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Modeling Attenuation of Nitrogen Loads Delivered to Coastal Bays From Ecological Restoration of Cultivated Wetlands

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Key Points:

- Restoring functioning wetlands on retired cranberry farms can substantially decrease watershed N inputs to coastal embayments
- Potential N attenuation due to wetland restoration was strongly influenced by the drainage area which varied 10,000-fold across ~1,000 farms
- Wetland restoration can be spatially targeted to areas with high N load reduction potential to make efficient use of conservation resources

Supporting Information:

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

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Abstract Nitrogen (N) pollution is a major threat to coastal ecosystems, worsened by the loss or degradation of natural wetlands, which historically acted as N sinks. In the glacial outwash plain of Southeastern Massachusetts, N pollution primarily from human waste and turf fertilizer has caused coastal eutrophication. Social and economic factors have driven ecological restoration efforts on wetlands previously modified for cranberry farming. These restoration projects offer a chance to enhance ecosystem N attenuation, but the extent and spatial distribution of watershed N loads through these farms remain poorly understood. To address this gap, we adapted a U.S. Geological Survey (USGS) groundwater model to identify wetland contributing areas and model potential N load reduction from the retirement and restoration of 984 cranberry farms. Using modeled contributing areas and data and assumptions about attenuation rates, we estimated N load reductions for farm retirement and restoration scenarios in 24 embayments. For restoration of all farms, median N load reductions were less than 3% in nine embayments, 3%–10% in seven embayments, and 10%–30% in eight embayments. Attenuation was limited by the contributing area intercepted by cranberry farms, ranging from 1% to 75% of watershed areas. Our model serves as a screening tool to identify farms with high potential to reduce watershed N loads, but more field monitoring is needed to refine N attenuation estimates in former cranberry wetlands. This work highlights the critical linkage between wetlands, development patterns, and ecosystem health, emphasizing the need for sustainable resource management approaches.

Plain Language Summary Coastal water quality in Southeastern Massachusetts is declining due to diffuse nitrogen pollution primarily from human waste, turf fertilizer, and atmospheric deposition. The region is densely covered with groundwater-fed wetlands that occupy glacially formed kettle-hole depressions and coastal valleys. Many of these wetlands have been modified for agriculture, particularly cranberry farming, which has decreased their extent and limited their capacity to attenuate (trap and remove) inflowing nitrogen. In most regions, watershed boundaries can be mapped using surface elevation, but in the coastal plain of Southeastern Massachusetts, nitrogen travels primarily with groundwater rather than overland flow. To address this, we used U.S. Geological Survey groundwater models to develop a tool to rapidly estimate contributing areas and identify which cranberry farms receive the greatest nitrogen loads from upstream sources. Our analysis shows that restoring ecologically functioning wetlands on select farms may substantially reduce nitrogen inputs to coastal bays and improve water quality. This approach demonstrates how targeted wetland restoration, informed by groundwater models, can enhance nutrient attenuation in agriculturally influenced landscapes. The findings offer practical guidance for improving the efficiency of restoration programs and may be applicable to other regions of the world that suffer from groundwater nitrogen pollution and degraded wetlands.

1. Introduction

Nitrogen (N) pollution is a pervasive problem in coastal estuaries that can result in harmful algal blooms, benthic hypoxia, and biodiversity loss (Nixon, 1995; Rabalais, 2002). N travels through watersheds most readily as nitrate (NO_3^-), the oxidized, reactive, and most mobile form of N in the environment (Galloway et al., 2003; Howarth et al., 1996). Because of this, NO_3^- is a primary cause of eutrophication and ecological decline in a growing number of estuaries worldwide (Orth et al., 2006). Despite these trends, recent examples of ecological resurgence

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in estuaries along the Atlantic coast of North America illustrate how watershed N mitigation plans can result in successful restoration of aquatic ecosystems (Castro et al., 2003; Tomasko et al., 2018).

The main sources of N to coastal zones in the United States (US) are agricultural and turf fertilizers, animal and human waste, and atmospheric deposition (Castro et al., 2003). In developed areas where agriculture is a minor component of land use, human waste and atmospheric deposition are the main sources of N (Kanakidou et al., 2016; Lloret & Valiela, 2016; Valiela et al., 2016). In these areas, N load management plans typically focus on (a) enhancing wastewater treatment, (b) reducing land based nonpoint inputs, and (c) increasing watershed attenuation (Boesch, 2002; McLaughlin et al., 2022).

Upgrading wastewater treatment systems is costly and often constrained by surrounding land use (Garfí et al., 2017; Hunter et al., 2018). It is also difficult to reduce local nonpoint inputs (e.g., fertilizer, animal manure, residential wastewater, or "septic systems") to watersheds since anthropogenic N sources are diffuse and widespread and because environmental managers, particularly in the US, lack strong mechanisms for regulatory enforcement (Boesch, 2002; Paerl et al., 2014). Many parts of the world still lack wastewater treatment all together, despite well-documented negative consequences for humans and ecosystems (Maxcy-Brown et al., 2021; WWAP, 2017).

Given these limitations, nature-based solutions (i.e., "green" infrastructure) offer a promising complementary strategy to enhance watershed N attenuation, particularly in areas where there is obsolete or abandoned hydrologic ("gray") infrastructure (e.g., mill dams), growing demand for outdoor recreation and ecological restoration, and/or where political or socioeconomic barriers have stalled the construction of source reduction programs and modern waste treatment facilities (Thorslund et al., 2017). Nitrogen traveling through watersheds can be attenuated through multiple pathways, including plant and microbial uptake, sedimentation, sorption, and biogeochemical transformations such as denitrification—a key process in which NO_3^- is converted to gaseous nitrogen forms (N_2O or N_2) under anaerobic conditions, effectively removing reactive N from the ecosystem (Burgin & Hamilton, 2007; Lamba et al., 2017).

Wetlands—whether natural, restored, or constructed—are particularly effective at *retaining* N in sediments and biomass and transforming and *removing* N through gaseous losses. The high capacity for N removal in wetlands is largely due to their saturated soils, slow water movement, and redox gradients that create favorable conditions for coupled nitrification and denitrification (Kadlec & Wallace, 2009; Mitsch & Gosselink, 2015). In addition to N removal, wetlands support multiple processes for N retention, such as immobilization of N in plant biomass, adsorption of ammonium (NH_4^+) to organic or clay particles, and burial of organic nitrogen in sediments (Reddy & DeLaune, 2008). These N retention and removal process are spatially and temporally variable but can act in concert to *attenuate* (i.e., decrease or reduce) N loads at the watershed scale, particularly when wetlands are located in areas with high N loading or within groundwater discharge zones. Many functioning wetlands have been degraded or destroyed across the world due to infrastructure development and conversion to agricultural use (Davidson, 2014; Jessop et al., 2015). Ecological restoration of wetlands, therefore, represents a multifunctional tool to increase N attenuation capacity within hydrologic networks while simultaneously enhancing an array of other ecosystem services (Browne et al., 2018; Thorslund et al., 2017).

At the same time that many coastal communities struggle to address the ecological and public health impacts of excess N (Lloret et al., 2022; Valiela & Bowen, 2002), agriculture in coastal regions is challenged by both climate change (Mondal et al., 2023) and economic factors that lead to farmland loss (Pottern & Barley, 2020). The combination of these factors—plus the demonstrated role of wetlands in N retention and removal—could create potential widespread opportunities for watershed-scale N attenuation by situating wetland restoration on retired farmlands. Developing tools to determine where to restore wetlands to enhance N attenuation is an important application of biogeochemistry to solve a critical regional and global problem.

In this paper, we examine the potential impacts of freshwater wetland restoration on the N loads delivered to the coast of Southeastern Massachusetts, United States (Figure 1). In this region, cranberries (*Vaccinium macrocarpon*) are the principal food crop and many of the historically functional wetlands in this region have been modified for the commercial production of cranberries. Low cranberry prices, aging cranberry farm ownership, and competition with other cranberry growing regions have increased the retirement of cranberry farmlands but also create opportunities for wetland restoration (Hoekstra et al., 2020; MassDAR, 2016).

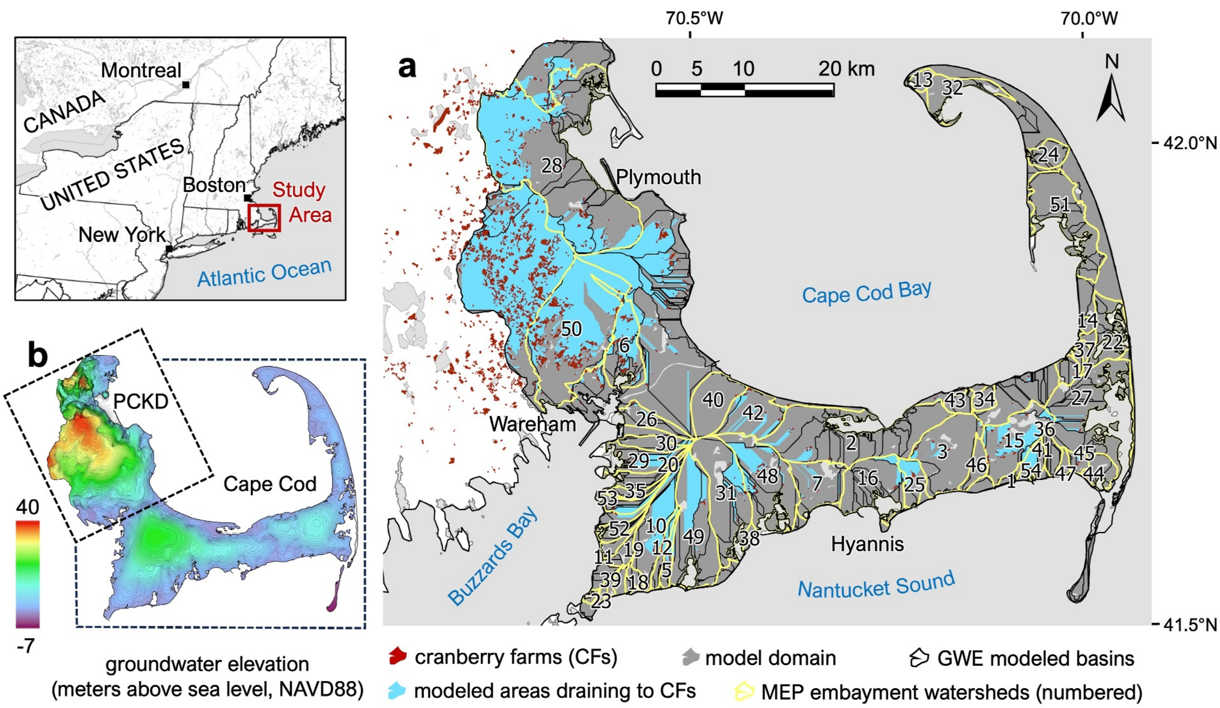


Figure 1. Map of the study area in Northeastern United States of America showing (a) modeled areas draining to cranberry farms (in light blue), the basins delineated by the groundwater elevation model (outlined in black), Massachusetts Estuaries Project (MEP) watersheds (outlined in yellow, numbered 1–54, see Figures 5 and 6) and (b) modeled groundwater elevation relative to sea level derived from MODFLOW simulations for the aquifers of Plymouth-Carver-Kingston-Duxbury (PCKD) and Cape Cod.

