Amounts, forms, and management of nitrogen and phosphorus export from agricultural peatlands

Casey D. Kennedy1 | Anthony R. Buda2 | Ray B. Bryant2

1Pasture Systems and Watershed Management Research Unit, USDA-ARS, East Wareham, MA
2Pasture Systems and Watershed Management Research Unit, USDA-ARS, University Park, PA

Correspondence
Casey D. Kennedy, Pasture Systems and Watershed Management Research Unit, USDA-ARS, One State Bog Rd., East Wareham, MA.
Email: casey.kennedy@usda.gov

Abstract
Peatlands provide a setting that is well suited for cranberry agriculture in the Northeastern United States. However, misconceptions exist about the amounts and forms of nitrogen (N) and phosphorus (P) export from cranberry farms. In this study, we report inorganic and organic forms of N and P export from five peatlands cultivated for cranberry production in southeastern, Massachusetts, United States. We then compare N loading rates among cranberry farms in southeastern Massachusetts, row crop farms in the Midwestern United States, and uncultivated peatlands in the United States and United Kingdom. Based on a fluvial mass balance analysis, we find that nonriparian cranberry farms export 2.56 kg of P ha\(^{-1}\) year\(^{-1}\) of total P and 12.1 kg of N ha\(^{-1}\) year\(^{-1}\) of total N. Total N export from riparian or “flow through” farms is two times higher than nonriparian farms due to less retention of N fertilizer in the vadose zone of riparian farms. Gross total N export from riparian and nonriparian cranberry farms consists of 35% particulate organic N, 26% dissolved organic N, 31% ammonium (NH\(_4^+\)), and 8% nitrate (NO\(_3^-\)). The low proportions of NO\(_3^-\) export (13% of total dissolved N [TDN]) for cranberry farms differ from NO\(_3^-\) export for row crop farms (75% of TDN; \(p < .001\)) but not for uncultivated peatlands (17% of TDN; \(p = .61\)). Despite being highly modified by fertilizers and artificial drainage, low NO\(_3^-\) export (2.2 kg of N ha\(^{-1}\) year\(^{-1}\)) from cranberry farms is consistent with field measurements of rapid N turnover in uncultivated peatlands. This finding suggests that state-funded wetland restoration efforts to restore denitrification in retired cranberry farms may be limited by NO\(_3^-\) rather than soil moisture or organic matter.

KEYWORDS
agriculture, cranberry, nitrogen, peatland, phosphorus, water quality

1 INTRODUCTION

Bogs and fens—“peatlands”—are located throughout the Northeastern United States (Johnson, 1985). The abundance of peatlands is controlled by topographic gradients, geological formation, climate, and hydrology (Davis & Anderson, 2001), with peatlands accounting for 3% (13 \times 10^3 \text{ km}^2) of the terrestrial area of the Northeastern United States (U.S. Geological Survey, 2008; Xu, Morris, Liu, & Holden, 2018). Despite their minor spatial footprint, peatlands provide valuable ecosystem services such as carbon storage (Yu et al., 2010), water quality regulation (Johnston, 1991), flood mitigation (Acreman & Holden, 2013), and habitat provision (Bonn, Allott, Evans, Joosten, & Stoneman, 2014). In Massachusetts, peatlands also provide a setting that is well

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suited for production of the American cranberry (*Vaccinium macrocarpon* Ait.), which is the state’s most valuable fruit crop with an economic input of $1.4 billion (Massachusetts Department of Agricultural Resources, 2016; Executive Office of Energy and Environmental Affairs, 2017).

Nearly 20% of North American cranberry production is harvested from 4,980 ha of farmland in southeastern Massachusetts (Statistics Canada 2017; U. S. Department of Agriculture National Agricultural Statistics Service, 2017). Although cranberries are native North American plants, commercial cranberry production requires careful management of water (Kennedy et al., 2017a), nutrients (Hart, Strik, DeMoranville, Davenport, & Royer, 2015), pests and disease (Polashock, Caruso, Averill, & Schilder, 2016). Cranberry farms are generally located in proximity to lakes or streams that are used for irrigation and flooding (Pelletier, Pepin, Gallichand, & Caron, 2016; Caron et al., 2017; Olszewski, Jeranyama, Kennedy, & DeMoranville, 2017; DeMoranville, 2008), but which also makes them potential sources of impaired water quality (Massachusetts Department of Environmental Protection [MA DEP], 2009a; Howes et al., 2014).

