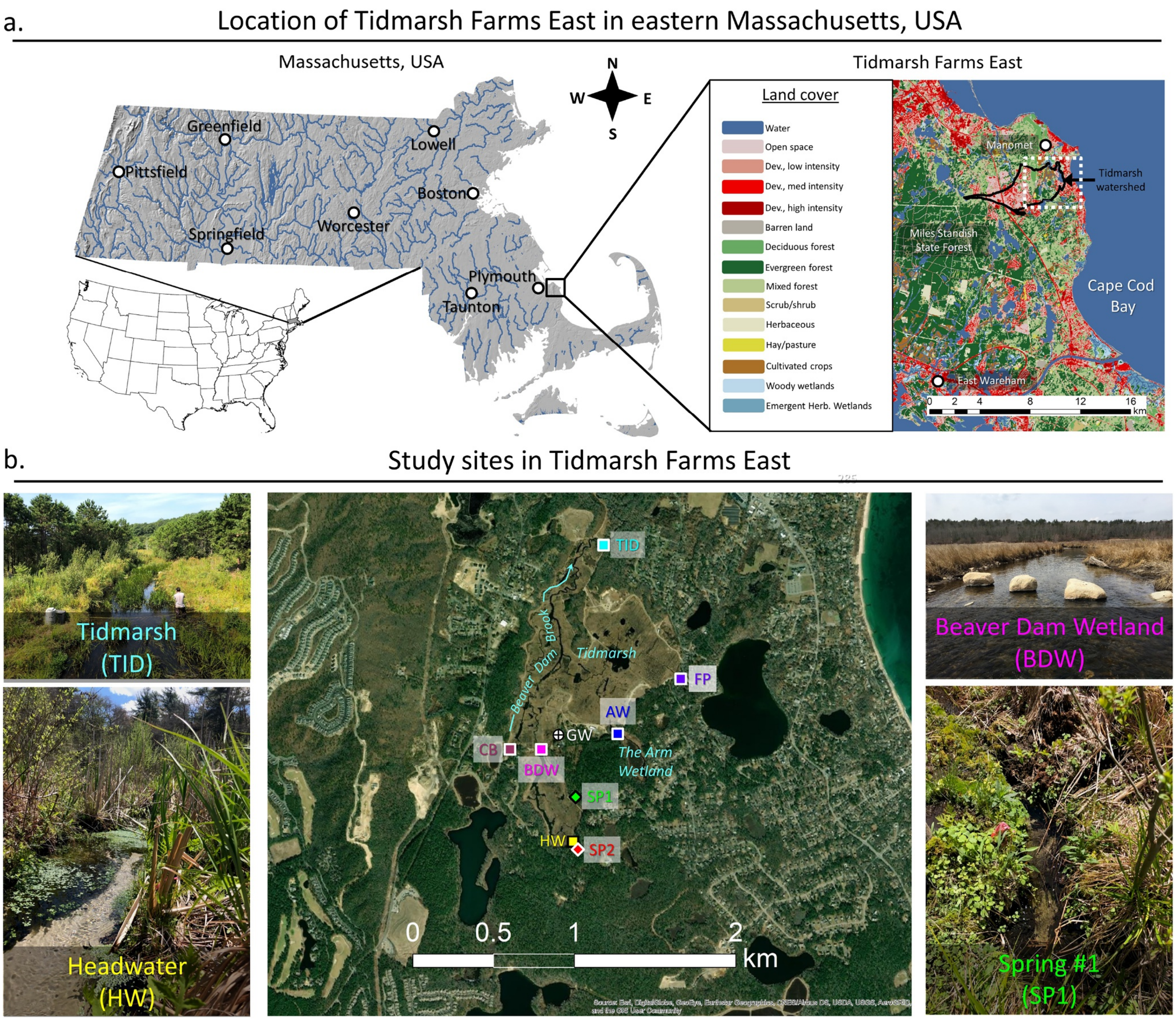

Figure 1. Site map for Tidmarsh Farms East ("Tidmarsh") with the watershed area for Beaver Dam Brook at the outlet of Tidmarsh (black line) and National Land Cover Database land use for 2020 (a). Aerial photograph of Tidmarsh for 2020 and sampling locations along Beaver Dam Brook (streamflow is northward): the outlet of Tidmarsh ("TID"; surface water downstream of Tidmarsh), Beaver Dam Wetland ("BDW"; surface water upstream of Tidmarsh), the Arm Wetland ("AW"; surface water input to Tidmarsh), Fresh Pond ("FP"; surface water input to Tidmarsh), baseflow at the headwater of Beaver Dam Brook ("HW"; groundwater input to Tidmarsh), groundwater spring #1 ("SP1"; groundwater input to Tidmarsh), groundwater spring #2 ("SP2" groundwater input to Tidmarsh), a shallow groundwater well ("GW"; groundwater input to Tidmarsh), and a small cranberry bog ("CB"; surface water input to Tidmarsh) (b).

meanders, riffle-pool structures, and over 3,000 pieces of large woody debris (Ballantine et al., 2020). Today, Tidmarsh is a functional wetland providing several ecosystem services such as habitat provision, recreation, and water quality.

While Tidmarsh was actively farmed for cranberries, it was "sanded," a common agricultural practice of applying 1–5 cm of sand every 2–5 years to promote reproductive growth (DeMoranville & Sandler, 2008), that resulted in 0.3–1.5 m layer of sand overlying parent soils (Hare et al., 2017). The subsurface structure of Tidmarsh consists of four isolated kettle-hole depressions with maximum peat thickness of ∼10 m (Hare et al., 2017). The uplands of Tidmarsh consist of outwash deposits, which are the primary water-bearing units of the Plymouth–Carver–Kingston–Duxbury (PCKD) aquifer system (Masterson et al., 2009), an unconfined aquifer with an area of 750 km^2 (Hatch & Ito, 2022) and a thickness ranging from 6 to 60 m and storing more than 1.9 billion m^3 of fresh water (J. R. Williams & Tasker, 1974). Aquifer sediments consist of coarse-medium sands with

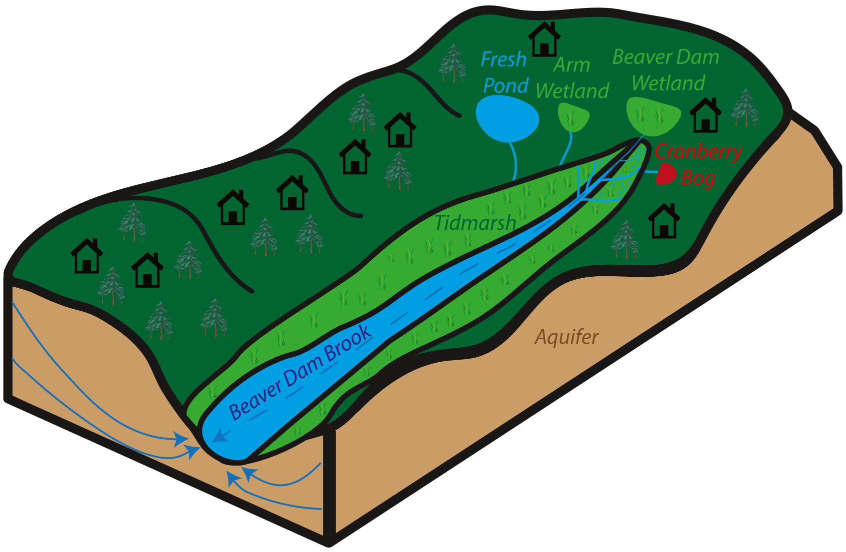

Figure 2. Conceptual model of hydrological and nitrogen (N) transport processes in Tidmarsh and Beaver Dam Brook. Surface water inputs include Fresh Pond, the Arm Wetland, Beaver Dam Wetland, and a small cranberry bog. Groundwater flow enters Beaver Dam Brook as upwelling discharge through channel sediments and groundwater springs, which are most prominent near the stream headwaters. The watershed is mostly forested with residential development and minor agriculture.

relatively high values of hydraulic conductivity, ranging from 30 to 46 m d^{-1} (Masterson et al., 2009). Approximately 1.35×10^6 m^3 d^{-1} of precipitation is recharged to the aquifer and discharged as groundwater to streams and ponds (70%), directly to coasts (25%), and pumping wells (5%) (Masterson et al., 2009).

Long-term (1991–2020) mean annual precipitation in Plymouth is 1271 mm, with monthly precipitation varying by 12% (NOAA, 2022). Long-term mean annual temperature is 9.9°C (NOAA, 2022) and characterized by warm summers (mean July temperature 22.2°C) and cold winters (mean January temperature 1.6°C). Tidmarsh receives surface water inputs from Fresh Pond, the Arm Wetland (a pond drained in 2011), Beaver Dam Wetland (BDW) (a pond drained in 2011), and a small (0.9-ha) commercial cranberry farm (Figure 1b). Discharge from Tidmarsh drains into Beaver Dam Brook, which flows northward before discharging to Bartlett Pond and then to Cape Cod Bay (Figure 1a). The watershed of Beaver Dam Brook at the outlet of Bartlett Pond is 21.5 km^2 in area, 62% of which drains through Tidmarsh (see Supporting Information S1).

3. Materials and Methods

3.1. Overview

As depicted in Figure 2, we conceptualize the hydrological processes of N transport at Tidmarsh based on the work of Hare et al. (2017) and Ballantine et al. (2020), conversations with landowners and former farm managers, and multiple field reconnaissance surveys. Hydrological fluxes of precipitation, surface water, and groundwater account for all abiotic N inputs to Tidmarsh. Atmospheric deposition of N, which has declined by about half since the 1990s (Bowen & Valiela, 2001), accounts for a N loading rate of about 5.1 kg N ha^{-1} yr^{-1} (see Supporting Information S1). Discrete surface water inputs of N include a pond (Fresh Pond), two wetlands (The Arm and Beaver Dam), and a small (0.9-ha) cranberry farm (Figure 2). Overland flow is generally a negligible surface water input of N due to sandy surface soils with high infiltration rates. Groundwater inputs of N are thought to occur both as upwelling matrix flow through the stream sediments and as groundwater springs, which were characterized by Hare et al. (2017) as diffuse lower-flux and discrete higher-flux discharges. During our field reconnaissance surveys, groundwater springs were observed visually in the stream channel, stream banks, and former drainage ditches, but were most prominent in the area where Beaver Dam Brook emerges as a headwater stream (Figures 1 and 2). The watershed for Tidmarsh is typical of regional land use: predominantly forested and developed (e.g., residential houses, commercial buildings, golf courses) land uses with minor amounts of agriculture.