Estimating influent N loads is essential for selecting wetland restoration sites with high N attenuation potential. However, quantifying N inputs to wetlands is often complicated by heterogeneous hydrogeologic settings, which, for example, can cause groundwater flow rates to vary by orders of magnitude. In Southeastern Massachusetts, especially on Cape Cod, thick and relatively homogenous glaciofluvial deposits of sand simplify models of groundwater flow and thus subsurface transport of N. In addition, sandy soils allow precipitation and N to quickly infiltrate groundwater before reaching streams, ponds, and estuaries (Masterson & Walter, 2009), making overland flow a small and almost negligible N input to cranberry farms. Moreover, the study area has been the focus of seminal research on groundwater flow dynamics (Garabedian et al., 1991; Leblanc et al., 1991) and coastal water quality (Howes et al., 2001; Valiela et al., 1992; Valiela, Collins et al., 1997; Valiela, McClelland, et al., 1997), which add to our understanding of the fate and transport of N in groundwater and surface water. These factors make Southeastern Massachusetts an excellent natural laboratory for evaluating tools that can help identify, design, and monitor solutions to attenuate groundwater N pollution.

Capitalizing on our understanding of N transport processes and availability of groundwater flow and water quality models of the region, we developed a new model that can be applied as a screening tool to identify cranberry farms that have the greatest potential to intercept and attenuate watershed N loads through wetland restoration. To this end, we first applied a regional model of groundwater elevation to estimate contributing areas for all cranberry farms in the model domain, then generated maps of surface water N concentrations by compiling multiple data sources from which N loads to cranberry farms could be estimated. We applied a confidence interval of load reduction efficiency based on literature values to estimate potential N load reduction from restoration within 24 coastal watersheds that included cranberry farms. We evaluated hypothetical scenarios for retirement and restoration of (S1) all farms in a watershed and (S2) the individual farm receiving the highest incoming load in the watershed. Finally, we present an application of our model to a case study in the Three Bays estuary on Cape Cod —an embayment with impaired water quality that is currently developing and implementing a total maximum daily load (TMDL) plan for N reduction.

2. Methods

2.1. Wetlands and Cranberry Farming

From a nutrient loading perspective, the commercial production of cranberry (*Vaccinium macrocarpon Ait.*) is the only substantial form of agriculture in Southeastern Massachusetts (Williamson et al., 2017). Today, Massachusetts produces about one quarter of the U.S. domestic yield of cranberries on about 4,500 ha (USDA-NASS, 2021). By area, the majority of cranberry production occurs in Plymouth County within areas draining to Buzzards Bay; however, a considerable production area also exists in watersheds draining to Nantucket Sound and Cape Cod Bay (Figure 1a).

Cranberry farming originated in Southeastern Massachusetts on low lying peatlands that receive groundwater inputs (Kennedy et al., 2018). Indigenous peoples harvested wild cranberries from natural peatlands for food, medicine, and dye. Commercial cultivation began in the early 19th century when European settlers adapted these wetlands for larger-scale cranberry farming by adding layers of sand to the growing platform and constructing ditches, dams, and water control structures around the perimeter. Commercial cranberries are grown under drained conditions and then seasonally flooded in fall for harvest, in winter for vine protection, and sometimes in spring for insect management, among other reasons. Conventional fertilizers and pesticides are used regularly in cranberry farming. Fertilizer N is applied most as urea or ammonium sulfate at rates of 30–80 kg N ha^{-1} yr^{-1} (Kennedy & Hoekstra, 2021). Legacy N fertilizer builds up in organic peat layers and can be exported from cranberry farm drainage as organic N or as ammonium. Nitrate concentrations in cranberry farm drainage are often well below 0.01 mg N L^{-1}, possibly due to acidic soil conditions that inhibit nitrification, rapid microbial uptake, and/or denitrification (Kennedy et al., 2020). Despite these characteristics, active cranberry farms are still a significant source of total nitrogen (TN) to portions of Buzzards Bay (Kennedy & Hoekstra, 2021; Williamson et al., 2017).

Because of demographic and economic trends, the total cranberry production area in Massachusetts has declined by 15% (~400 ha) over the past decade, as less-productive older beds have been retired from production (Hoekstra et al., 2020). Retired cranberry beds are often colonized by upland plant species due to hydrologic disconnection between the water table and bog surface (Neill et al., 2023), as farms in the region typically consist of anthropogenic sand underlain by poorly drained low-lying peatland soils (Kennedy et al., 2018). Therefore, restoration is often needed for wetland plants, hydrology, and biogeochemical functions to return (Ballantine et al., 2017; Neill et al., 2023; Rubin et al., 2021).

Consequently, the Massachusetts Division of Ecological Restoration established a program for cranberry farm restoration in 2018 (MassDER, 2024). Current hydrologic restoration practices on former cranberry farms include selective removal of sand to introduce microtopography and promote groundwater exchange, removal, or modification of agricultural hydraulic control structures, excavation of meandering stream channels, and placement of in-stream woody debris (Ballantine et al., 2020). In addition to enhancing habitat diversity and promoting carbon sequestration, these hydrologic and geomorphic changes can promote N removal (Rubin et al., 2021). Therefore, restoring wetlands on retired cranberry farms is viewed by many in the region as a unique and timely opportunity to conserve biodiversity, mitigate climate change, and protect water quality.

2.2. Study Area and Data Sets

The study area consists of the U.S. EPA designated sole-source drinking water aquifers of Cape Cod (Sagamore, Monomoy, Lower Cape) and the towns of Plymouth, Carver, Kingston, and Duxbury (PCKD) (Figure 1). These aquifers supply drinking water for ~300,000 permanent residents (MassGIS, 2022) and nearly 600,000 people during the peak tourist season between May and September. Combined, the PCKD and Cape Cod aquifers are ~1,700 km^2 in area with ~850 km of coastline and ~3,500 ha of active and retired cranberry farms.

The study area has been the focus of extensive hydrologic research by the U.S. Geological Survey (USGS) (Carlson et al., 2017; LeBlanc, 1984; Masterson et al., 2009; Persky, 1986; Walter & Masterson, 2011; Walter et al., 2019). Over the past few decades, USGS developed and maintained MODFLOW groundwater models for the PCKD aquifer and Cape Cod aquifers to study water availability and pollutant transport (Carlson et al., 2017; Masterson et al., 2009). In the 1980s, USGS research documented groundwater pollution from centralized wastewater treatment and household septic systems (LeBlanc, 1984; Persky, 1986). Since then, ecological researchers have linked declining water quality and loss of key species to land use and housing density in the region

(Cole et al., 2006; Valiela et al., 2000; Valiela, McClelland, et al., 1997). In response to these findings, the Massachusetts Estuary Project (MEP) was initiated in 2001 as a collaborative effort among the Massachusetts Department of Environmental Protection (DEP), University of Massachusetts Dartmouth, and communities of Southeastern Massachusetts.

The objective of the MEP was to determine N loads for TMDL planning in 70 coastal embayments within Southeastern Massachusetts including embayments on the mainland, Cape Cod, and the islands of Nantucket and Martha's Vineyard. The MEP applied the linked watershed-embayment model to estimate N loads to coastal embayments and to investigate the potential impacts of future buildout and N load reduction scenarios on coastal water quality (Howes et al., 2001). The MEP model estimates of N loads were based on land use, the number of households, parcel size, and water use, among other factors (Howes et al., 2001; Kennedy & Hoekstra, 2021). In addition, USGS MODFLOW models were used to delineate contributing areas for the embayments investigated by MEP (Carlson et al., 2017). Between 1999 and 2007, the MEP monitored surface water discharges and N concentrations weekly for one year or more in 96 streams across the region (MEP, 2025). We synthesized these data to inform our modeling.

Other state and nonprofit organizations have also developed water quality monitoring programs in collaboration with research laboratories in the study area. For instance, annually since 1992, the Buzzards Bay Coalition (BBC) has performed summer water quality monitoring on streams and bays (Jakuba et al., 2021). The Massachusetts DEP Watershed Planning Program (WPP) has conducted sporadic water quality monitoring of streams and lakes areas since 2005 (WPP, 2024). Most recently, Cape Cod Rivers Observatory (CCR) initiated a weekly to bimonthly water quality monitoring program in 2016, including periodic measurements of stream discharge, for several large streams (Woodwell Climate Research Center, 2024). As with the MEP, we leveraged data from each of these research programs to inform our modeling in this study (Table S1 in Supporting Information S1).

We adapted the USGS-simulated MODFLOW groundwater elevations to estimate surface and groundwater discharges traveling through cranberry farms. We verified these modeled estimates against MEP, USGS, and U.S. Department of Agriculture (USDA) field measurements of surface water discharges. Then, we leveraged concentration data from each of the programs listed above to develop regional maps of N concentrations in surface water (Table S1 in Supporting Information S1), which were multiplied by discharges to estimate N loads traveling through cranberry farms in surface and groundwater, as described below.