The mechanisms that control nutrient transfers from cranberry farms to surface water have not been rigorously evaluated, but some general trends have emerged to describe the wide variability in nutrient losses from cranberry farms. Cranberry farms can be classified based on their connection to surface water: riparian ("flow-through") farms, which Hoekstra, Neill, and Kennedy (2020) described as having a large central channel (usually meandering) with farm units on either side of the channel, and nonriparian farms, which are relatively isolated from surface water and include upland farms. Riparian cranberry farms export up to three times the N and P from nonriparian cranberry farms (MA DEP, 2009b; Howes et al., 2014), although continuously flowing nonriparian farms appear to export similar amounts of N compared with riparian farms (Howes & Teal, 1995; Neill, Jakuba, Kennedy, & DeMoranville, 2017). The seasonal timing of nutrient export from cranberry farms also varies between riparian and nonriparian farms: Howes and Teal (1995) observed high N and P losses from a riparian cranberry farm that were attributed to the manipulation of floodwaters, while Neill et al. (2017) reported 55% to 77% of gross N and P export from three nonriparian farms during nonflood periods. In a multi-year study, Kennedy et al. (2018a) observed elevated N and P export from a nonriparian cranberry farm when extreme summer rainfall (111 mm day⁻¹) coincided with fertilizer application.

As is the case for most peatlands, N export from cranberry farms is governed by the abiotic and biotic processes that produce, consume, and transport N in the unsaturated zone. Cranberry farms are perhaps best described as “cultivated peatlands” (Howes & Teal, 1995; Kennedy et al., 2018b), which consist of anthropogenic sand overlying peat. The sand layer is generally acidic (mean pH = 4.5; Davenport et al., 2003), consists of minor amounts inorganic N (NO₃⁻, <0.2 kg of N ha⁻¹; NH₄⁺, 1.0 to 1.8 kg of N ha⁻¹; Stackpoole, Workmaster, Jackson, & Kosola, 2008; Ballentine et al. 2017), and exhibits low rates of net mineralization and net nitrification (<0.30 kg of N ha⁻¹ day⁻¹; Stackpoole, Workmaster, Jackson, & Kosola, 2008; Ballentine et al, 2017). By comparison, uncultivated peatlands are also acidic (pH < 5) and exhibit relatively high rates of biological uptake of inorganic N (Westbrook & Devito, 2004). As a consequence, N export from peatlands is generally high in organic N relative to inorganic N (Hemond, 1980, 1983; Verry & Timmons, 1982; Worrall, Clay, & Burt, 2012; Hill et al., 2016).

To better understand the processes of nutrient transport in cultivated peatlands, we quantified the amounts and forms of N and P export from a large, 20.2-ha cranberry farm in Plymouth, Massachusetts, which was implicated in a 2010 P total maximum daily load (MA DEP 2009a). We then compiled N loading rates from empirical budget studies for cranberry farms, as well as for croplands and peatlands. We hypothesize that the major N inputs to cranberry farms are similar to those for row crop farms, but that the forms of N outputs differ between these two types of agriculture. In terms of nutrient management, the objective of this study was to refine cranberry farm N and P loading rates and by doing so, elucidate the processes that supply, store, and transport N and P in cranberry farms.

2 | METHODS

2.1 | Study area

The study area is located in the upper Buttermilk Bay watershed on a commercial cranberry farm 13 km south of Plymouth (Howes et al., 2014; Figure 1). Local climate is typical of New England with mild summers (mean July temperature of 22°C), cold winters (mean January temperature of −2°C), and moderate precipitation (2–18% seasonal variation; 1981–2010; https://w2.weather.gov/Climate/xmacis.php?wfo=box, accessed November, 2019). During the study period, precipitation of 1,529 mm year⁻¹ was within 2% of long-term (1981–2010) mean annual precipitation (National Oceanic and Atmospheric Administration, 2014). Although the region experienced a severe drought in 2016–2017, climate conditions were generally "normal" during the study (U.S Drought Monitor, https://droughtmonitor.unl.edu, accessed August, 2019).