Table 1
Measurement Types, Duration, Frequency, Locations Used for Three Research Objectives

Objective	Measurement	Date	Frequency	Site(s)
Nitrogen Loading Model	Q_{VA}	February 2016–November 2017	15-min	TID
Nitrogen Loading Model	Q_{FT}, v_{FT}, A_{FT}	February 2016–November 2017	Bi-monthly	TID
Nitrogen Loading Model	TN	August 2016–December 2017	Daily	TID
Nitrate Retention (F_{ret}, a^*_{TID})	Q_{FT}, v_{FT}, A_{FT}	February 2018–November 2019	Variable[a]	FP, AW, CB, TID, BDW
Nitrate Retention (F_{ret}, a^*_{TID})	NO_3^-	January 2020–January 2021	Weekly	FP, AW, CB, TID, BDW
Nominal Travel Time (NTT)	SpCond	December 2022–January 2024	Bi-monthly	TID
Nominal Travel Time (NTT)	Temperature	December 2022–January 2024	Bi-monthly	TID
Nominal Travel Time (NTT)	Q_{FT}, v_{FT}, A_{FT}	December 2022–January 2024	Bi-monthly	TID

Note. Q_{VA} = stream discharge measured with an acoustic-doppler area-velocity meter (Teledyne ISCO 2150), Q_{FT} = stream discharge measured with Sontek FlowTracker2, v_{FT} = stream velocity measured with Sontek FlowTracker2, A_{FT} = stream cross-sectional area measured with Sontek FlowTracker2, TN = concentration of total nitrogen (N) in surface water, NO_3^- = nitrate concentration measured in surface water or groundwater, SpCond = specific conductivity of surface water. See Figure 1 for site names and locations. [a]see Supporting Information S1.

Methods combined field measurements and modeling to quantify the transport and fate of N in Tidmarsh. Specifically, we collected geospatial and field data in pursuit of three research objectives (Table 1): (a) watershed-scale water quality modeling to determine sources of N to Tidmarsh, (b) weekly field measurements to quantify seasonal patterns of NO_3^- retention by Tidmarsh, and (c) bi-monthly injections of a conservate tracer to quantify solute residence time and its effect on NO_3^- retention by Tidmarsh. Briefly, objective (a) required a geospatial analysis of watershed land uses and a combination of standard field measurements of surface water discharge and NO_3^- concentration in surface water (see Supporting Information S1) to inform a steady-state water quality model (Valiela et al., 1997; see below). Objective (b) involved solving for NO_3^- retention in a stream-reach mass balance equation (see Section 3.6) with inputs from weekly measurements of surface water discharge and samples of surface water and groundwater analyzed for concentrations of NO_3^- (see Supporting Information S1). Finally, objective (c) consisted of deliberate injections of a conservative tracer (sodium chloride) to Beaver Dam Brook and measurement of the tracer breakthrough curve at a distance of 2.01 km downstream of the injection location (see Section 3.7), which was used to calculate solute travel time in surface water and to evaluate potential mechanisms underlying in-stream NO_3^- retention in the portion of Beaver Dam Brook draining Tidmarsh. For objective 3, field data on NO_3^- retention collected in 2020 were compared with solute travel time data collected in 2023, with the 3-year gap representing an additional source of uncertainty in our analysis. However, annual precipitation in Plymouth, Massachusetts increased by only 13% from 2020 to 2023, and although uncertainty associated with changes in watershed land uses, including wetland soils and vegetation, over the 3-year period was not well known, their effects on both solute travel time and in-stream retention of NO_3^- were assumed to be relatively minor.

3.2. Nutrient Analysis

Concentrations of nitrate (NO_3^-) were determined by colorimetry on a flow injection analyzer (Lachat, Quik-Chem 8500 Series 2) at the Pasture Systems and Watershed Management Research Unit's laboratory in East Wareham, Massachusetts. Concentrations of total (suspended particulate and dissolved) N were digested with an alkaline persulfate reagent in an autoclave (Patton & Kryskalla, 2003). Quantitative check standards (2.0 mg L^{-1} solutions of urea and nicotinic acid) indicated >97% recovery of organic and inorganic compounds. The method detection limit for NO_3^- was 5 μg N L^{-1} (APHA, 2005). Analytical uncertainty in NO_3^- was <10% based on >90% recovery of a 20 μg N L^{-1} check standard for NO_3^-.

3.3. Historical Land Use

The US Geological Survey's National Land Cover Database (NLCD) was used to quantify changes in watersheds land use changes between 2001 and 2019. We consolidated land uses into five categories: agriculture, natural vegetation, development, surface water, and barren. ArcMap v. 10.8 was used for geospatial analysis and the

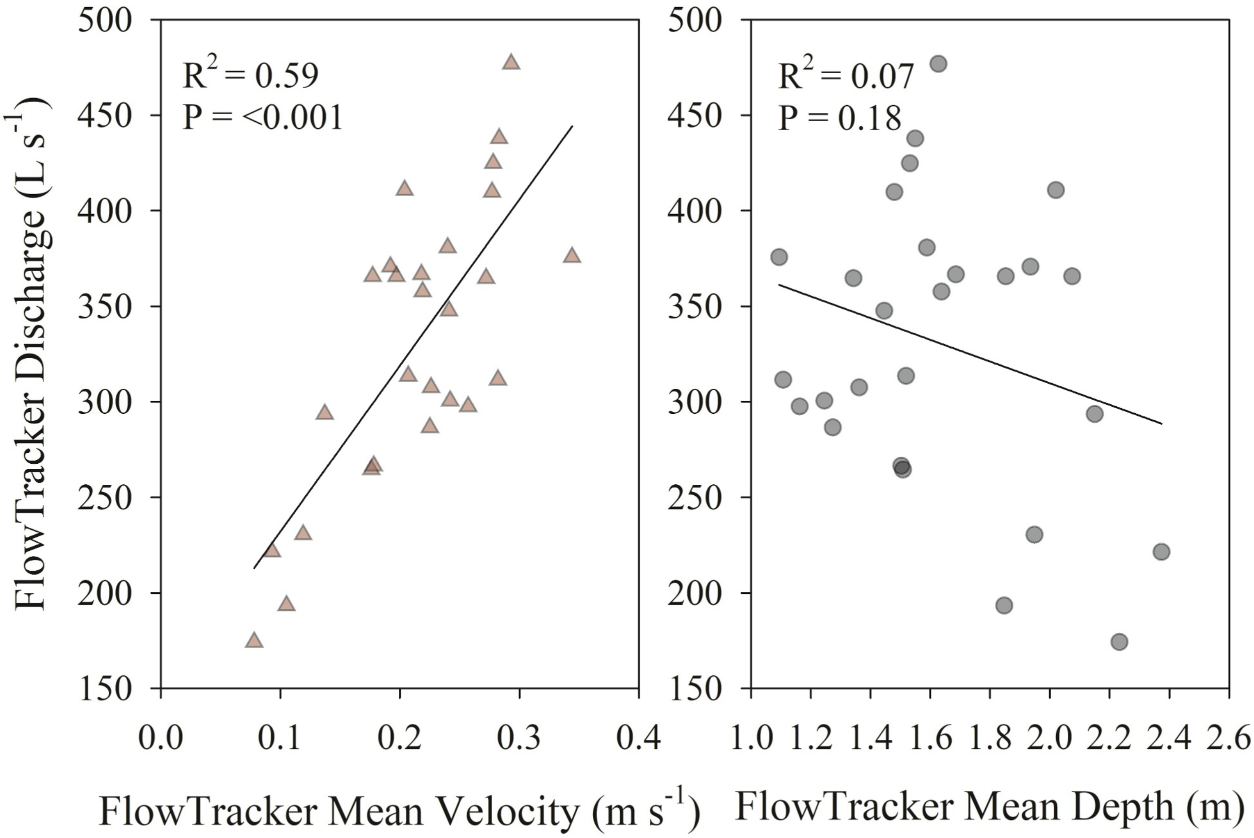

Figure 3. Stream discharge, mean velocity, and mean depth at the outlet of Tidmarsh (site TID) from February 2016 to December 2019. Measurements were made with a Sontek FlowTracker2 acoustic-doppler area-velocity meter.

Tabulate Area tool was used to calculate the total area of each land use for 2001, 2004, 2006, 2008, 2011, 2013, 2016, and 2019.

3.4. Index-Velocity Discharge

Stage-discharge relationships at the outlet of Tidmarsh were notably affected by downstream obstructions in flow that were presumed to be caused by aquatic vegetation in the stream channel (Cassan et al., 2015; Hearne & Armitage, 1993; Plew & Hoyle, 2017). When stage was regressed against discharge, a non-significant relationship was observed, contrasting with regressions between velocity and discharge (Figure 3). Based on these observations, the index-velocity method was employed to estimate discharge (Levesque & Oberg, 2012). Two separate rating curves were developed based on (a) continuous measurements of stage and discrete measurements of channel area ($R^2 = 0.64$, p value $= 0.003$) and (b) continuous measurements of velocity and discrete measurements of mean velocity ($R^2 = 0.57$, p value <0.001), using a Sontek FlowTracker2 area-velocity meter for discrete measurements and a Teledyne ISCO 2150 area-velocity meter for continuous measurements. Rating curves of channel area and mean velocity over time were used to calculate a continuous record of discharge for the 2017 water year. For the 2017 water year, the mean of the relative percent differences between measured and predicted discharges was 13%.