2.3. Cranberry Farm Polygons

We obtained a geospatial data set of cranberry farm polygons from Massachusetts DEP (Jim McLaughlin, MassDEP 2013, personal communication; MassDEP, personal communication). This layer included data indicating whether the farm was "active," "inactive", or "abandoned." The cranberry farm polygons were preprocessed in ArcGIS Pro to separate portions of cranberry farms that were (a) separated by long distances or (b) obstructed by features like roadways that would prohibit stream reconnection. Polygons that were within 30 m of one another were coded as one farm, while polygons that were separated by 30 m or more were split apart. Polygons of $<100\ m^2$ were excluded from the analysis. For the remainder of this paper, the term "cranberry farm" or "farm" is defined (based on our preprocessing described above) as a hydrologically connected cluster of one or more cranberry beds that are separated by a horizontal distance of 30 m or less and are owned/registered by the same entity.

2.4. Water Table Model for Delineating Contributing Areas of Cranberry Farms

For each cranberry farm (as defined above), we determined the groundwater contributing area using ArcGIS Pro by applying topographic watershed delineation techniques to groundwater elevation contours exported from USGS MODFLOW simulations for the PCKD aquifer and aquifers on Cape Cod (Carlson et al., 2017; Masterson et al., 2009). To do this, we collapsed outputs of a 3D steady-state flow model into a 2D groundwater elevation model without particle tracking. We converted 2 ft (0.6096 m) groundwater elevation contours from the MODFLOW output (simulated with 121.92 m horizontal resolution) to a raster digital elevation model (DEM) with 10 m horizontal resolution using the Topo to Raster tool in ArcGIS. Then, we modified the groundwater elevation DEM using surface LiDAR and flowlines from the National Hydrography Data set (NHD) to develop an enhanced model of water table elevation (surface and groundwater) that more precisely resolved discharge/recharge zones and routes for surface water flow in ponds, stream networks, and cranberry farms. We subtracted

1 m (the approximate water table depth in cranberry farms) from raster cells that overlapped within 30 m buffer NHD flowlines, and then we filled "sinks." This process of "hydrologic enforcement" promotes drainage into streams channels, allows water to pass through raised surfaces that would artificially impede flow routes (e.g., a road crossing a stream with a culvert that is not detected by LiDAR), and ensures that all flows reach the coast. After resolving flow routes, we computed flow direction and flow accumulation across the entire domain of the enhanced groundwater elevation model. We then calculated discharge from flow accumulation assuming cranberry farms intercept 100% of surface water and groundwater from all cells in their upgradient contributing area, where the contributing area was equal to the raster cell with maximum flow accumulation inside the polygon of each cranberry farm (i.e., assuming no groundwater underflow, Valiela, Collins et al., 1997; Valiela, McClelland, et al., 1997; see Section 3.1 "Watershed Hydrology of Cranberry Farms").

During development, we tested the performance of various models of water table elevation used to route flow (see Text S1, Table S2 in Supporting Information S1) and three different flow direction algorithms available from ArcGIS Pro Spatial Analysis toolbox (see Text S1, Table S3 in Supporting Information S1). We verified our estimates against estimates of annual surface water discharge from 7 USGS stream gauges, 73 streams monitored by MEP, and 9 cranberry farms monitored by USDA, for a total of 89 surface water discharge monitoring points (Table S4 in Supporting Information S1). Sixteen of the verification sites were located at the outlet of an active or retired cranberry farms. Ultimately, we selected a model in which areas where the MODFLOW groundwater elevation was below the LiDAR surface elevation were represented by the MODFLOW groundwater elevation and areas where groundwater elevation was above the LiDAR surface elevation were replaced by the LiDAR surface elevation (see Text S1, Table S2 in Supporting Information S1). We used the "D8" flow direction algorithm, since D8 performed similarly to the other available methods, but unlike the other methods, D8 could also be readily used to delineate drainage areas (Table S3 in Supporting Information S1). A workflow diagram for preparation of the selected model is provided in Figure S1 in Supporting Information S1.

2.5. Geospatial Model of Nitrogen Concentrations

Given the ramifications of N pollution in the region, several N loading models have been developed for watersheds in Southeastern Massachusetts (Howes et al., 2001; Kennedy & Hoekstra, 2021; Valiela, Collins, et al., 1997; Williamson et al., 2017). Such models, while generally accurate, require numerous assumptions and are expensive (often prohibitively so) to develop. About 60% of the study domain was covered by watersheds with models developed for planning TMDLs by the MEP (Howes et al., 2001). Rather than develop complex N loading models to cover the remaining area, we developed maps of N concentration in surface water using a geostatistical analysis in ArcGIS Pro.

We used the empirical Bayesian kriging (EBK) function in ArcGIS Pro (Gribov & Krivoruchko, 2020; Zaresefat et al., 2023) to generate maps of mean TN and NO_3^- concentrations in surface water from monitoring locations in lakes, streams, and rivers within Southeastern Massachusetts. The data were compiled from the MEP, CCR, WPP, BBC, and USGS (Table S1 in Supporting Information S1). We filtered the raw, combined data sets to include surface water monitoring locations that satisfied the following criteria: (a) at least one observation for TN or NO_3^-, (b) latitude less than 42.5°N, and (c) longitude less than 71.5°W. The filtered data set included 558 monitoring locations, with 199 sites located within the model domain. Either NO_3^- or TN observations were not reported for 229 of the 558 locations. In these instances, we fit a linear regression model to the remaining 329 sites to estimate NO_3^- or TN concentration in surface water (Figure S2 in Supporting Information S1). Within the model domain, TN was not available for all 31 sites in the CCR data set and NO_3^- was not available for 57 of the 86 sites in the WPP data set, while MEP (67 sites), BBC (17 sites), and USGS (1 site) data included both NO_3^- and TN for all sites (Table S1 in Supporting Information S1).

We verified that N concentrations were spatially autocorrelated using Moran's I test in ArcGIS Pro (alpha = 0.05). We examined correlations between concentrations and landscape variables, such as population density and impervious cover, which serve as proxies for septic systems and have previously been associated with N loading rates (Cole et al., 2006; Persky, 1986; Valiela et al., 2016). To this end, we fed the D8 flow direction raster to the ArcGIS "Basins" function to delineate zones that collect flow that terminates at a sink on the edge of the domain. For each drainage basin, we tabulated population density from the 2020 census (MassGIS, 2022) and impervious cover from the 2016 landcover data set (MassGIS, 2019) and then extracted their values for each water quality monitoring site used in the EBK model to generate interpolated maps of TN and NO_3^- concentrations (199 sites total). We then computed a Spearman rank correlation matrix for basin-level mean TN, NO_3^-, NO_3^-/TN, impervious cover (%), and population density (people per km^2).

2.6. Modeling Nitrogen Loads to Cranberry Farms

Nitrogen loads delivered to cranberry farms by surface water and groundwater were calculated using two methods, both of which were based on contributing areas draining to cranberry farms. For the first method (L1), N loads in surface water and groundwater were determined as the product of model estimates of discharge (Q) and TN or NO_3^- concentrations (C_x, where x is TN or NO_3^-). Values of Q were estimated by multiplying the contributing area (i.e., watershed) for a cranberry farm (A_{fw}) by the average groundwater recharge rate for the region (k_{gr}; 0.692 m yr^{-1}, Walter & Masterson, 2011) (Equation 1), which was then multiplied by the 50th percentile predicted value of the mean N concentration in surface water derived from the EBK interpolation model (C_x):

$$\mathrm{L1} = \left(A_{\mathrm{fw}}\, k_{\mathrm{gr}}\right) C_x. \tag{1}$$

For the second method (L2), we calculated TN loads in surface water and groundwater using the attenuated watershed N load for the embayment (M_{ew}; kg N yr^{-1}) that was determined based on modeling conducted by the MEP. We then converted M_{ew} to an areal N loading rate by dividing by the area of the embayment watershed (A_{ew}). Finally, we calculated the fraction of the watershed N load moving through a cranberry farm by multiplying the areal N loading rate (kg N ha yr^{-1}) by the cranberry farm contributing area (Equation 2):

$$\mathrm{L2} = A_{\mathrm{fw}}\left(M_{\mathrm{ew}}/A_{\mathrm{ew}}\right). \tag{2}$$

The L2 method was only applied to cranberry farms located in watersheds of 52 embayments that were analyzed in the MEP reports. Of these 52 embayments, the 24 that contained active or retired cranberry farms within their watersheds were the focus of our analysis. Although both L1 (concentration) and L2 (MEP) methods relied on values of A_{fw} (see Equations 1 and 2), the two methods used different input data and approaches for estimating areal N loading rates. Thus, comparison of the two methods provided an independent assessment of uncertainty in N loading rates for contributing areas of cranberry farms.

2.7. Nitrogen Load Reductions From Ecological Restoration of Wetlands

We modeled potential N load reduction (LR), representing retention and removal (i.e., attenuation) of N traveling in surface water and groundwater through cranberry farms using a constant rate decay function (Equation 3):

$$\mathrm{LR} = L_x\,(E_r/100), \tag{3}$$

where L_x is the modeled influent watershed N load that enters and the farm based on L1 and L2 methods ($x = 1$ or 2, respectively), and E_r is the N load reduction efficiency expressed as the percent difference between N inputs and N outputs. E_r was derived from two recent meta-analyses: a review of E_r in wetland buffer zones across temperate regions in Europe and North America (Walton et al., 2020) and a global review of E_r in created and restored wetlands (Land et al., 2016). We combined these two data sets and filtered the values of E_r to include created and restored surface water flow wetlands from latitudes between 30°N and 60°N, resulting in 78 unique values of E_r for TN. We applied confidence intervals to LR estimates based on the 5th, 25th, 50th, 75th, and 95th percentile values E_r from the combined data set.

2.8. Ecological Restoration Scenarios

We report potential embayment-level N load reductions for the following two distinct restoration scenarios (Table 1). S1 ("all farms"), all active cranberry farms draining to an embayment are retired from production and all farm polygons (previously active or not) are ecologically restored to wetlands. S2 ("one farm"), a single cranberry farm, with the highest potential for reducing N loads to the embayment, is retired (if active) and restored to wetlands.