Historically, the cranberry farm was drained to White Island Pond, a 118-ha freshwater lake consisting of two basins (Figure 1). The site is one of four cranberry farms located along the shoreline of White Island Pond, which is mostly developed with residential homes (Eichner and Howes, 2011). The site is also one of two active cranberry farms implicated in a 2010 P total maximum daily load (TMDL) for the east basin of White Island Pond (MA DEP, 2009), which assigned a target allocation of 10 kg of P year⁻¹ for the cranberry farm. To comply with the TMDL, agricultural drainage waters were diverted from the lake and pumped to an upland leaching field (Figure 1) with two exceptions, which were discharges to lake in June 2014 and 2017.

The cranberry farm consists of six farm units that are separated by sand berms (or raised dikes) but hydrologically connected via culverts (Figure 1; Table 1). Soils are mapped mostly as Freetown coarse sand and muck (dyic, mesic Typic Haplosaprist; Natural Resources Conservation Service, 2017), with soil test P of 25.9 to 46.8 mg of P kg⁻¹
(Mehlich-3) and total N of 0.2 to 0.8 g of N kg$^{-1}$ at the 0 to 15-cm depth (Table 1; Kennedy, unpublished data). Soil P saturation (i.e., P/[Al + Fe]) ranges from 9 to 10% for the unrenovated farm units and 15% for the renovated unit FF7 (Kennedy, 2019). Soils are planted with native cultivars 'Early Black' and 'Howes' with the exception of a 3.16-ha field, which was replanted with the 'Ben Lear' variety in 2007. Fertilizers are applied annually (i.e., via helicopter) as urea, ammonium sulfate, and phosphorus pentoxide 1 to 3 times per year between May and July, with doses of 34.1 kg of N ha$^{-1}$ year$^{-1}$ for N and 5.6 kg of P ha$^{-1}$ year$^{-1}$ for P (3-year mean; Table 1). Water management includes open ditches for drainage, sprinkler irrigation for soil moisture and frost protection, and periodic flooding for harvest and winter vine protection (Caron et al., 2016; Caron et al., 2017; Jeranyama, DeMoranville, & Kennedy, 2017; Kennedy et al. 2017a; Figure 1).

There is considerable interest in defining cranberry farms based on their hydrologic and edaphic properties (Hoekstra et al., 2020; Howes et al., 2014). Currently, cranberry farms are classified as wetland or upland farms using soil maps (Kennedy et al., 2018b) and as riparian or nonriparian farms using remote sensing techniques (Hoekstra et al., 2020). Within wetland farms, subclasses include marshes and fens or bogs, collectively referred to as peatlands (Mitsch & Gosselink, 2007). As defined by Hoekstra

![Map of study area (a) and cranberry farm and its six farm units (b). Surface water is pumped underground from the north end of unit FF10 to the leaching field. Data sources: 2014 National Geographic Society (a; USGS Topo Map) and 2012 GoogleEarth (b; aerial photograph).](image)

**TABLE 1** Study site characteristics

| Unit   | Area | Cultivar | Yield | pH | P | Al | Fe | P | N |
|--------|------|----------|-------|----|---|----|----|---|---|
| FF7    | 3.16 | BL       | 28.1  | 4.4 | 47 | 185 | 131 | 7.2 | 49.3 |
| FF10   | 3.63 | EB + H   | 14.2  | 4.3 | 26 | 201 | 195 | 5.5 | 32.1 |
| FF12   | 4.89 | EB + H   | 15.0  | 4.2 | 28 | 209 | 188 | 5.2 | 30.4 |
| FF15   | 4.47 | EB + H   | 15.8  | 4.0 | 31 | 202 | 186 | 5.4 | 31.4 |
| FFLT   | 0.30 | H        | 14.7  | n.d. | n.d. | n.d. | 5.2 | 30.9 |
| FF17   | 2.74 | EB       | 14.7  | 4.4 | 29 | 189 | 184 | 5.1 | 30.3 |
| Farm   | 19.2 | M        | 17.1  | 4.2 | 32 | 200 | 178 | 5.6 | 34.1 |

Abbreviations: BL, Ben Lear; EB, Early Black, H, Howes; M, mixed; n.d.; not determined.

aData from Ocean Spray (yield, 2013; fertilizer, 3-year mean from 2011 to 2013).

bData from Kennedy (2019).

c16% BL, 61% EB, and 22% H.
Hydrological inputs and outputs were identified based on earlier studies | channel and connected hydrological features (e.g., streams, ponds, et al. (2020), riparian cranberry farms include a large meandering cThis study.

| TABLE 2 | Field measurements of harvest and winter flood inputs to the cranberry farm |
|---------|---------------------------------------------|
| Year    | Harvest | Winter |
|         | N       | mm year⁻¹ | N       | mm year⁻¹ |
| 2013–2014  | 1       | 442       | 3       | 1,003   |
| 2014–2015  | 1       | 473       | 1       | 267     |
| 2015–2016  | 1       | 530       | 1       | 139     |
| 2016–2017  | 1       | 572       | n.d.    | n.d.    |

Abbreviations: N, number of floods; n.d., not determined.