3.5. Nitrogen Loading Model

We applied the Waquoit Bay Nitrogen Loading Model (NLM; Valiela et al., 1997) to estimate the sources of N inputs to Beaver Dam Brook. The NLM is a steady-state model that tracks the movement of N in watersheds characterized by groundwater flow in coarse glacial sediments (Valiela et al., 1997, 2000). The model was initially developed for the Waquoit Bay on Cape Cod but has since been applied to coastal watersheds in Massachusetts (Kennedy & Hoekstra, 2021; Williamson et al., 2017), New York (Kinney & Valiela, 2011), New

Jersey (Bowen et al., 2007), Virginia (Giordano et al., 2011), and Nova Scotia, Canada (McIver et al., 2018). Application of the NLM requires information on (a) the watershed area, (b) the amount of N delivered to watersheds by atmospheric deposition, fertilizers, wastewater, and landfills, and (c) the proportion of N that is transmitted through the surface, vadose zone, aquifer, ponds, and wetlands of watersheds (Kennedy & Hoekstra, 2021; Valiela et al., 1997). Details on model development and application, including watershed area land uses, N loading rates, and N transmission coefficients, are provided in Supporting Information S1.

3.6. Wetland Retention of NO_3^-

We applied a mass-balance approach to quantify net NO_3^- retention (F_{ret}) as the difference between hydrologic inputs and outputs of NO_3^-:

$$F_{ret} = F_{sw} + F_{gw} - F_{TID}, \tag{1}$$

where F_{TID} is surface water discharge of NO_3^- from Tidmarsh, F_{sw} is the cumulative surface water input of NO_3^- to Tidmarsh, and F_{gw} is the net groundwater input of NO_3^- to Tidmarsh. We assume that soils represent a negligible source of NO_3^- to surface water, based on minor amounts of NO_3^- in soils at Tidmarsh (Ballantine et al., 2017) and low NO_3^- losses from active cranberry farms (Howes & Teal, 1995; Kennedy et al., 2020; Neill et al., 2017) and natural peatlands (Hemond, 1980, 1983; B. H. Hill et al., 2016; Verry & Timmons, 1982; Worrall et al., 2012).

Net NO_3^- retention, F_{ret}, can also be expressed (as a percent) in terms of amount of NO_3^- that is retained or attenuated by the wetland, as in Equation 1 (a^*_{TID}):

$$a^*_{TID} = 1 - \left(\frac{F_{TID}}{(F_{sw} + F_{gw})}\right)(100), \tag{2}$$

where $F_{TID} = Q_{TID}C_{TID}, F_{sw} = Q_{FP}C_{FP} + Q_{AW}C_{AW} + Q_{BDW}C_{BDW} + Q_{CB}C_{CB}$, $F_{gw} = (Q_{TID} - Q_{sw})\,C_{gw}$, and Q_{sw} is the cumulative surface water input to Tidmarsh from Fresh Pond (Q_{FP}), Arm Wetland (Q_{AW}), Beaver Dam Wetland (Q_{BDW}), a small commercial cranberry bog (Q_{CB}), and Q_{TID} is discharge from Tidmarsh (Q_{TID}). Values of Q were determined approximately monthly to calculate mean seasonal values of similar sample size ($N = 7$–11) from January 2020 to January 2021: spring (1 April to 30 June), summer (1 July to 30 September), autumn (1 October to 31 December), and winter (1 January to 31 March). Although these 3-month time periods do not correspond exactly with meteorological seasons, they capture a range of air temperatures with long-term (1991–2020) mean values of 20.2°C in the summer and 0.1°C in the winter (NOAA, 2022). Discharge measurements were made over short time intervals (~3 hr) when there was no precipitation to the wetland, evaporative losses were likely negligible, and changes in wetland storage were minor (<5%), based on stream stage at the outlet of Tidmarsh. Concentrations of NO_3^- in surface water were determined at the outlets of Tidmarsh (C_{TID}), Fresh Pond (C_{FP}), Arm Wetland (C_{AW}), Beaver Dam Wetland (C_{BDW}), and a small commercial cranberry bog (C_{CB}) on a weekly to bi-weekly basis from January 2020 to January 2021. We considered four sources of groundwater for C_{gw}: a shallow monitoring well, baseflow (winter discharge: 11.2 L s^{-1}) at the headwaters of Beaver Dam Brook, and two groundwater springs (referred to as SP1 and SP2) (Figure 1b), which we discuss further in Section 4.4. Daily NO_3^- fluxes were calculated as the product of seasonal discharge and daily NO_3^- concentration and used to calculate the mean daily NO_3^- flux for each season. Total NO_3^- export per season was determined as the mean daily flux times the number of days per season. Annual NO_3^- fluxes were determined by aggregating seasonal NO_3^- export.

3.7. Nominal Travel Time

A deliberate pulse injection of a conservative tracer was used to estimate nominal travel time (NTT) along a 2.01 km reach of Beaver Dam Brook, which extended from sampling sites BDW to TID (Figure 1). The conservative tracer (22.7 kg of NaCl mixed with stream water) was added at the upstream end of the reach (site BDW) seven times roughly every other month between December 2022 and January 2024. At the downstream end of the reach (site TID), specific conductivity (S) breakthrough curves were quantified using an EXO3 multiparameter sonde (YSI, Yellow Springs, Ohio) programmed to automatically record S on 30-s time intervals. Values of NTT were determined as the time required for S to reach peak concentration (Baker & Webster, 2017). Repeated pulses injections were conducted to evaluate NTT as a factor controlling temporal variation in F_{ret} and a^{*}_{TID}. In addition, stream discharge, mean velocity, channel area, and stream temperature were measured concurrently with a SonTek FlowTracker2 acoustic Doppler velocity meter (ADV) at the downstream end of the reach (i.e., site TID).

3.8. Uncertainty

Uncertainty was estimated using standard methods of propagation of error (Kline, 1985) in the measurements of discharge and N concentration in surface water and groundwater. Two estimates of uncertainty are reported: (a) uncertainty in the TN load in surface water at the outlet of Tidmarsh for the 2017 water year, using the index-velocity method for discharge and daily-composite samples of surface water for analysis of TN, and (b) uncertainty in daily values of F_{RET} for NO_3^- from Jan. 2020 to Jan. 2021 ($N = 38$), using area-velocity discharge and grab-based samples of surface water and groundwater. Uncertainty in discharge using the index-velocity method was 13%, based on the difference between the means of measured and predicted discharge. Analytical uncertainty in NO_3^- of 2.2% was calculated as the mean relative percent difference from a 20 μg N L^{-1} analytical standard. For TN, we used the analytical uncertainty in NO_3^- plus an additional 3% based on uncertainty in the digestion (i.e., 97% recovery of check standards of nicotinic acid and urea). For area-velocity discharge, we assumed two components of uncertainty: the number of verticals in the measurements and accuracy in the calibration. We used a minimum of 20 vertical for area-velocity discharge with the associated uncertainty of 2.5% (Huhta & Sloat, 2007) and assumed the calibration uncertainty of 1% (Huhta & Sloat, 2007). For the 2017 water year, uncertainty in TN export by surface water was 14%. Uncertainty in daily values of F_{RET} ranged from 0.3 to 1.1 kg N d^{-1}, with mean uncertainty of 26% (median uncertainty, 18%).

4. Results

4.1. Watershed Hydrology and Nitrogen Fluxes

Discharge from Tidmarsh (Q_{TID}) ranged from 76 to 859 L s^{-1} (mean ± SD: 329 ± 95 L s^{-1}) for the 2017 water year (1 October 2016 to 30 September 2017; Figure 4). Mean values of seasonal Q_{TID} varied by 22% (as the coefficient of variation about the mean), ranging from 229 L s^{-1} in summer to 403 L s^{-1} in autumn. Annual Q_{TID} normalized by the watershed area was 781 mm yr^{-1}, which was within 12% of the long-term (1967–2006) mean groundwater recharge rate of 686 mm yr^{-1} for the PKCD aquifer (Masterson et al., 2009).

Total N (TN) flux in surface water from the watershed ranged from 1.8 to 28.8 kg N d^{-1} (mean ± SD: 10.1 ± 5.2 kg N d^{-1}) for the 2017 water year. Temporal variation in TN flux was more strongly correlated with TN concentration than Q_{TID} (Figure 5), owing in part to minor temporal variation in Q_{TID} (Figure 4). Seasonally, a similar pattern emerged with TN concentration driving variation in TN flux, with greater variation in mean seasonal values of TN concentration and TN flux (30% and 40%, respectively) than in Q_{TID} (22%). Annual TN export from the watershed was 3,669 kg N yr^{-1} (Table 2), which was determined using continuous measurements of discharge and daily concentrations of TN in surface water (e.g., Figure 4). Weekly surface water samples ($N = 51$) were analyzed for NO_3^-, which varied from 0.007 to 0.37 mg N L^{-1} (mean ± SD: 0.16 ± 0.11 mg N L^{-1}). Comparing weekly mass fluxes of TN and NO_3^-, we estimated that NO_3^- comprised 51% of the TN load in surface water, or 1,865 kg N yr^{-1} of NO_3^- was exported by surface water from the watershed for the 2017 water year.

4.2. Historical Changes in Watershed Land Use

From 2001 to 2019, natural vegetation and development accounted for 87%–89% of the land use within the watershed for Tidmarsh, but the relative proportions of these land uses changed rapidly over the past 19 years (Figure 6). Declines in natural vegetation coincided with increases in development (slope = −1.12, $R^2 = 0.997$), primarily in the southwest portion of the watershed where two golf courses and a residential development were constructed as part of the Pinehills village in ∼2000. Prior to 2000, the watershed was almost entirely forested, based on visual observations of aerial photographs from Google Earth®. By 2019, development was the dominant land use, representing 50% of the watershed area.