Table 1
Descriptions of Embayment-Level Scenarios (S1, S2) and Methods for Calculating Total Nitrogen (TN) Loads (L1 and L2)

Code	Description	Notes
Scenario		
S1 (all farms)	- Retire 100% cranberry farms and restore 100% of the farm area to wetlands in the watershed (marginal LR summed up for all farms)	- Maximum potential for N load reduction (LR) - Unrealistic for embayments with watersheds that contain many cranberry farms
S2 (one farm)	- Retire the cranberry farm with the highest total incoming load and restore 100% of the farm area to wetlands (load determined based on the L1 method described below)	- Best case for restoring a single farm - More realistic than S1
TN Load		
L1 ($Q \times C$)	For S1: farm "marginal" contributing area (A_{fw}) is applied to Equation 1 For S2: farm total A_{fw} is applied to Equation 1	- L1 was applied to all farms within the MODFLOW domain for PCKD and Cape Cod aquifers
L2 (MEP)	For S1: farm "marginal" A_{fw} is applied to Equation 2 For S2: farm total A_{fw} is applied to Equation 2	- L2 was applied only within watersheds where MEP models were developed

Note. See Table 2 for descriptions of variables and parameters.

As a first step, we aggregated the cranberry farm data to the watershed level for each embayment. As would be expected, many contributing areas of cranberry farms overlapped, as sections of cranberry farms are often hydrologically connected by a network of streams and ditches. To account for this overlap, we calculated the "marginal" contributing area for each farm using the "watershed" function in ArcGIS Pro. This was done by supplying the function with the drainage points for all cranberry farms and a flow direction raster derived from the water table elevation model. The marginal contributing area associated with a given drainage point was defined as the portion of the landscape that drains to that point, excluding any area that drains to an upstream point of interest (i.e., another farm). In other words, it represents the incremental drainage area between a downstream cranberry farm and the next upstream farm along the network. For example, if two cranberry farms with equal marginal contributing areas were situated sequentially along a stream, the total contributing area of the downgradient farm would be twice that of the upgradient farm, since it receives flow from both its own marginal area and the full contributing area of the upstream farm.

Nitrogen loads for the *marginal* and *total* contributing areas were calculated for each farm. For the "all farms" scenario (S1), N loads for the *marginal* contributing areas were summed for each embayment. For the "one farm" scenario (S2), the contiguous farm section with the greatest total N load was selected, as determined by the *total* contributing area, which was often located downgradient of all other farms in the watershed.

We also calculated the attenuated N load from fertilizers based on the area of "active" cranberry farms using a model that accounts for cranberry-specific coefficients for attenuation during transit through the vadose zone (Kennedy & Hoekstra, 2021). For our calculations, we assumed a fertilizer application rate of 45.8 kg N ha^{-1} yr^{-1} (Kennedy & Hoekstra, 2021) and that 20% of farms had "flow-through" hydrology, defined as farms with a central stream channel and connected to upstream and downstream surface water features (e.g., streams, ponds, or wetlands) (Hoekstra et al., 2020). We assume zero fertilizer loads for "inactive" and "abandoned" farms (i.e., retired). Fertilizer N loads are reported separately from watershed N reductions due to restoration, since farm retirement and restoration may differ in timing and motivation. Descriptions of model parameters are described in Table 2.

2.9. Computational Software

All geospatial data transformations were computed with ArcGIS Pro (version 3.2) using Jupyter Notebooks (Python, version 3.9) via the ArcPy package and the Spatial Analyst extension (Wiegman, 2025). All geospatial data were projected to the coordinate system of NAD83 UTM Zone 19°N (EPSG 26919) prior to subsequent processing. Preprocessing of N concentration and surface water discharge data and postprocessing of geospatial

Table 2
Descriptions of Selected Model Variables and Parameters

Variable (abbreviation)	Value	Units	Description
Farm surface area (A_{fs})	Spatial input	m^2	Area of the polygon from the MassDEP cranberry farm database (after preprocessing to split beds >30 m apart)
Fertilizer rate	Constant 45.8	kg N ha yr^{-1}	Annual N fertilizer application rate on active cranberry farms (Kennedy & Hoekstra, 2021)
Farm watershed area (A_{fw})	Simulated output	m^2	Based on D8 flow accumulation predicted for a water table elevation model derived from MODFLOW (Carlson et al., 2017; Masterson et al., 2009; Walter & Masterson, 2011) (see Figure S1 in Supporting Information S1)
Groundwater recharge rate (k_{gr})	Constant 0.692	m yr^{-1}	Precipitation minus evapotranspiration integrated across a watershed, estimate based on the calibrated value of 27.25 inches per year for the groundwater model (Masterson et al., 2009; Walter & Masterson, 2011) (see Equation 1)
Inflow concentration (C_x)	Simulated output	mg N L^{-1}	50th percentile (median) estimate of the empirical Bayesian kriging prediction of N concentration entering the cranberry farm (see Equation 1)
Watershed TN loading rate (M_{ew}/A_{ew})	Spatial input	kg N ha yr^{-1}	Attenuated watershed total N load comprised of atmospheric deposition, fertilizer, septic systems, landfills, and wastewater treatment facilities (Howes et al., 2001) divided by the total watershed area
TN reduction efficiency (E_r)	Stochastic 5–83	%	Confidence interval based on the 5th, 25th, 50th, 75th, and 95th percentile values E_r from 78 created and restored surface water flow wetlands from temperate region latitudes between 30° and 60°N (see Equation 3 and Figure 6)

modeling outputs were performed in R statistical software (version 4.3.2) using RStudio and the "tidyverse" suite of packages (Wickham et al., 2019; Wiegman, 2025).

3. Results and Discussion

3.1. Watershed Hydrology of Cranberry Farms

Within the model domain, we identified 984 of active or retired cranberry farms (clusters of beds separated by <30 m) ranging in the farm surface area from ~0.07 to ~61 ha (median ~2 ha) for a total area of 4,080 ha (Table 3). We estimated that approximately 30% (49,400 ha = 494 km^2) of the land area within the model domain drained through cranberry farms (Figure 1). Contributing areas for cranberry farms spanned from ~0.03 ha to ~8,230 ha (median 13.8 ha) (Table 3). The corresponding discharge via surface or groundwater ranged between ~0.57 and ~156,000 m^3 d^{-1} (median 262 m^3 d^{-1}). Across five orders of magnitude, our model reproduced ~93% of the variability ($R^2 = 0.93$) in the measured mean annual discharge of surface water for 89 verification points (Figure 2).

For the modeled verses observed discharge relationships, the root mean squared error (RMSE) was 6,360 m^3 d^{-1} and the mean absolute error was 3,705 m^3 d^{-1}, which was equivalent to a respective contributing area of ~1 ha and ~0.5 ha. The model performed similarly across different data sources and for streams with and without cranberry farms. However, linear regression analysis of modeled (x) versus observed (y) discharge yielded a slope of 0.89 ± 0.04 ($n = 89$) (Piñeiro et al., 2008). This suggests the possible "loss" of ~11 ± 4% of water beneath the watershed as interbasin groundwater flow or underflow (e.g., Genereux et al., 2005; Heberlig et al., 1997; Tóth, 1963; Valiela, Collins et al., 1997; Valiela, McClelland, et al., 1997). This interpretation is consistent with estimates from regional hydrogeologic modeling (Carlson et al., 2017), which suggests 10% underflow for ponds (Valiela, Collins et al., 1997; Valiela, McClelland, et al., 1997). The model tended to overpredict when modeled discharge was near or below the RMSE (Figure 2). In such small watersheds—which are less likely to be prioritized for restoration based on N load reduction—model estimates should be treated as an upper bound. We recommend site-specific hydrologic assessments (e.g., measuring channel dimensions and flow velocity) if N load reduction to coastal embayments is a priority for restoration.

3.2. Patterns and Sources of Nitrogen in Surface Water

For the 200 surface water monitoring locations located within the model domain, observed concentrations of NO_3^- ranged from <0.005 to 2.82 with a median of 0.21 mg N L^{-1}, while TN concentrations ranged from 0.16 to 3.38 with a median value of 0.82 mg N L^{-1} (Figures 3a and 3b; Figure S2 in Supporting Information S1). The

Table 3
Summary of Model Outputs for All Cranberry Farms Within the Study Area

Variable	Units	Minimum	$Q1$	Median	Mean	$Q3$	Maximum
Farm surface area (A_{fs})[a]	ha	0.069	0.92	2.05	4.1	5	61
Total contributing area (A_{fw})	ha	0.030	2.34	13.80	120	71	8,230
Marginal contributing area	ha	0.040	2.05	9.66	47	35	1,780
Discharge (Q)	$m^3\ d^{-1}$	0.57	44	262	2,270	1,340	156,000
Marginal discharge	$m^3\ d^{-1}$	0.76	39	183	894	662	337,00
Hydraulic loading rate (HLR = Q/A_{fs})	$m\ d^{-1}$	0.00005	0.003	0.011	0.099	0.042	15
NO_3^- concentration (C)	$mg\ N\ L^{-1}$	0.016	0.10	0.14	0.14	0.15	1.72
TN concentration (C)	$mg\ N\ L^{-1}$	0.37	0.64	0.69	0.70	0.72	2.25
NO_3^- load (L1)	$kg\ N\ yr^{-1}$	0.140	11.20	68.1	573	343	39,200
TN load (L1)	$kg\ N\ yr^{-1}$	0.026	1.81	12.7	108	61	9,530
NO_3^- loading rate (L1)	$kg\ N\ ha^{-1}\ yr^{-1}$	0.127	8.69	27.2	255	101	41,700
TN loading rate (L1)	$kg\ N\ ha^{-1}\ yr^{-1}$	0.021	1.59	4.87	47	18	8,230

Note. N loads are estimated using the L1 method. $Q1$ and $Q3$ represent the first and third quartiles, that is, the 25% and 75% percentiles, respectively. [a]Model input.

concentrations of TN and NO_3^- in surface water were the highest in the mid-Cape region and in densely populated coastal areas (Figure 3). Basin-level mean values of observed concentrations of TN and NO_3^- in surface water (Figures 3a and 3b) were both positively correlated with impervious cover (Figure 3e, $p < 0.001$) and population density (Figure 3d, $p < 0.001$) (Figure 3f). These findings are consistent with previous research that links surface and groundwater N concentrations with housing density on Cape Cod (Cole et al., 2006; Kroeger et al., 2006; Persky, 1986).