Data from Kennedy et al. (2017a) but recalculated using the farm area of 20.2 ha.

Unpublished data (C. D. Kennedy).

This study.

et al. (2020), riparian cranberry farms include a large meandering channel and connected hydrological features (e.g., streams, ponds, or wetlands), both up and down gradient of the farm. Based on this classification scheme, the study site is best described as a non-riparian fen, as it comprises peat and receives spring groundwater inflows (see Section 3.1), but lacks a large meandering stream channel.

2.2 | Hydrometric measurements

Hydrological inputs and outputs were identified based on earlier studies of the cranberry farm (Kennedy, 2019; Kennedy, Jeranyama, & Alverson, 2017a; Kennedy, Kleinman, & DeMoranville, 2015) and measured or estimated between 1 May 2017 and 30 April 2018. Daily precipitation data were obtained from a National Weather Service gauge (site 192451) at the University of Massachusetts Cranberry Station in East Wareham, Massachusetts. Errors in annual precipitation data were about 5%, on average, but as high as 20% seasonally (Legates & DeLiberty, 1993; Winter, 1981). Irrigation inputs were measured weekly with mechanical propeller-type flow meters (±2%; McCrometer Propeller Flow Meter Operation manual; available online at mccrometer.com, accessed 9/2019). We measured floodwater with acoustic Doppler-based velocimeters (Isco/Teledyne model 2150) (Table 2), which were installed in culverts and recorded velocity and stage on 15-min intervals, from 2013 to 2016 for the harvest and from 2013 to 2015 for winter vine protection. In our calculations, we used the 2016 harvest flood input and the the 3-year mean winter flood input (Table 2). Error in the harvest flood was based on laboratory performance tests of flow meters (±8%; Heiner and Vermeyen, 2012), whereas that in the winter flood was estimated as the coefficient of variation about the 3-year mean winter flood (±39%; Table 2).

Hydrometric measurements

2.3 | Sample collection

During the study, the main sources for irrigation water were a lake (i.e., White Island Pond), a retention pond, and a shallow well, whereas the sole source for floodwaters was the lake (Figure 1). We collected monthly samples from the lake and retention pond by manually dipping a sample bottle at roughly the depth of the pump intake screen. For floodwater, we collected three lake water samples during the harvest flood and another three samples prior to the winter flood, based on the relatively low nutrient variation of the incoming floodwaters (Kennedy, 2019; Kennedy et al., 2015). We collected samples of surface water discharge as surface (<15-cm depth) grab samples on a weekly basis, with more frequent sampling during flood discharges (66 samples in total).

2.4 | Analytical methods

We collected filtered and unfiltered water samples in 60-ml polyethylene bottles using a plastic syringe and pre-rinsed (0.45 μm) cellulose membrane filters. Total P (TP) and total N (TN) were determined on unfiltered samples, whereas total dissolved P (TDP), dissolved reactive P (DRP), total dissolved N (TDN), nitrate plus nitrite (NO₃⁻), and ammonium (NH₄⁺) were analysed on filtered samples. Concentrations were estimated by difference for dissolved organic N (DON = TDN + NH₄⁺ - NO₃⁻), particulate organic N (PON = TN – TDN), dissolved organic P (DOP = TDP – DRP), and total particulate P (TPP = TP – TDP). All samples were frozen prior to analysis.
Nitrogen and phosphorus concentrations were determined by colorimetry on a flow injection analyser (Lachat, QuickChem 8500 Series 2) at the Pasture Systems and Watershed Management Research laboratory in East Wareham, Massachusetts. Concentrations of TP, TN, TDP, and TDN were digested with an alkaline persulfate reagent in an autoclave (Patton & Kryskalla, 2003). Quantitative check standards (2.0-mg L\(^{-1}\)) of urea, nicotinic acid, trimethyl phosphate, and sodium tripolyphosphate indicated >97% recovery of organic and inorganic compounds. Method detection limits were 10 μg of N L\(^{-1}\) for NH\(_4^+\) and NO\(_3^-\), 20 μg of N L\(^{-1}\) for TDN and TN, 0.2 μg of P L\(^{-1}\) for DRP, and 4 μg of P L\(^{-1}\) for TDP and TP (American Public Health Association, 2005, p. 1-17). Analytical uncertainty in NO\(_3^-\), NH\(_4^+\), and DRP was <8%.