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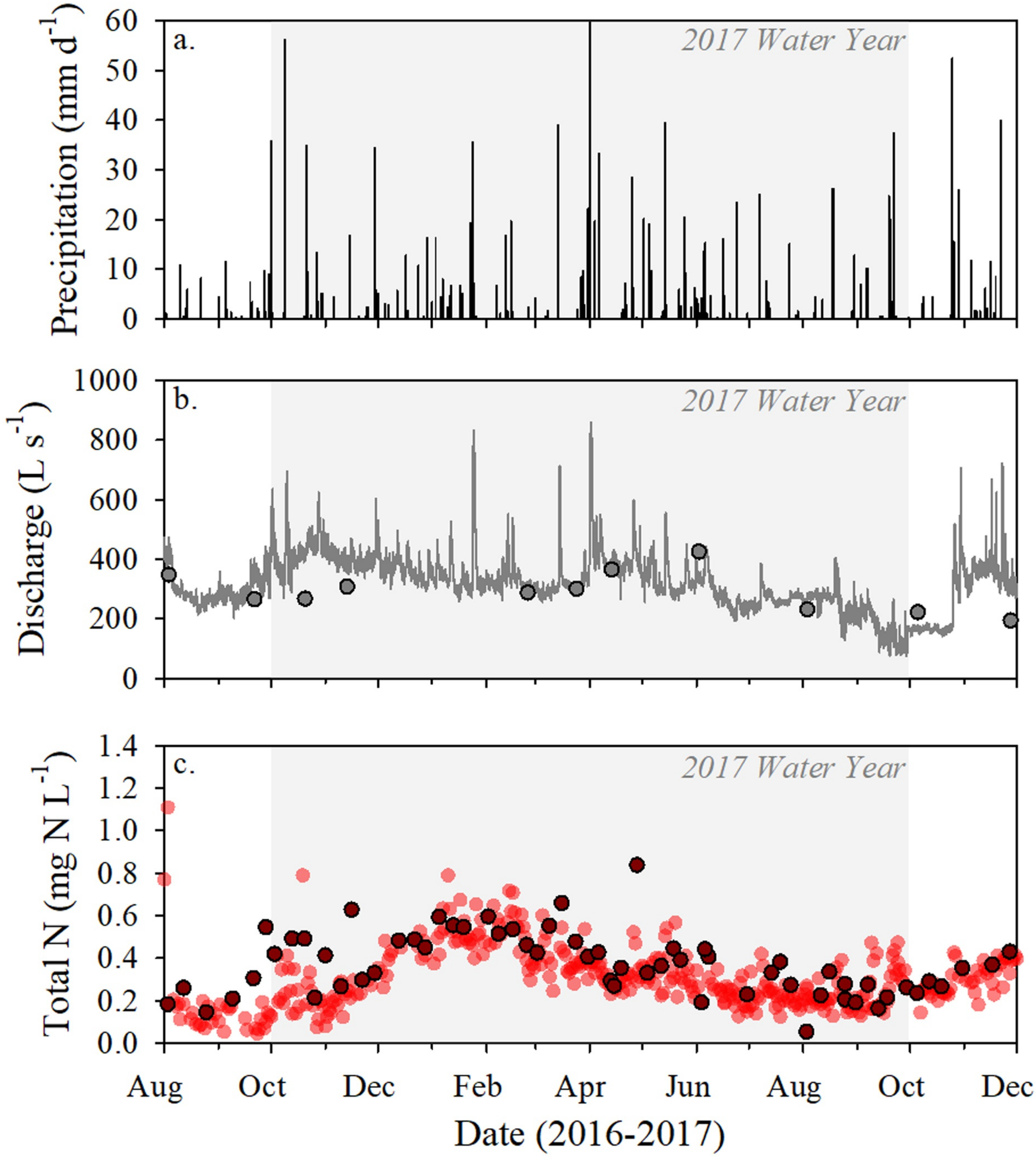

Figure 4. Daily precipitation at Plymouth (a). Measured discharge using the area-velocity method (gray circle) and estimated using the index-velocity method (gray line) (b), and total N (TN) concentrations in surface water collected as discrete grab samples (dark red circle) and daily composites using Isco automatic samplers (light red circle) (c) at the outlet of Tidmarsh (site TID; see Figure 1). Data collected from August 2016 to December 2017 with the 2017 water year shaded in gray.

4.3. Watershed-Scale Water Quality Modeling

The total amount of N delivered to the watershed from wastewater, fertilizers, atmospheric deposition, and a landfill was 32,475 kg N yr^{-1}, 79% of which was removed at surface, in the vadose zone, and in the aquifer of the watershed. The amount of N that flowed directly into or through surface water bodies connected to Tidmarsh was 6,668 kg N yr^{-1}, which we refer to as the watershed-attenuated export (M_{TID}^{WA}) (Table 2), as it omits N attenuation through ponds and wetlands. Of this amount, Fresh Pond removed 2%, BDW removed 28%, and the AW removed

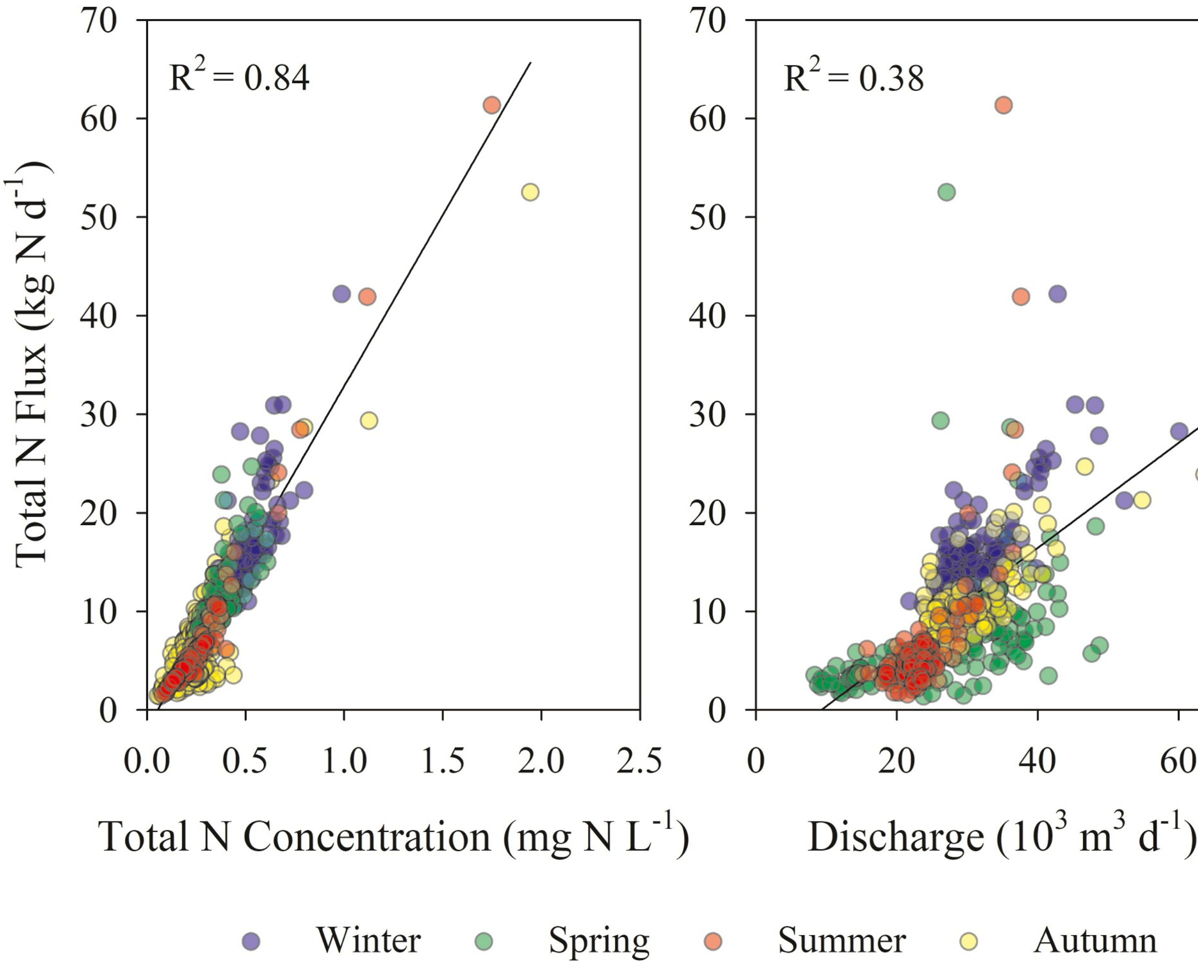

Figure 5. Daily discharge and total N (TN) concentration in surface water from Tidmarsh (site TID; see Figure 1) grouped by season from August 2016 to December 2017. Total N flux calculated as product of discharge and TN concentration.

6% (see Supporting Information S1). We estimate that Tidmarsh removed an additional 9% of watershed inputs of N, based on 28% retention of N moving through Tidmarsh (see Section 4.4 and Supporting Information S1).

Accounting for attenuation in ponds and wetlands, the total N load in surface water for Beaver Dam Brook (M^{A}_{TID}) was 3,669 kg N yr^{-1} (Table 2). Turf grass and septic systems were the major source of N to Beaver Dam Brook, providing 39% and 28% of M^{A}_{TID}, respectively. The Pinehills community was also a significant source of N, contributing 14% from the wastewater treatment facility, in addition to contributions from turf grass fertilizers. The Manomet-Plymouth landfill accounted for 7% and atmospheric deposition 13% of M^{A}_{TID}. The contribution from agricultural fertilizers was relatively minor (<1%) compared to wastewater or turf-grass fertilizers. This is generally the case for most watersheds in southeastern Massachusetts, although the Weweantic River, Sippican River, and Wareham River watersheds are notable exceptions (Kennedy & Hoekstra, 2021; Williamson et al., 2017). When Tidmarsh was an active cranberry farm, it was primarily cultivated with the native variety "Early Black." Assuming a mean fertilizer rate for "Early Black" cranberries of 31 kg N ha^{-1} yr^{-1} (Kennedy et al., 2026), N transmission coefficients of 0.61 for flowthrough-farms (Kennedy & Hoekstra, 2021) and 0.65 for aquifer sediments (Valiela et al., 1997), Tidmarsh would have contributed 685 kg N yr^{-1} to Beaver Dam Brook, or roughly half the N contributed from turf grass fertilizers.