Across the model domain, EBK predicted median values of NO_3^- and TN concentrations in surface water ranged from 0.01 to 2.16 mg N L^{-1} and 0.24–2.82 mg N L^{-1}, respectively (Figures 3a and 3b). Predicted surface water

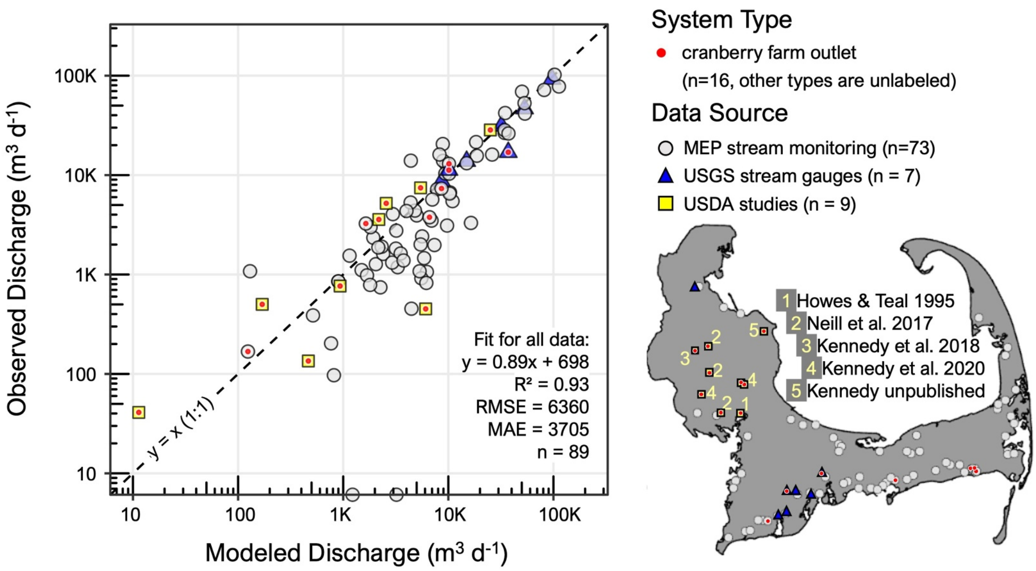
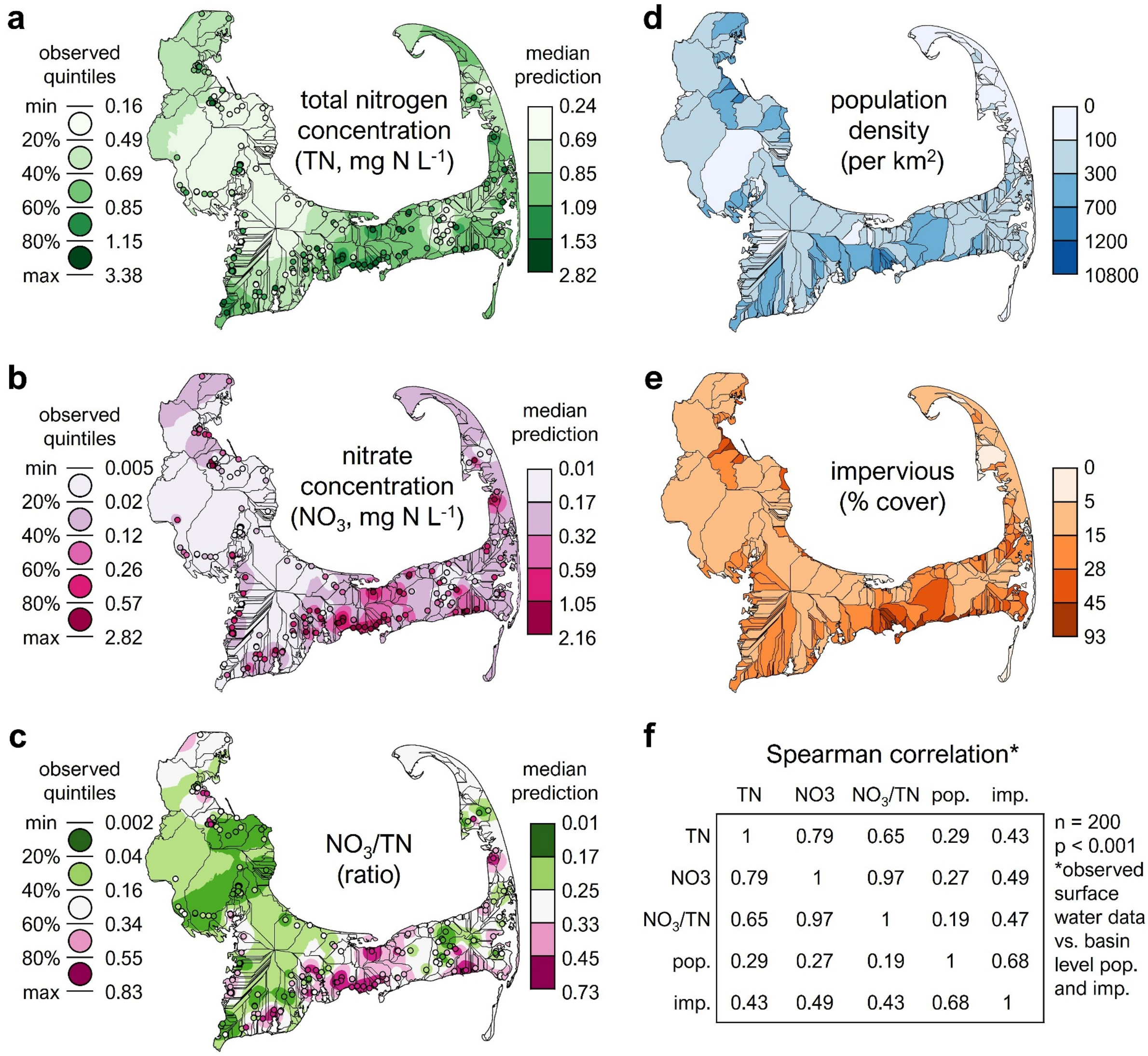


Figure 2. Comparison of modeled versus observed discharge (left) with map of monitoring locations by data source (right). The black line represents a perfect 1:1 fit ($y = x$); values below the line are overestimated, while values above the line are underestimated. Color and shape indicate different data sources: USGS (blue triangles), USDA studies (yellow squares), and MEP (gray circles). Sixteen of the 89 sites are located at the outlet of cranberry farms and are indicated by red dots. Note four MEP gauging stations are on the same stream reach as USGS gauging stations.



	TN	NO3	NO_3/TN	pop.	imp.
TN	1	0.79	0.65	0.29	0.43
NO3	0.79	1	0.97	0.27	0.49
NO_3/TN	0.65	0.97	1	0.19	0.47
pop.	0.29	0.27	0.19	1	0.68
imp.	0.43	0.49	0.43	0.68	1

Figure 3. Median predictions from the empirical Bayesian kriging model for surface water concentrations of (a) nitrate, (b) total N, and (c) the nitrate to total N ratio; (d) basin level mean population density for the 2010 census; (e) impervious cover from the 2016 landcover database; and (f) correlation matrix between observed concentrations and basin level population density (pop.) and impervious cover (imp.). Points on a, b, and c represent surface water records used for interpolation (quantiles for concentration data are given on the left side); modeled basins are outlined in black.

TN concentrations from cranberry farms were less variable than the broader model domain, ranging from 0.4 to 2.25 mg N L^{-1}, with a median of 0.69 mg N L^{-1} (Table 3, Figure S3 in Supporting Information S1). When averaged across an embayment watershed, mean TN concentrations in surface water from cranberry farms exhibited similar variance to MEP-based watershed N loading rates, which ranged from ~1.2 to ~32 kg N ha yr^{-1}, with a median of 8.4 kg N ha yr^{-1} (Figure S4 in Supporting Information S1). Embayment-level mean N concentrations predicted in surface water from cranberry farms were well correlated with MEP-based watershed N loading rates (Figure S4 in Supporting Information S1, Spearman's rho = 0.77 for TN, and rho = 0.68 for NO_3^-).

Watershed characteristics and biogeochemical processes strongly influence N speciation (Groffman et al., 2018; Reddy & DeLaune, 2008). Differences in land use, hydrologic connectivity, and soil biogeochemistry likely drive the observed variability in surface water NO_3^-/TN ratios, with forested, hydrologically complex systems favoring organic and reduced N forms, and urbanized systems favoring NO_3^--dominated export. Furthermore, soils throughout the study region tend to be acidic, which can limit nitrification and reduce NO_3^- availability relative to other forms of N such as ammonium (NH_4^+) and dissolved organic nitrogen (Robertson & Groffman, 2024).

Cranberry farming introduces N inputs as ammonium sulfate or urea and nitrification appears to be limited (Kennedy et al., 2020). As a result, when NO_3^- is observed in Southeastern Massachusetts, it is generally attributable to residential wastewater inputs (Valiela, Collins et al., 1997; Valiela, McClelland, et al., 1997). The Wareham River watershed (embayment #50 in Figures 1, 5 and 6), for example, is characterized by low population density and extensive cranberry farms, forests, lakes, and wetlands favoring lower NO_3^-/TN ratios. In contrast, the more urbanized Cape Cod aquifers have fewer forests and freshwater wetlands to attenuate NO_3^- as it leaches from developed areas into streams, favoring a higher proportion of TN as NO_3^- (Valiela, Collins et al., 1997; Valiela, McClelland, et al., 1997).

Wastewater-derived NO_3^- pollution has been observed in groundwater on Cape Cod for decades (Cole et al., 2006; Kroeger et al., 2006; Persky, 1986; Valiela et al., 2000, 2016). Our surface water analysis now confirms that many Cape Cod watersheds have exceeded their capacity for N attenuation, illustrating the impact that decades of unregulated wastewater inputs can have on water quality. The severity of water quality impairment on Cape Cod, with 90% of Cape Cod's embayments classified as eutrophic (APCC, 2023; Costa et al., 1992), is a microcosm of the rapid degradation of ecological status in coastal areas globally, with eutrophication recognized as a widespread and increasing threat to marine ecosystems (Diaz & Rosenberg, 2008), The spatial patterns we observed—higher NO_3^- and TN concentrations in high population areas and strong associations with land use—demonstrate that anthropogenic N loading increasingly overwhelms sources of natural attenuation of N, thereby reinforcing the urgent need for source control and watershed-scale N management plans.