### 2.5 Data analysis

We quantified annual groundwater exchanges with the cranberry farm as the residual term in a water budget equation for a wetland (Kennedy, 2015; Mitsch & Gosselink, 2007):

\[
Q_{gw} = Q_{sw} + ET - Q_f - Q_i \pm \Delta S, \tag{2}
\]

where annual volumes of surface water discharge (\(Q_{sw}\)), ET, precipitation (\(Q_{pr}\)), irrigation (\(Q_{irr}\)), floodwater (\(Q_{f}\)), and storage change (\(\Delta S\)) were used to solve for net groundwater exchange (\(Q_{gw}\)). Methods for \(Q_{sw}\), \(Q_{pr}\), \(Q_{irr}\), and \(Q_{f}\) were described in the previous section, as were daily mean values of \(Q_{gw}\) and ET, which were aggregated and multiplied by the farm area (20.2 ha) to determine volumetric fluxes (Table 1). Annual \(\Delta S\) was calculated as in Kennedy et al. (2018a), and because it was generally low (<5%), was set to zero for water budget calculations.

Assuming steady state with respect to changes in nutrient storage and negligible nutrient loss from the underlying peat layer, a mass balance equation was applied to calculate net TP and TN fluxes from the cranberry farm. We express the nutrient load in surface water discharge (\(F_{sw}\)) as a function of nutrient inputs to the farm:

\[
F_x = c_{x}^t c_f^t F_{fert} + c_{x}^s c_f^s F_{fert} + c_{x}^F c_f^F F_f + c_{x}^{\text{atm}} c_f^{\text{atm}} F_{atm}, \tag{3}
\]

where \(F_{fert}\), \(F_f\), and \(F_f\) are nutrient inputs of fertilizer, floodwater, and irrigation, respectively, and the term \(c_f^x\) is the fraction of the nutrient input \(x\) that is not lost at the surface (\(y = s\)) and in the vadose zone (\(y = vz\)) of the cranberry farm (i.e., the amount of \(F_{fert}\), \(F_f\), and \(F_f\) that contributes to \(Q_{gw}\)). Groundwater flows (\(Q_{gw}\)) and nutrient fluxes (\(F_{gw}\)) may represent an input or output, yet we show in Section 3.1 that \(Q_{gw}\) is a minor (3%) output from the cranberry farm. As such, \(F_{gw}\) was set to zero in Equation (3).

In the case of TP, we assumed \(c_{x} c_f^t = c_{x} c_f^s = c_{x}^{\text{atm}} c_f^{\text{atm}} = 1\), as in previous studies (Neill et al., 2017; Kennedy et al., 2018a). For TN, Valiela et al. (1997) assumed \(c_f^t = 0.61\), \(c_f^s = 0.39\), \(c_{x}^{\text{atm}} = 0.38\), and \(c_f^{\text{atm}} = 0.39\) for agricultural lands. Based on the typically low soil pH of cranberry farms, which would likely inhibit ammonia volatilization from fertilizers (Jones, Koenig, Ellsworth, Brown, & Jackson, 2007; Whitehead & Raistrick, 1990), we assumed \(c_{x}^{\text{atm}} = 1\) but retained the value of \(c_{x}^{\text{atm}} = 0.38\) introduced by Valiela et al. (1997). We calculated \(c_f^t\) and \(c_f^s\) as the fraction of atmospheric dissolved inorganic N (DIN) and dissolved organic N (DON) that leaves soils (i.e., 0.15 for DIN, 0.63 for DON; Lajtha, Seely, & Valiela, 1995; Valiela et al., 1997) and assumed negligible retention of particulate organic N (PON; \(c_f^{\text{atm}}\) = 1). Assuming these nutrient loss coefficients and the relative amounts of DON, DIN, and PON in floodwater and irrigation (Neill et al., 2017; this study), we calculated mean values of \(c_f^t = 0.65\) and \(c_f^s = 0.78\) for TN. Assuming \(c_f^{\text{atm}} = c_{x}^{\text{atm}}\), as in Valiela et al. (1997), we solved for site-specific values of \(c_f^t\) and \(c_f^s\) in Equation 3. The net flux term, \(F_{net}\), which represents N fertilizer export from the cranberry farm, was then calculated as

\[
F_{net} = c_f^t F_{fert} + F_{sw} - 0.65 F_f - 0.78 F_f - 0.38 c_{x}^{\text{atm}} F_{atm}. \tag{4}
\]