4.4. Nitrogen Mass Balance

Mean annual Q_{sw} was 231 L s^{-1} and comprised 69% of Q_{TID}, with discharge from BDW accounting for 77% of Q_{sw} (Figure 7). Q_{gw} averaged 102 L s^{-1}, and represented 31% of Q_{TID}. Seasonal variation in Q_{TID} and Q_{sw}

Table 2
Modeled N Loads for Beaver Dam Brook at the Outlet of Tidmarsh (Percent of Total N Load in Parentheses)

Source	Unattenuated N Load (kg N yr^{-1})	Watershed-attenuated N Load (kg N yr^{-1})	Attenuated N Load (kg N yr^{-1})
Fertilizer			
Cranberry farms	149	30	17 (0.4)
Turf (mowed grass)	16,703	2,580	1,419 (39)
Wastewater			
Septic systems	7,208	1,855	1,021 (28)
Treatment facility	1,670	904	497 (14)
Landfill	–	445	245 (7)
Atmospheric deposition			
Watershed	6,744	854	470 (13)
Total	32,475	6,668	3,669[a]

Note. Unattenuated N loads include N inputs to the watershed from fertilizers, wastewater, atmospheric deposition, and a landfill. Watershed-attenuated N loads include unattenuated N loads minus watershed attenuation except N attenuation in ponds and wetlands. Attenuated N loads include watershed-attenuated N loads minus N attenuation in ponds and wetlands. [a]Calculated based on field measurements of continuous discharge and composite samples of TN concentration in surface water from 1 October 2016 to 30 September 2017.

was relatively minor, ranging from 14% to 21% (Figure 7), as was seasonal variation in Q_{gw}, which varied by 9%. Despite the minor seasonal variations in hydrology, significant seasonal variation was observed in concentrations of NO_3^- in surface water, with values of C_{TID} varying by 44% compared with 14% for Q_{TID}. Seasonal variation in C_{BDW} and C_{AW} was 26% and 18%, respectively. C_{AW} was higher than both C_{BDW} and C_{TID} in all seasons except for autumn (Figure 8d), but the AW was a relatively minor hydrologic input to Tidmarsh (i.e., discharge from BDW was 4–5 times than that from the AW). In the spring and autumn, C_{BDW} and C_{TID} were not statistically different (Figures 8b–8d), but C_{TID} was 25% lower than C_{BDW} in the summer and 31% higher than C_{BDW} in the winter (Figure 8c).

Accordingly, relationships between C_{BDW} and C_{TID} were used to evaluate four potential sources that best represented the flow-weighed concentration of NO_3^- in groundwater to Tidmarsh, C_{gw}: a shallow groundwater well, baseflow at the headwaters of Beaver Dam Brook (HW; winter discharge: 11.2 L s^{-1}), and two groundwater springs (SP1 and SP2) (Figure 1). Concentrations of NO_3^- in shallow groundwater ranged from <0.005 to 0.019 mg N L^{-1}, which were 5–19 times lower than the minimum value of C_{TID}. Concentrations of NO_3^- in Tidmarsh baseflow (at its headwaters) (y) and groundwater spring 1 (x) were strongly correlated ($y = 0.87x + 0.02$, $R^2 = 0.78$), but were less than the maximum value of C_{TID} (0.44 mg N L^{-1}). In contrast, groundwater spring 2 exhibited the highest NO_3^- concentrations in either groundwater or surface water (Figure 8), which led us to conclude that it was most representative of C_{gw}. Notably, we observed significant (31%) seasonal variation in C_{gw}, with the highest concentrations in the winter, and the lowest concentrations in the summer.

Seasonal values of F_{ret} were the highest in the summer, when Tidmarsh retained 361 kg N of NO_3^- (Figure 9). Tidmarsh received inputs of NO_3^- from groundwater and surface water that totaled 3,641 kg N yr^{-1}, of which

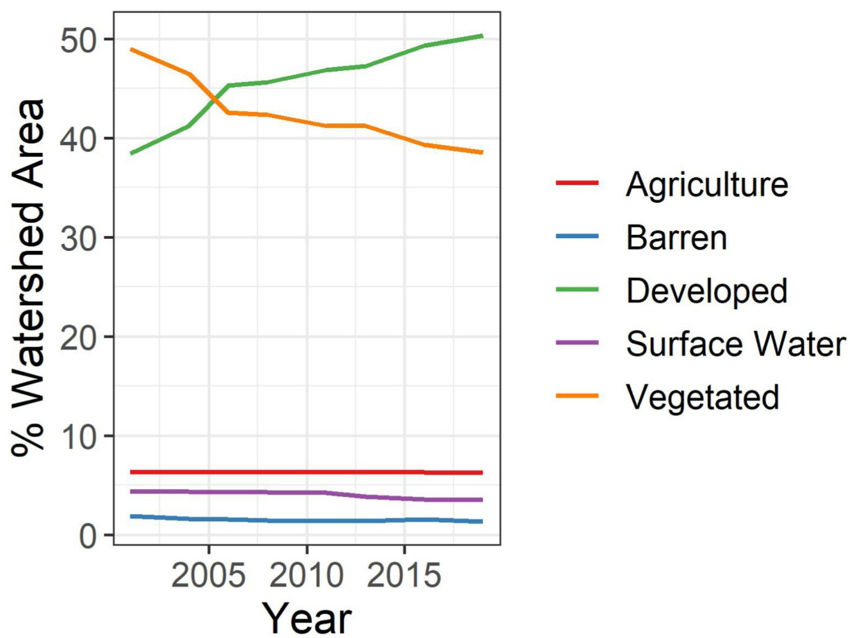

Figure 6. Historical land use change in the watershed of Tidmarsh Farms East ("Tidmarsh"), Plymouth, Massachusetts.

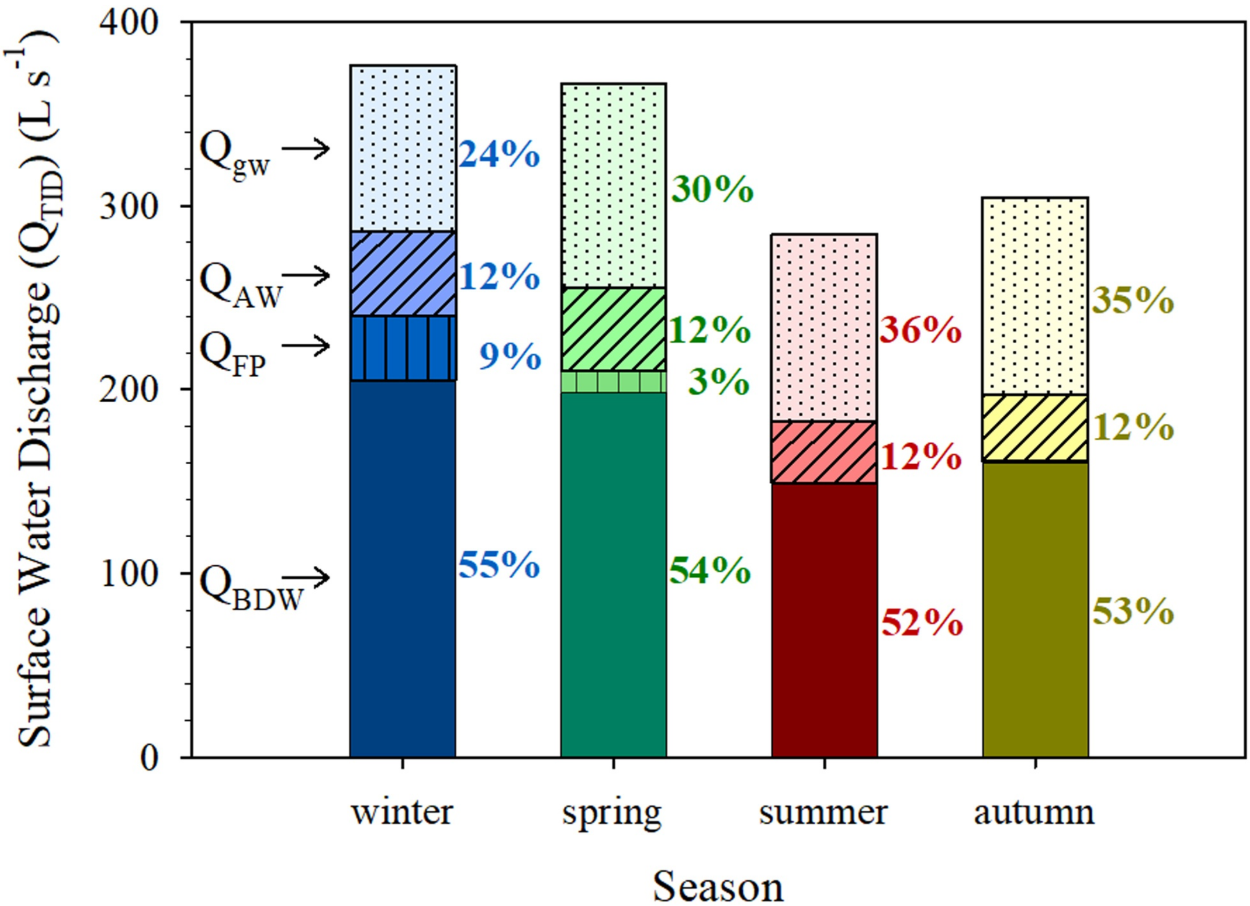

Figure 7. Discharge from Tidmarsh (Q_{TID}) and hydrologic inputs of groundwater (Q_{gw}) and surface water from the Arm Wetland (Q_{AW}), Fresh Pond (Q_{FP}), and Beaver Dam Wetland (Q_{BDW}). Discharge for the active cranberry bog to the southwest of Tidmarsh (not shown) was 0.1 L s^{-1}. Because Fresh Pond was not flowing between July and January, discharge was set to zero for this period. Seasons correspond to the following dates: spring (1 April to 30 June), summer (1 July to 30 September), autumn (1 October to 31 December), winter (1 January to 31 March) from 2020 to 2021.