3.3. Modeling Nitrogen Loads to Cranberry Farms

Our concentration-based L1 (Equation 1) estimates of TN loads delivered from surface water and groundwater to cranberry farms ranged from ~0.5 to ~39,000 kg N yr^{-1}, with a median of 115 kg N yr^{-1} across all cranberry farms (Table 3). By comparison, these TN loading rates were 33% lower, on average, than those generated using the MEP-based L2 method (see Equation 2) (Figure 4). A limitation of the L2 method is that it assumes spatially constant N loading rates for cranberry farms within a given watershed. This may be appropriate when factors affecting N loading are relatively constant in space (e.g., the intensely agricultural HUC8-watersheds modeled by Cheng et al., 2020). However, anthropogenic inputs of N were spatially variable within our study watersheds. Housing density, for example, was greater near the coasts (Persky, 1986). As a result, L2-based estimates of N loading rates could either be (a) erroneously high for cranberry farms located in mostly undeveloped and forested areas that are generally present in upper portions of the watershed or (b) erroneously low if the farm's contributing area is more developed than the rest of the embayment watershed. In contrast, the L1 method is less sensitive to spatial variations in N loading rates but is limited by spatial and temporal gaps (and biases) in N concentration data (Zaresefat et al., 2023). For our study area, data on N concentrations in surface water were collected between 1995 and 2024 (Table S2 in Supporting Information S1) and, spatially, data were more densely located on Cape Cod than the area representing the PCKD aquifer (Figure 3).

The L2 method is similar in principle to methods employed by a recent national assessment of NO_3^- removal from wetland restoration by Cheng et al. (2020). In contrast to Cheng et al. (2020), our L2-based estimates of N loads were determined for TN rather than NO_3^-, composed of both human and agricultural sources of N rather than just agriculture, and conducted at a wetland scale rather than a much larger (HUC-8) watershed scale. Additionally, Cheng et al. (2020) assumed that the wetland contributing area was proportional to the wetland surface area, an assumption that proved to be invalid for our study area (see above). As a result, the MEP-based L2 estimates of watershed loading rates for TN (mean ~ 10, max 32 kg N ha yr^{-1}) were approximately double that of Cheng et al.'s (2020) NO_3^- loading rates for watersheds in New England (mean ~ 4, max ~ 15 kg N ha yr^{-1}). Although national-scale assessments, such as Cheng et al. (2020), are important to help decision makers prioritize best management practices (BMPs) for mitigating N pollution, large-scale (HUC-8) watershed models provide insufficient resolution to help managers identify suitable locations for BMP implementation. Our approach to regional-scale modeling may provide a more robust, targeted approach for identifying high-priority wetlands for ecological restoration to improve coastal water quality.

3.4. Potential Nitrogen Reduction From Ecological Restoration

Of the 24 watersheds that contained cranberry farms, we identified 419 cranberry farms that drained approximately 25% of the land area across these watersheds. Collectively, these farms represented seepage faces for the

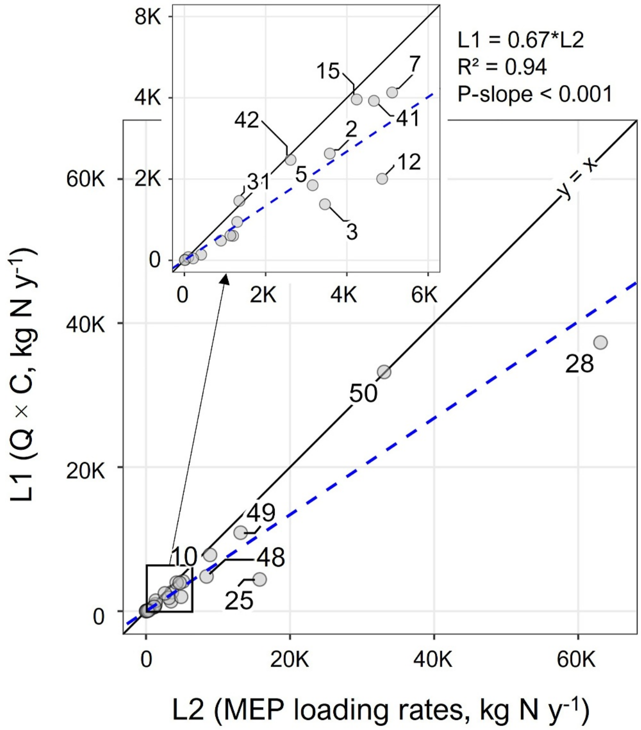


Figure 4. Comparison of estimated watershed N loads (kg N day^{-1}) discharging to cranberry farms for the concentration times discharge method (L1) versus contributing area times embayment level loading rates (L2) (see Table 1 for method details). Linear regression fit (slope of 0.67, $p < 0.01$) shown with blue dashed lines. Number labels on the plot correspond to embayment numbers on Figures 1, 5, and 6.

discharge of 130,000 (L1) to 195,000 (L2) kg N yr^{-1} of TN. The amount of TN transported via groundwater and surface water through cranberry farms corresponded to 17% (L1) and 25% (L2) of the combined watershed TN load delivered to these 24 embayments.

Individually, TN loads that discharged through cranberry farms represented between <1% and ~75% of TN delivered to the coast for the 24 embayments (Figure 5). In 10 of these embayments, cranberry farms intercepted >25% of TN loads, including >50% of the TN load to the Wareham River estuary, Parkers River estuary, Saquatucket Harbor, Green Pond estuary, and Bournes Pond estuary. Plymouth Harbor and the Wareham River ranked highest in terms of aggregate TN loads delivered to cranberry farms with respective loads of ~65,000 and ~35,000 kg N yr^{-1}, owing to the relatively large size of their watershed areas, the high number of cranberry farms, and the large proportion of their watersheds that drained through cranberry farms.

Cranberry fertilizer runoff from active farms was a minor component of the TN load in surface water with one exception, which was the Wareham River (yellow squares in Figure 5c). Effectively >99% of cranberry farms were active in each study embayment, except for Waquoit Bay (17% active), Plymouth Bay (55% active), Three Bays (78% active), and Popponesset Bay (95% active). All the farms selected by the one farm scenario (S2) were active. The percentage of the watershed TN load delivered to the coast from cranberry fertilizer runoff ranged from 0.1% to 16%, with a median of 0.9%. Of the 24 embayments, the Wareham River estuary (at 16%) was the only site where the aggregate cranberry fertilizer N load delivered to the coast exceeded 5% of the total watershed N load. The cranberry fertilizer runoff load was ≥2.5% of the watershed N load in Saquatucket Harbor, Plymouth Bay, Herring River (Harwich, MA), and Green Pond (Falmouth, MA). In all other bays, cranberry fertilizer contributed <2.5% to the total N load to the embayment.

Figure 6a shows modeled reductions in TN loads to an embayment resulting from wetland restoration if all cranberry farms within the watershed were retired and restored to natural wetlands (see "all farms" scenario, Table 1). Estimates of potential TN reductions for the concentration-based L1 method (Equation 1) were generally lower than estimates generated by the MEP-based L2 method (Equation 2). The numbers of

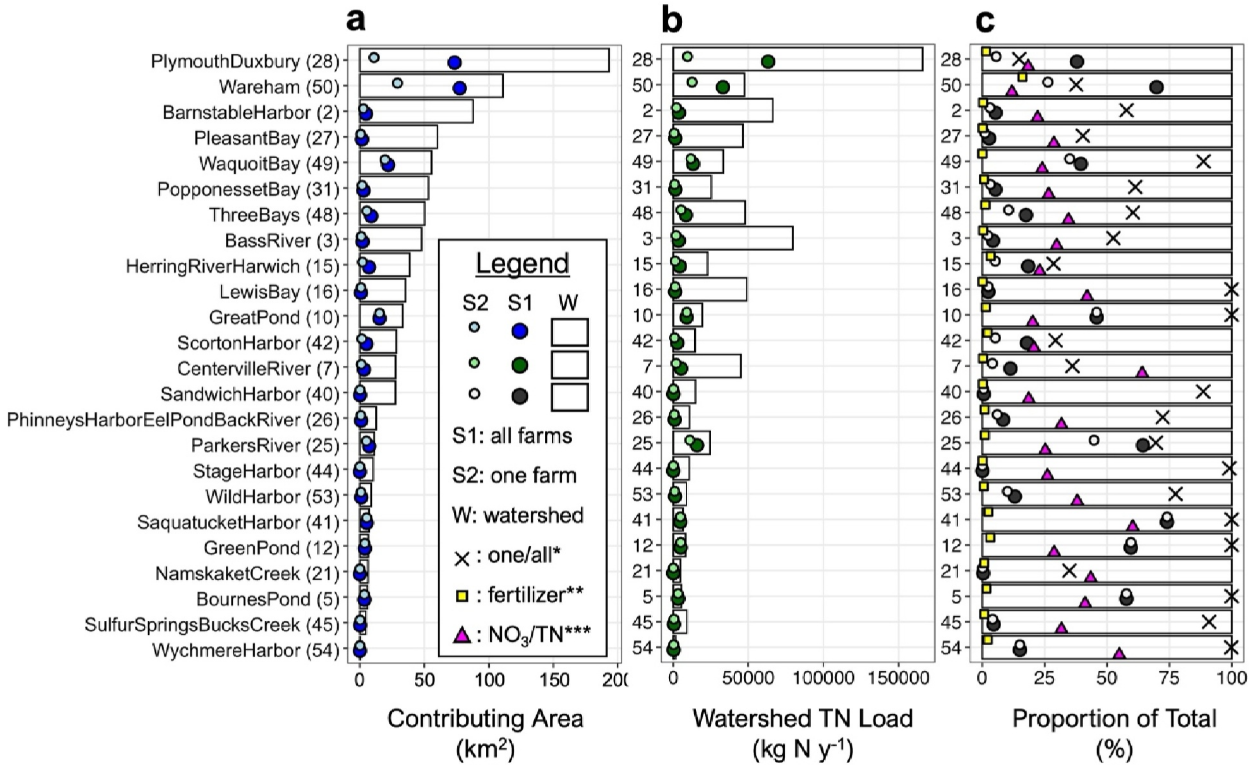


Figure 5. (a) Comparison of the contributing area, (b) watershed N loads, and (c) proportion of total area or load draining through "all farms" and the "one farm" that receives the highest N load. In each panel, small circles correspond to S2, larger circles to S1, and boxes to the entire embayment. *The Xs in "c" show the area (and/or load) draining through the "one farm" (S1) as a proportion of the area (or load) draining through "all farms" (S2). **Yellow squares represent the fertilizer load from all active cranberry farms within the watershed assuming an application rate of 45.8 kg N ha yr^{-1}. Numbers correspond to embayment map labels on other figures. ***Magenta triangles show the mean percent of TN as nitrate for "all farms." Loads reported here are for method L2 (Equation 2). See Table 1 for scenario descriptions.

embayments that had ≥10% reduction in watershed TN load were 4 and 9 with the L1 and L2 methods, respectively. For both L1 and L2 methods, 10 embayments had TN reductions from restoration of all farms that were <3%.