2,612 kg N yr^{-1} was exported by surface water from Tidmarsh. The net amount of NO_3^- retained by Tidmarsh, F_{ret}, was 1,029 kg N yr^{-1} (a^*_{TID} = 28%), or 18.4 kg N ha^{-1} yr^{-1}. Modeled F_{ret} (460 kg N yr^{-1}) was ~2.2 times lower than measured F_{ret}, in part because measured F_{ret} represents attenuation of all hydrologic inputs rather than just surface water, and because modeled estimates are not meant to be interpreted as well-defined values of F_{ret} (Valiela et al., 1997).

4.5. Stream Solute Travel Time

Nominal travel time (NTT) of surface water transmitted through Tidmarsh was evaluated using pulse injections of a conservative tracer (NaCl). Values of NTT ranged from 4.7 hr in March to 15.7 hr in September, with the highest values of NTT observed in the summer (Table 3). The seasonal pattern of increasing NTT from spring to summer followed by gradual declines in NTT from fall to winter was consistent with aquatic vegetation growth and decay cycles and their effects on stream velocity (Green, 2006; Pitlo & Dawson, 1990). For instance, values of NTT were negatively correlated with mean stream velocity (R^2 = 0.56, P = 0.05), but positively correlated with stream channel area (R^2 = 0.40, P = 0.13) (Table 3). The increase in NTT with wetted channel area is likely due to the presence of extensive aquatic vegetation in the summer months, which caused obstructions to flow that increased water stage while reducing mean flow velocities (Green, 2006; Pitlo & Dawson, 1990).

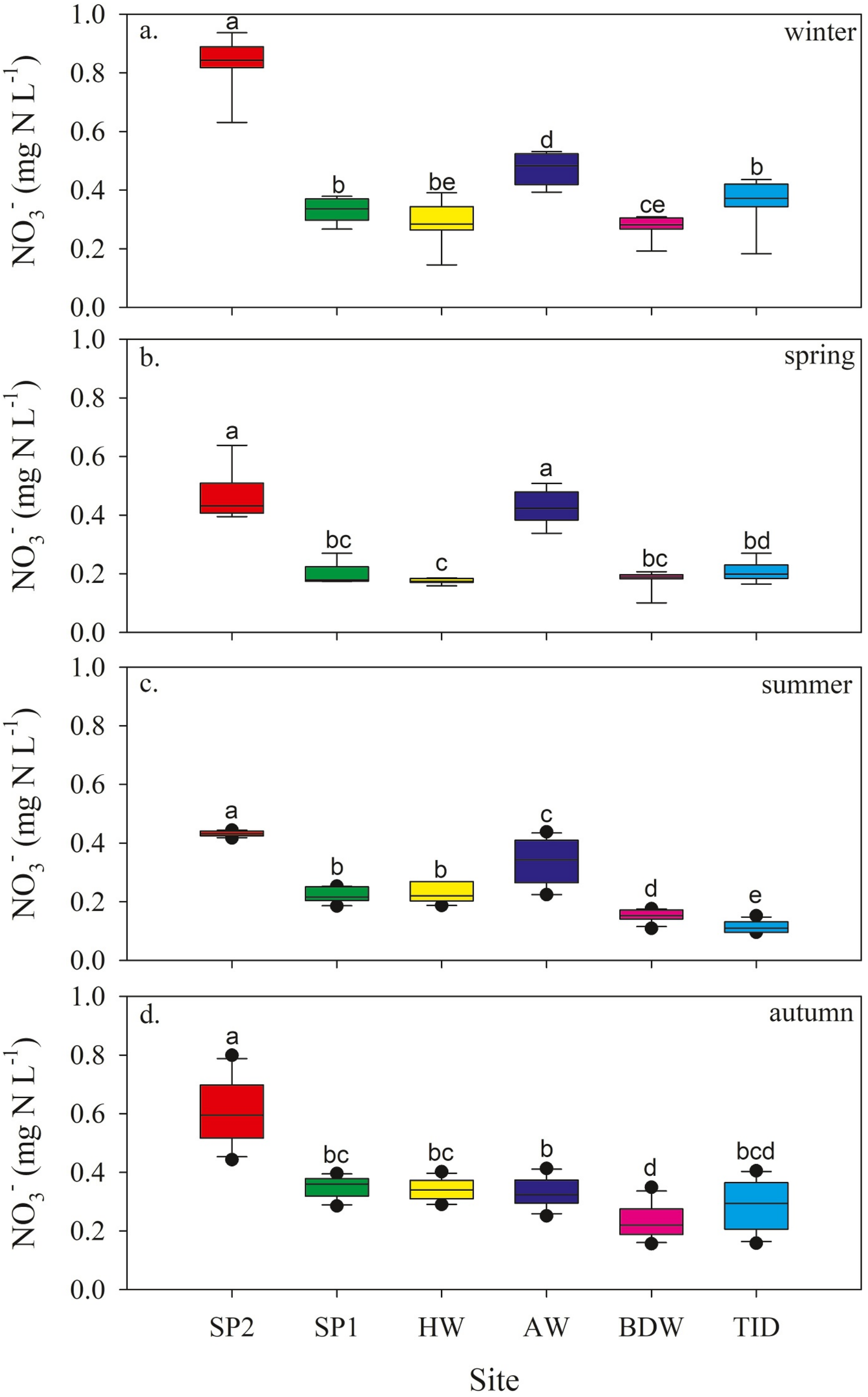

Figure 8. Box plots of mean nitrate (NO_3^-) concentrations in groundwater (Spring 2, SP2; Spring 1, SP1; Headwaters, HW) and surface water (Arm Wetland, AW; Beaver Dam Wetland, BDW; Tidmarsh, TID) ($N = 7–11$ per season) for the winter (a), spring (b), summer (c), and autumn (d). Seasons correspond to the following dates: spring (1 April to 30 June), summer (1 July to 30 September), autumn (1 October to 31 December), winter (1 January to 31 March) from 2020 to 2021. Data not shown for the outlet of Fresh Pond, which was dry between July and January and the active cranberry bog, which had very low discharges and NO_3^- concentrations. Boxes not connected by the same letter are significantly different at 95% confidence (two-tailed *t* test; *p* value ≤0.05).

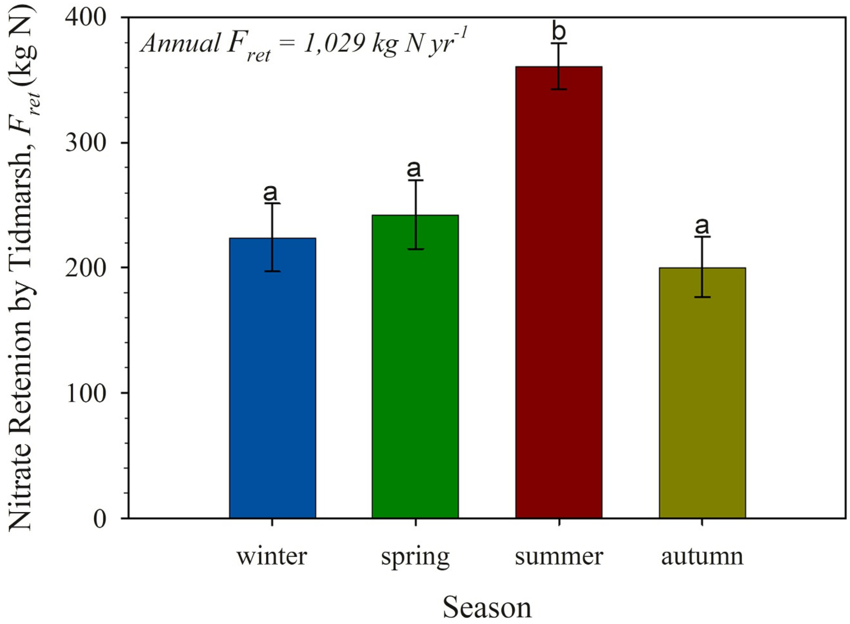

Figure 9. Seasonality of NO_3^- retention (F_{ret}) for Tidmarsh between January 2020 and January 2021 ($N = 7$–11 per season). Seasons correspond to the following dates: spring (1 April to 30 June), summer (1 July to 30 September), autumn (1 October to 31 December), winter (1 January to 31 March). Bars not connected by the same letter are significantly different at 95% confidence (two-tailed t test; p value ≤ 0.05). Error bars represent the standard error about the mean.

5. Discussion

5.1. Spatiotemporal Dynamics of Nitrate Concentration in Surface Water

We observed seasonal patterns of longitudinal declines in NO_3^- concentrations along Beaver Dam Brook that were consistent with biologically medicated in-stream uptake of NO_3^-. For instance, values of C_{BDW} and C_{TID} were not statistically different (Figures 8b–8d) in the spring and autumn but in the summer, C_{TID} was 25% lower than C_{BDW} (Figure 8c), suggesting increases in N retention downstream (Mulholland et al., 1995) due to temperature effects on in-stream N removal processes, such as plant uptake and denitrification, or to increases in transient storage and possibly sedimentation and denitrification at low discharges (Holland et al., 2004). Longitudinal declines in NO_3^- concentrations in Beaver Dam Brook in the summer contrasted with increasing concentrations of NO_3^- in the winter (Figures 8a–8c). However, in-stream NO_3^- production is generally limited by low nitrification rates in the winter (Scheible et al., 1993; Stark, 1996; Starry et al., 2005), possibly pointing to higher winter concentrations and inputs of NO_3^- from groundwater, which accounted for 31% of streamflow on an annual basis. Notably, we observed similar patterns of seasonal variation in NO_3^- concentration in stream baseflow and groundwater (spring 1; Figures 8a–8c), which was consistent with seasonal variation of NO_3^- in coastal groundwater on Cape Cod (Valiela & Costa, 1988; Valiela et al., 1990). These results may suggest that the seasonality in NO_3^- concentrations in groundwater (C_{gw}) is caused by denitrification that occurs either in shallow streambed sediments (Valiela et al., 1990) or along the edge of the discharge area of the peatland (Böhlke et al., 2002).

Table 3
Stream Characteristics of Pulse Injections to Quantify Nominal Travel Times (NTT) Along a 2.01 km Reach of Beaver Dam Brook (Figure 1)

Date	Q_{TID} (L s^{-1})	A (m^2)	v (m s^{-1})	T (°C)	NTT (hr)
13 December 2022	260	1.49	0.18	4.6	7.3
8 March 2023	292	1.54	0.19	8.4	4.7
9 May 2023	256	1.59	0.16	17.7	5.1
12 July 2023	239	1.93	0.12	24.7	9.1
8 September 2023	225	1.85	0.12	21.2	15.7
8 November 2023	255	1.71	0.15	10.2	10.8
2 January 2024	370	1.72	0.22	6.0	5.7

Note. All pulse injections consisted of 22.7 kg of NaCl mixed with stream water. Values of discharge (Q_{TID}), cross-sectional stream area (A), mean stream velocity (v), and mean stream temperature (T) were measured at the downstream end of the stream reach of Beaver Dam Brook (i.e., site TID).

5.2. Nitrate Retention by Wetlands

Our results suggest that the amount of N removed by Tidmarsh ($a^*_{TID} = 28\%$) may be considerably less than previous N removal estimates assumed for natural wetlands in watershed-scale N loading models used across eastern North America (Bowen et al., 2007; Giordano et al., 2011; Kinney & Valiela, 2011; McIver et al., 2018; Valiela et al., 1997). However, the reported N retention in this study is within the range of other natural, restored, and constructed wetlands. Saunders and Kalff (2001) found that wetlands retain an average of 64% of N inputs compared with 34% retention for lakes and 2% retention for rivers, but N retention rates for wetlands varied widely, ranging from 1% to 100%. Similarly, Jordan et al. (2011) report average N removal of 43% from a meta-analysis of 500 wetland samples, with a range of 35%–62%. Several factors may account for the relatively low values of N retention, including shorter residence times (Holland et al., 2004), channelized surface flow, and/or high N loading rates, which may limit N removal efficiency (Saunders & Kalff, 2001; Walton et al., 2020).

The main hydrologic control on N retention in aquatic systems is residence time (Mitsch & Gosselink, 1993), which promotes N retention processes such as denitrification and sedimentation (Saunders & Kalff, 2001; Seitzinger et al., 2006; Svendsen & Kronvang, 1993). With respect to amounts of N removed (i.e., F_{ret}) rather than N removal efficiency (i.e., a^*_{TID}), we report F_{ret} equal to 18.4 kg N ha^{-1} yr^{-1}, which was lower than the mean NO_3^- retention rate of 156 kg N ha^{-1} yr^{-1} for natural and created wetlands, but comparable to the low-end of the range for river-fed low flow wetlands (30 kg N ha^{-1} yr^{-1}; Mitsch et al., 2014). Often, values of F_{ret} are directly proportional to N loading to wetlands (Jordan et al., 2011; Poe et al., 2003; Saunders & Kalff, 2001). Though greater overall N retention may occur with high N loading, many studies have documented an inverse relationship between N removal efficiency and N loading rates across natural, restored, and constructed wetlands around the world (Hunter et al., 2018; Land et al., 2016; Walton et al., 2020). A recent meta-analysis of riparian buffer wetlands indicated that variation in N retention efficiency is also affected by the nature of influent water (Walton et al., 2020). Walton et al. (2020) found that removal efficiencies of NO_3^- and TN tend to be above 50% in groundwater ($N = 7$) and precipitation-fed ($N = 9$) wetlands, while upland surface runoff ($N = 19$) and stream-fed wetlands ($N = 22$) exhibit NO_3^- and TN removal efficiencies below 50%.

Along these lines, another potential factor affecting F_{ret} was a 28% increase in C_{TID} (and presumably C_{gw}) between 2017 and 2020, which is consistent with observations of increasing F_{ret} with N loading to wetlands (Fleischer & Stibe, 1991; Saunders & Kalff, 2001). At Tidmarsh, the cause for increasing C_{TID} is not exactly known, but evidence to implicate peatland restoration is sparse, as NO_3^- production is often limited in natural and restored peatlands (Stark & Hart, 1997; Westbrook & Devito, 2004; Williams & Wheatley, 1988) while the conditions for denitrification are often enhanced (Ballantine et al., 2017). The fraction of cranberry agriculture in the watershed has remained relatively constant between 2001 and 2019 (Figure 6), and even if it had increased, cranberry agriculture would be an unlikely source of NO_3^- (Kennedy et al., 2020). Instead, increases in C_{TID} are consistent with the effects of development on water quality: (a) more N is delivered to the watershed from wastewater and turf fertilizers and (b) less N is removed from the watershed by natural vegetation. As a result, the 30% increase in development over the past 19 years (Figure 6) has led to both increases in watershed inputs of N and decreases in N attenuation in the watershed. Consequently, not only have N inputs to the watershed increased over time, but so too has the fraction of N that is transmitted from the watershed to groundwater and surface water.

By comparison, similar increases in development occurred in the watershed for Waquoit Bay on Cape Cod between 1990 and 2014 but were offset by declines in atmospheric deposition (Lloret et al., 2022; Valiela et al., 2016). Unlike Waquoit Bay, Tidmarsh drains a relatively small watershed (roughly four times smaller than the Waquoit Bay watershed) that was almost completely forested ∼20 years ago. As a result, the response in surface water quality to watershed land use changes was more pronounced at Tidmarsh, where development (per parcel) represented a greater proportion of N inputs to the watershed than for Waquoit Bay. More broadly, the urbanization of forested land is consistent with historical (1973–2000) changes in land use for the Northeastern Coastal Zone (NECZ) ecoregion, which includes central to coastal portions of Maine, New Hampshire, Vermont, Massachusetts, Rhode Island, Connecticut, New York, and New Jersey (Omernik, 1987; U.S. Environmental Protection Agency, 1999). Within the NECZ ecoregion, forest and developed land uses experienced the greatest change between 1973 and 2000, with development increasing by 4.0% area and forest decreasing by 3.7% (Sayler et al., 2016).

5.3. Mechanistic Controls on Nitrate Retention

Temporal variation in NO_3^- removal (a^*_{TID} ; see Equation 2) ranged from 7% to 67%, offering an opportunity to explore relationships between a^*_{TID} and (a) hydrological inputs of N, (b) NTT, and (c) stream temperature (Figure 10). Because of differences in sampling scaling (i.e., a^*_{TID} was determined roughly weekly, while stream temperature and NTT were determined bimonthly), we calculated mean values of a^*_{TID} on a bimonthly basis.

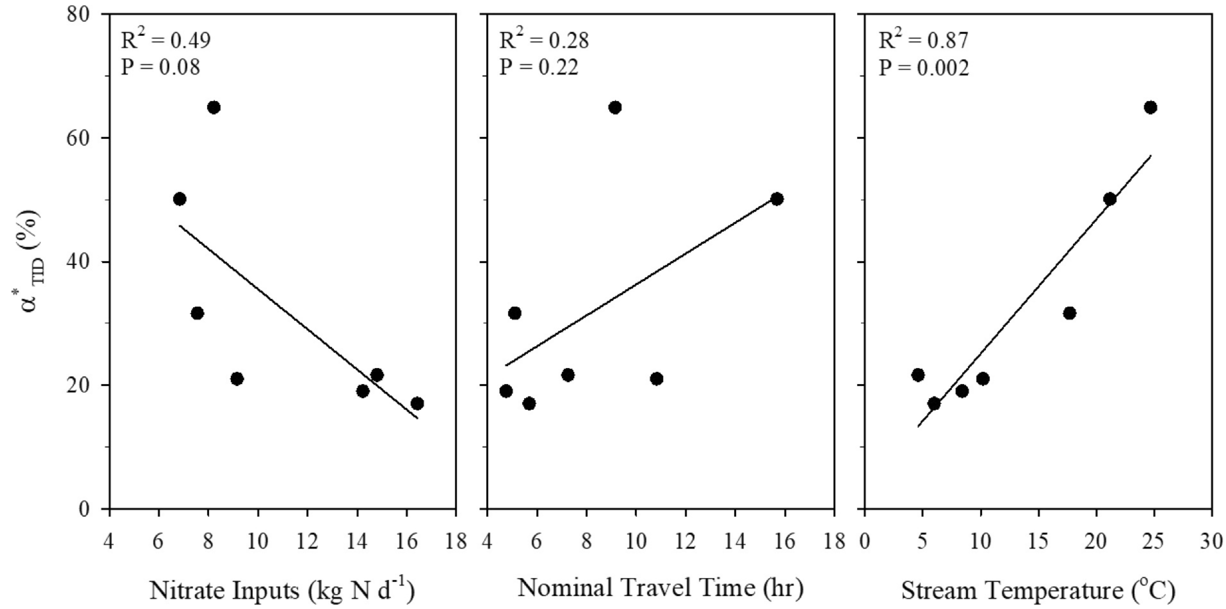

Figure 10. Controls on percent nitrate retention in the restored wetland (a^*_{TID} ; see Equation 2) from January 2020 to January 2021. Nominal travel times (NTT) were measured along a 2.01 km reach from sites BDW to TID and stream temperature was measured at site TID from December 2022 to January 2024. For comparison, values of a^*_{TID} and nitrate inputs ($F_{sw} + F_{gw}$) were calculated as mean values using a 30-day window around sampling dates for NTT and stream temperature (see Table 3 for dates), and were measured in the same season but different year as NTT and stream temperature.