For most of the 24 embayments we analyzed, one farm intercepted more than half the TN load that flowed through all cranberry farms (black Xs in Figure 5c). Moreover, there were 7 embayments where one cranberry farm received >25% of the watershed TN load to the embayment, including the estuaries of Bournes Pond, Green Pond, Saquatucket Harbor, Parkers River, Great Pond, Wareham River, and Plymouth Harbor (small white circles in Figure 5c). For these seven embayments, restoring wetlands on the one cranberry farm that received the highest load (S2) resulted in a median watershed TN load reduction to the embayment of ≥10% based on MEP loading rates (L1) (Figure 6b). These results indicate that a spatially targeted approach could improve the cost effectiveness of wetland restoration at reducing N loads to embayments across much of the region.

3.5. Factors Influencing N Attenuation

Across all the cranberry farms that we modeled, contributing area and discharge spanned five orders of magnitude, while TN concentration and areal TN loading rate spanned less than two orders of magnitude. As such, the contributing area was the primary factor explaining the spatial variation in the magnitude of TN delivered to cranberry farms and attenuated by wetland restoration. Our model assumed an average TN load reduction efficiency (E_r) based on reported literature values. In reality, N attenuation is influenced by the array of processes that retain and remove N and are dependent on site-specific factors, including, but not limited to, surface area, water depth, hydraulic residence time, temperature, and soil biogeochemistry (Cheng & Basu, 2017; Crumpton et al., 2020; Kadlec, 2012; Kadlec & Wallace, 2009; Land et al., 2016; Walton et al., 2020).

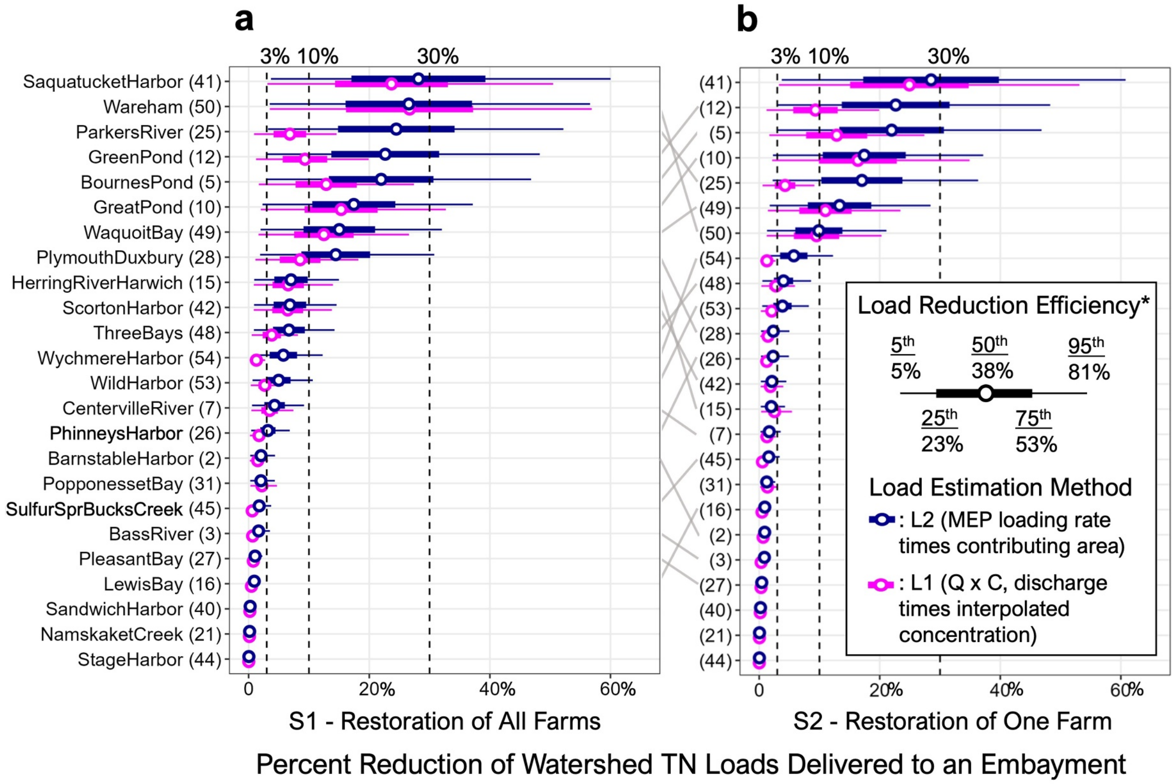


Figure 6. (a) Potential reduction in watershed TN load (%) resulting restoration of all farms within an embayment and (b) restoration of the one farm that receives the highest N load. *Load reductions are shown for percentiles of load reduction efficiency (E_r) for created and restored temperate surface flow wetlands compiled from two meta-analyses (Land et al., 2016; Walton et al., 2020). Magenta points/lines show load estimation method L2 (MEP), and dark blue points/lines show L1 ($Q \times C$). Note the change in rank between scenarios (gray lines connecting a and b). See method details in Table 1.

Physical hydrology exerts particularly strong influence over N load reduction across wetland types (see refs. above). For example, E_r generally declines asymptotically toward zero with the increasing hydraulic loading rate (HLR = Q/A_{fs}), although the overall attenuation rate (e.g., kg N ha^{-1} yr^{-1}) increases as a function of HLR (Kadlec, 2012). For the restore "one farm" scenario (S2), HLR ranged from ∼0.01 to ∼0.6 (m d^{-1}) (Table S5 in Supporting Information S1) and was more strongly influenced by discharge than the farm area (i.e., the size of potentially restorable wetland, Figure 7).

Low NO_3^- concentrations may limit the capacity for N removal (gaseous loss) within the study region (Ballantine et al., 2017; Kadlec, 2012; Klionsky et al., 2024). Our analysis suggests that TN and NO_3^- concentrations in surface water entering cranberry farms are far lower (>10x) than nutrient assimilation wetlands in other regions that receive agricultural runoff from row-crop farms and municipal wastewater (e.g., Crumpton et al., 2020; Kadlec, 2012). Of the 24 embayments, NO_3^- comprised less than 50% of TN in all but 3 and less than 30% of TN in all but 8 (magenta triangles in Figure 5c). Notably, in the Wareham River watershed, which is heavily impacted by cranberry agriculture, NO_3^- constituted just 12% of TN. Klionsky et al. (2024) found that groundwater seeps (discharge zones where N loads are high) were hotspots for denitrification within an ecologically restored wetland on Cape Cod, yet denitrification rates were relatively low compared to other reference wetlands in the region (Ballantine et al., 2017). Klionsky et al. (2024) suggested that denitrification rates in seeps were perhaps limited by organic carbon; however, low NO_3^- concentrations and short hydraulic residence time (McPhillips et al., 2015) could also have been limiting.

Finally, it is important to consider that the hydrologically connected ditches and stream networks of active and retired cranberry farms have inherent capacity for N attenuation. For example, Eichner et al. (2016) estimated a 22%–35% reduction of watershed N loads traveling through a retired yet unrestored cranberry farm located in the

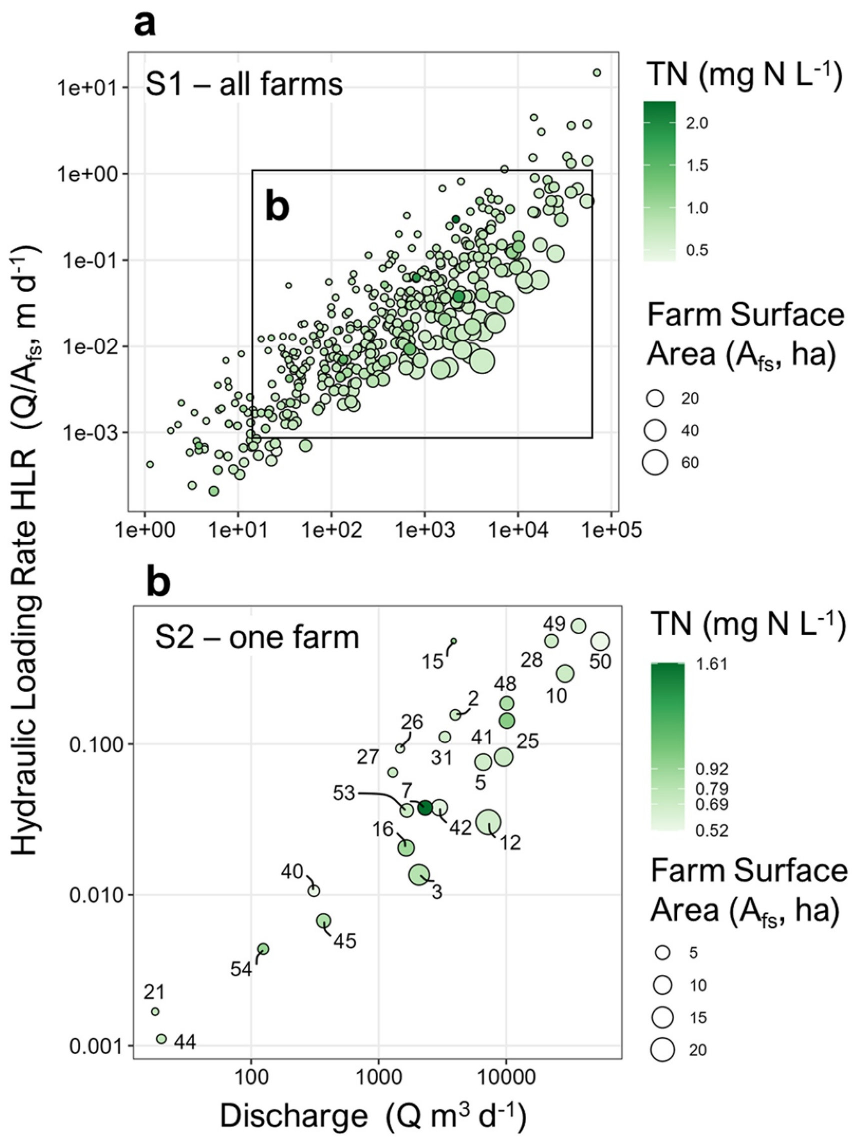


Figure 7. Scatter plot of comparison of hydraulic loading rate (HLR, m d^{-1}), discharge (Q, m^3 d^{-1}), and farm surface area (ha, i.e., restorable wetland area), TN concentration (mg/L, color), for "S1—all farms" (a) and the "S2—one farm" scenario (b). See Table 1 for scenario descriptions. Numbers next to points correspond to embayment numbers labeled on Figures 1, 5, and 6. Note the different scales for a and b.