Values of a^*_{TID} were inversely correlated with NO_3^- inputs (i.e., F_{sw} and F_{gw}) (Figure 10), which was consistent with previous research (Hunter et al., 2018; Land et al., 2016; Walton et al., 2020). However, F_{ret} was not correlated with NO_3^- inputs, which may simply be an artifact of the relatively low variation in NO_3^- inputs to Tidmarsh, at least compared to studies that evaluated relationships across wetlands (Saunders & Kalff, 2001; Saunders & Kalff, 2001). Although natural restoration of stream channels can increase residence times that enhance N uptake and denitrification (Bukaveckas, 2007; Ensign & Doyle, 2005; Kaushal et al., 2008; Klocker et al., 2009), we did not observe a strong correlation between NTT and a^*_{TID} , possibly due, in part, to the relatively young age (7 years, referring to the time since restoration activites were completed; Ballantine et al., 2020) of the resorted wetland. We expect that as the wetland matures, NTT will increase and exert a greater control on a^*_{TID} , perhaps by enhancing residence time in the hyporheic zone, a potential hot spot for denitrification in fluvial systems (McClain et al., 2003). Indeed, as much as 97% of whole-stream denitrification may occur in the hyporehic zone (Bohlke et al., 2009) with longer residence times increasing denitrification rates (Zarnetske et al., 2011). In this regard, as the wetland matures and function is restored, increases in NTT may promote greater exchange with floodplain sediments and storage in the hyporheic zone, both of which would enhance NO_3^- removal.

However, at its current state of establishment, values of a^*_{TID} for the restored wetland were most strongly correlated with stream temperature, implying environmental controls on biologically-mediated processes such as denitrification and plant uptake (e.g., Birgand et al., 2007). These findings are broadly consistent with N removal by macrophyte biomass, vegetative N uptake, and microbially-mediated N removal processes that are enhanced by warmer temperatures in the summer (Clarke, 2002; Lin et al., 2021; McGuire et al., 1992; Vymazal, 2007). Temperature effects on whole denitrification rates in streams have been studied extensively in laboratory experiments (Herman et al., 2007; Opdyke & David, 2007; Pfenning & McMahon, 1996), but can be confounded by other variables in field studies (Bohlke et al., 2009). For instance, Pfenning and McMahon (1996) report a 77% reduction in denitrification rates in streambed sediments when incubation temperatures were lowered from 22 to 4°C. Over approximately the same temperature range (21.2–4.6°C), we find a lower but comparable reduction in a^*_{TID} of 57%, with values of a^*_{TID} decreasing from 50% to 22%. Although the mechanism by which temperature reduces a^*_{TID} was not determined, the observed relationship of decreasing values of a^*_{TID} with colder temperatures is consistent with temperature effects on both biological uptake in surface water and denitrification in benthic sediments.

5.4. Implications for Regional Water Quality

On Cape Cod, it is well established that the primary source of N pollution is wastewater from septic systems, but centralized sewering is often viewed as impractical due to the immense financial burden in capital costs ($4.2–6.2 billion; Cape Cod Commission, 2013) placed on residents of relatively rural towns. Consequently, N management strategies that serve as alternatives to sewering, such as restoration of wetlands on retired cranberry farms, are eagerly being pursued by researchers and practitioners alike. Due to a variety of factors, including economics of the cranberry industry (MDAR, 2016), the harvested acreage in Massachusetts has declined by 12% (648 ha) over the past two decades (USDA-NASS, 2024), leaving many formerly cultivated wetlands available for possible ecological restoration (Hoekstra et al., 2020). Recognizing this opportunity, the state of Massachusetts's Division of Ecological Restoration (DER) developed the Cranberry Bog Restoration Program in 2017 to provide a "green exit strategy" for growers interested in retiring their farm operation. Today, DER's Cranberry Bog Restoration Program has supported the ecological restoration of 21 retired cranberry farms in Massachusetts (https://projects.livingobservatory.org/projects/der-monitoring).

Restoration of wetlands on retired cranberry farms is one of the most promising alternatives to sewering to improve coastal water quality in southeastern Massachusetts. Agricultural fertilizer management may improve water quality in select embayments of Buzzards Bay (Buzzards Bay National Estuary Program, 2013), but is unlikely to curtail N loading to embayments on Cape Cod, where septic systems account for 87% of the controllable (non-atmospheric) N load to coastal embayments (Cape Cod Commission, 2022). Given the estimated cost of the restoration ($3,000,000; O'Connor, 2017; Truslow, 2018) and the amount of the N removed by restoration (F_{ret} = 1,029 kg N yr^{-1}), a back-of-envelop calculation suggests that ecological restoration of retired cranberry farms costs approximately ~$2,910 per kg of N. However, the restoration effort at Tidmarsh balanced multiple goals, not just water quality, and so ~$2,910 per kg of N is perhaps not the best estimate of the cost of N removal by wetland restoration as the exact amount of N removed depends not only on the specific goals of the restoration project, but also on the properties of the wetland and its watershed, as both strongly influence the amount and fraction of N removed by wetlands.

To fill this knowledge gap, Wiegman et al. (2025) developed a model that quantified N loading to 24 coastal embayments in response to ecological restoration of all cranberry farms within the embayment's watershed. Although the "all farms" scenario, in which all farms in an embayment's watershed were restored to wetlands, seems unlikely over the immediate future (i.e., the state's cranberry acreage has declined by only 4% over the past 10 years; USDA-NASS 2024), it provided an upper estimate of the coastal water quality benefits of ecological restoration of retired cranberry farmland. To estimate N removal, Wiegman et al. (2025) derived removal efficiency quartiles (25th: 23%, 50th: 38%, 75th: 53%) for 78 restored and created wetlands in temperate regions based on data filtered from two meta-analyses (Land et al., 2016; Walton et al., 2020). For the model simulations, the median value of 38% N removal equated to N reductions of less than 3% in nine embayments, 3%–10% in seven embayments, and 10%–30% in eight embayments for the "all farms" scenario. The 28% N removal that we observed at Tidmarsh suggests that removal potential in this region may be somewhat lower, closer to the 25th percentile of comparable wetlands in the analysis by Wiegman et al. (2025). Applying Tidmarsh's 28% N removal to the Wiegman et al. (2025) "all farms" scenario above corresponds to N reductions to coastal embayments of <2% in nine embayments, 2%–7% in seven embayments, and 7%–22% in eight embayments. In addition, the 28% N removal rate does not account for N removed after retirement but prior to restoration; that is, N removal in wetlands formerly cultivated for cranberry production (i.e., retired from commercial production) but not ecologically restored. Assuming N is removed in retired farms, 28% N removal would be an upper limit for restored wetlands. Either way, results of Wiegman et al. (2025) and this study indicate that ecological restoration of retired cranberry farmland is an unlikely substitute for sewering (or other forms of enhanced wastewater management) to meet water quality goals of coastal emabayments on Cape Cod and in southeastern Massachusetts.

6. Summary and Conclusions

Field measurements and modeling were used to evaluate N retention in Tidmarsh Farms East ("Tidmarsh"), the largest freshwater wetland restoration project ever conducted in Massachusetts. The goals of this study were to estimate (a) the magnitudes and sources of watershed inputs of N and (b) the rates and mechanisms of N retention in Tidmarsh. Historical land use analysis was used to document extensive urban development in the watershed between 2001 and 2019, when ~30% of the watershed transitioned from natural vegetation to development. Watershed-scale water quality modeling revealed that wastewater and turfgrass fertilizers were the major sources of N, contributing 80% of the N inputs to Tidmarsh. Based on a stream reach mass balance, field measurements showed that Tidmarsh retained 28% of NO_3^- inputs from surface water and groundwater, which was slightly lower than measured values of N retention for other restored wetlands (Land et al., 2016), ~2.5 times lower than average N retention in natural wetlands (Saunders & Kalff, 2001), and ~3 times lower than N retention assumed for wetlands in the NLM (Valiela et al., 1997). Many factors may influence N retention rates for wetlands (Lee et al., 2009), but perhaps none more significantly than N loading to wetlands, which has been shown to increase the amount N retained (Saunders & Kalff, 2001) while decreasing the overall fraction of N retention (Fisher & Acreman, 2004; Hansen et al., 2018). However, N inputs were not strongly correlated with F_{ret}, perhaps because the amount of N retained by Tidmarsh (18.4 kg N ha^{-1} yr^{-1}) was on the low end of the range of N retention rates for created and natural wetlands (Mitsch et al., 2014). Although these results point to factors other than N inputs, such as hydrologic connectivity between the stream and floodplain and hydraulic residence time, that may contribute to the relatively low N retention at Tidmarsh, we found that NTT was not strongly correlated with N retention, possibly due to the age of the restored wetland. As a relatively recent restored wetland (~7 years), we expect N retention in Tidmarsh to increase with time, particularly as the vegetation, hydrologic connectivity, and residence time increases and plants and soils mature (Aber et al., 1989; Ballantine et al., 2017; Hernandez & Mitsch, 2007; Jordan et al., 2011). In Tidmarsh's current state of establishment, the strongest predictor of N retention was stream temperature, which likely governed in-stream processes of plant uptake and denitrification. Although design and construction play an important role in water quality benefits of restored wetlands, inherent edaphic and hydrologic properties, such as stream temperature, soil pH, and watershed hydrology, may limit the potential for water quality improvements via wetland restoration (Wiegman et al., 2025). Still, ecological restoration of wetlands on retired cranberry farms can reduce watershed inputs of N to coastal embayments and provide other environmental benefits, but will likely not be adequate to meet N abatement goals for most impaired coastal embayments in Massachusetts.

Conflict of Interest

The authors declare no conflicts of interest relevant to this study.

Availability Statement

Chemical and physical data for analysis of nitrogen fluxes and nominal travels times in surface water are available on Ag Data Commons at https://doi.org/10.15482/USDA.ADC/30927932. Figures were made using commercial-available software, including SigmaPlot v. 15. (grafiti.com). and Adobe Illustrator (adobe.com). Maps and geospatial analysis were conducted in ArcMap v. 10.8 (Esri, 2020). Statistical analysis was carried out in JMP® 12 Pro (jmp.com) for macOS (v. 15.7.4).

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