Saquatucket Bay watershed, based on the difference between modeled inputs and measured outputs of N for the cranberry farm. Thus, to enhance N retention and removal beyond the level of existing conditions, wetland restoration designs should focus on the increasing wetted surface area and hydraulic residence time, as increases in these parameters have been demonstrated to promote higher E_r across a range of constructed and natural wetlands (Cheng & Basu, 2017; Crumpton et al., 2020; Kadlec, 2012).

3.6. Three Bays Case Study: Verification and TMDL Context

Here we compare N attenuation benefits of cranberry farm restoration against the TMDL reduction target for a site located in the Three Bays estuary (embayment #48 in Figures 1, 5, and 6). Three Bays is under a TMDL, and plans are actively being implemented to reduce watershed N loads. A TMDL target of ~26,500 kg N yr^{-1} was established for Three Bays in 2007. The estimated total watershed (controllable) load for the Three Bays is ~46,200 kg N yr^{-1}, requiring an N load reduction of ~20,500 kg N yr^{-1} to meet the TMDL target. The cranberry farm receiving the highest TN load within the watershed is a 5.4-ha cranberry farm located at the downgradient end of a ~31.6-ha cluster of cranberry beds (Table 3). This cranberry farm complex is scheduled for restoration in 2025 as part of The Marston Mills River Ecological Restoration Project; the primary goal of this project is to attenuate N and improve water quality in the Three Bays estuary (MassEEA, 2023).

At a USGS stream gauge (#0110588332) located at the outlet of this cranberry farm, field measurements of N loads in surface water were of similar magnitude to those modeled by L1 and L2. For the 3-year period between 1 October 2021 and 30 September 2024, the measured mean surface water discharge was 11,276 m^3 d^{-1} and the flow-weighted mean concentration of TN (n = 28) in surface water was 1.25 mg L^{-1}. Based on these field measurements, the resulting average annual N load in surface water was 5,138 kg N yr^{-1} for TN. At this location, USGS reports a contributing area based on the MODFLOW 3D particle tracking of 761 ha, yielding a surface water discharge of 14,436 m^3 d^{-1} when assuming a recharge rate of 0.692 m yr^{-1}. For comparison, we modeled a contributing area of 534 ha with a resulting discharge of 10,126 m^3 d^{-1} and a TN concentration in the surface water of 0.83 mg N L^{-1} at this location. Accordingly, we estimated TN loads of ~3,498 kg N yr^{-1} (L1) and ~5,075 kg N yr^{-1} (L2), with the latter closer to USGS field-based measurements (Table S5 in Supporting Information S1).

Our modeling based on L2 suggests that retirement and restoration of 31.6 ha of cranberry beds would result in TN load reduction of ~979 kg N ha yr^{-1} (when $E_{r,5}$ = 8%) to ~4,785 kg N ha yr^{-1} (when $E_{r,95}$ = 83%) (Table S6 in Supporting Information S1), depending on site conditions and restoration design (see "Factors Influencing N Attenuation" above). This level of N load reduction corresponds to between ~4.1% and ~23.3% of the ~20,500 kg N yr^{-1} TMDL reduction target. Equivalent N removal from sewered wastewater treatment would cost between ~$205,000 and ~$2,479,000 annually (Table S6 in Supporting Information S1). In this case, N attenuation from ecological restoration would exceed that from farm retirement (via fertilizer cessation estimated at 573 kg N yr^{-1}) by a factor of ~1.7 to ~8.4. This case study illustrates that with informed siting and ecological engineering, wetland restoration on retired farmland can provide high returns on investment for N load reduction and coastal water quality improvement.

3.7. Limitations and Intended Use

Our modeling approach aimed to estimate large (order-of-magnitude) differences in the amount of N that ecological restoration of wetlands could potentially attenuate within watersheds of Southeastern Massachusetts. To this end, we developed steady-state models of discharge and N concentrations in surface water, which have inherent limitations (discussed in Sections 3.2, 3.3 and 3.4). For instance, our 2D model of groundwater elevation, which was derived from a relatively coarse (121 m × 121 m) 3D groundwater elevation model, may slightly overestimate contributing areas (~11%) by neglecting interbasin groundwater flow (i.e., underflow) compared with more sophisticated 3D simulations of groundwater flow (McDonald & Harbaugh, 1988; Tóth, 1963).

Our model is intended to be a screening-level tool to help prioritize retired cranberry farms for their potential to reduce N inputs to coastal embayments. We recommend that managers use our model results to select a subset of candidate sites for more detailed assessment, including monitoring temporal dynamics in surface flows and N concentrations, and potentially 3D groundwater modeling, to inform site-specific restoration design and estimate N load reduction for baseline conditions and under different restoration designs. On Cape Cod, this tool could be particularly valuable because municipalities are now required to produce watershed-scale plans to meet strict N load reduction targets (MassDEP, 2023) and there are many retired or eligible cranberry bogs in Cape Cod watersheds. Wetland restoration can potentially offer a more cost-effective alternative to the expansion of municipal sewers and replacement of standard septic tanks with denitrifying septic tanks; however, its feasibility is limited to watersheds where retired farms intercept a substantial portion of the N load. Our model can help identify where wetland restoration is likely to be a practical and cost-effective N load reduction strategy.

This study highlights the importance of investing in foundational hydrologic science and building accessible data sets and models. Southeastern Massachusetts has benefited from decades of hydrologic research, long-term water quality monitoring, and models such as MODFLOW. These resources, along with the relatively uniform hydraulic properties of the study aquifers, allowed us to develop a relatively simple, steady-state modeling approach grounded in first principles. Our methods can be adapted to other regions where the geology is characterized by similarly uniform and transmissive unconfined aquifers, often found in unconsolidated sand and gravel deposits (e.g., glacial outwash) (Masterson & Walter, 2009) draining to coastal oceans and inland seas in northern latitudes. Estimating and verifying spatial heterogeneity of key variables—such as groundwater depths, recharge rates, and N concentrations—through targeted hydrologic and water quality monitoring is an essential first step to apply our approach (Carlson et al., 2017; LeBlanc, 1984; Masterson et al., 2016).

4. Summary and Conclusions

Groundwater N pollution originating primarily from septic systems as well as turf fertilizers (Williamson et al., 2017) has degraded water quality and ecological health in coastal embayments across Southeastern Massachusetts (Cole et al., 2006; Persky, 1986). We leveraged existing field data and models to estimate contributing areas and watershed TN loads delivered to cranberry farms in two sole source aquifers that serve up to ~600,000 people. We modeled potential N load reductions to coastal embayments resulting from farm retirement and wetland restoration. We found that, in the absence of wetland restoration, future farm retirements will only slightly reduce N inputs to coastal embayments. Export of fertilizer N from cranberry farms comprised ~2.5% or less of watershed TN loads in surface water in all but one embayment, the Wareham River (Figure 5c). On the other hand, wetland restoration on retired cranberry farms could substantially reduce the amount of N delivered to coastal embayments, with potential N load reductions exceeding 10% of the total watershed TN load in 8 of the 24 embayments included in this study (Figure 6). Contributing areas of cranberry farms varied widely and were the dominant factors influencing N load reduction (Figure 5a).

Our findings suggests that spatially targeting wetland restoration efforts on retired farmland can limit costs associated with land procurement and lost production while offering a viable strategy to improve coastal water quality in regions where nitrogen overloading is a primary stressor. However, watershed N inputs must also be reduced in order to realize long-term water quality improvements. In our study area, N source reductions can be achieved through septic upgrades (in low density areas) and installation of centralized sewer systems (in high density areas), with priority given to land areas with the shortest transit times to the estuary (Balogh et al., 2022; Merrill et al., 2021). Building this infrastructure will, however, require billions of dollars and may take decades to complete, reflecting a global challenge of upgrading water infrastructure to support a growing population and build resilience to climate change (FAO & UN-Water, 2024; OECD, 2024). Ecological engineering of wetlands (e.g., this study) and benthic ecosystems in estuaries (e.g., Merrill et al., 2021) can help mitigate further water quality degradation in the interim while providing additional ecosystem services (e.g., carbon sequestration, water flow regulation, and wildlife habitat), economic opportunities, and cultural benefits (Everett et al., 2018; Lefcheck et al., 2018; Liversage, 2020; Radinger et al., 2023; Rose et al., 2021).

Wetland restoration is a complex task, usually with many objectives. Practitioners may need to consider indigenous heritage, property rights, public access, invasive species, and potentially problematic chemical releases (agrichemicals, greenhouse gases, and soluble phosphorus, e.g., Wiegman et al., 2024). We emphasize that potential water quality benefits of restoration designs should be weighed against potential tradeoffs with other ecosystem services, project life cycle costs (engineering, construction, and maintenance, e.g., Rutherford et al., 2018; Wiegman et al., 2018), as well as the opportunity cost of putting additional land into conservation without restoration. Our model can aid in such efforts at site and watershed scales by acting as a screening tool that can quickly estimate potential N removal for a range of sites and scenarios before detailed site monitoring is conducted.

Conflict of Interest

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

Data Availability Statement

Water quality, stream flow, and MODFLOW data files are publicly available and cited within the main article (Jakuba et al., 2021; MEP, 2025; WPP, 2024; Woodwell Climate Research Center, 2024; USGS-NWIS, 2024). These data as well as scripts for preprocessing, geospatial modeling, and postprocessing are hosted on the authors GitHub: https://github.com/arhwiegman/Cran_Q_C repository and preserved on Zenodo https://doi.org/10.5281/zenodo.15757982 (Wiegman, 2025).

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