We calculated values of \(F_x\) as the product of volumetric discharge (\(Q_{x}\)) and nutrient concentration (\(C_{x}\)) where the subscript \(x\) specifies the hydrologic flux and its concentration of N or P. We calculated cumulative surface water discharge over 4 d, on average, and used the mean nutrient concentration to calculate a flux.

We assumed \(F_{p}\) of 0.18 kg of P ha\(^{-1}\) year\(^{-1}\) for atmospheric deposition of TP, which was calculated as the mean of low (0.05 kg of P ha\(^{-1}\) year\(^{-1}\)) and high (0.3 kg of P ha\(^{-1}\) year\(^{-1}\)) estimates for atmospheric deposition of TP (Cadmus Group, 2008; Reckhow, Beaulac, & Simpson, 1980). Values of \(F_p\) for TN deposition, wet and dry deposition of particulate NH\(_4^+\) and NH\(_3\), and wet and dry deposition of particulate NO\(_3^-\) were obtained from U.S. Environmental Protection Agency’s Clean Air Status and Trends Network (Schwede & Lear, 2014; ftp://ftp.epa.gov/castnet/tdep/grids/). We calculated \(F_p\) for DON and PON based on the fractions DON:TON and PON:TON in precipitation on Cape Cod, Massachusetts (Valiela, Teal, Volkman, Shafer, & Carpenter, 1978).

Cranberry farm area was delineated by MA DEP (2013), which reported the area of 20.2 ha, whereas the farm area provided by the cranberry grower was 19.2 ha. Data analysis was based on the MA DEP (2013) area of 20.2 ha, as in MA DEP (2009a).

Statistical significance is reported at the 90% confidence level using JMP® 12 Pro (SAS; Cary, NC) for macOS (v. 10.14.6). Least squares regression analysis was performed in SigmaPlot (v. 13). Cook’s distance (D; Cook, 1977) was used to measure the influence of each data value on the linear regression fit, and D values greater than \(4/n\) were considered high-influence outliers. Cook’s D was calculated in R (R Core Team, 2015).

### 3 Results and Discussion

#### 3.1 Water budget

Hydrological inputs to the cranberry farm totalled 2,854 mm year\(^{-1}\), which consisted of 54% precipitation, 29% floodwater, and 18%
irrigation (Table 3). Hydrological outputs of surface water and ET were 2,764 mm year\(^{-1}\), indicating minor net groundwater recharge (−91 mm year\(^{-1}\)) on an annual basis (Table 3). Of hydrological outputs, 75% was surface water, 22% was ET, and 3% was groundwater. 

Irrigation totalled 507 mm year\(^{-1}\), of which 49% was applied for soil moisture between 1 June to 15 September, 2017 (Table 3, Figure 2). Soil moisture irrigation of 251 mm year\(^{-1}\) was 48% and 12% higher than in 2014 and 2015, respectively (Kennedy et al., 2017a), perhaps due to changes in management following the sale of the farm in 2016 (Walter Morrison, pers. comm., 2016). Managed hydrological inputs (i.e., irrigation and floodwater) to the cranberry farm totalled 1,325 mm year\(^{-1}\) (Tables 2 and 3). 

Seasonally, ET resulted in the loss of 623 mm year\(^{-1}\) from the farm, with the majority (82%) occurring in the spring and summer (Table 3, Figure 2). Precipitation inputs totalled 1,529 mm year\(^{-1}\), which were mostly (55%) winter and spring rainfall. Monthly values of P−ET ranged from 22 to 168 mm year\(^{-1}\) between August and May and were −4 and −42 mm year\(^{-1}\) in June and July, respectively. From April to September, positive values of P−ET accounted for 15 to 52% of monthly surface water discharges, which included 52% and 50% of precipitation in August and September, respectively. Surface water discharge averaged 14 L s\(^{-1}\) (217 gal min\(^{-1}\)) but was highly variable with 2 to 6-week periods when the discharge pump was turned off, including several days in October due to the cranberry harvest, 2 weeks in December and all of January due to the winter flood, and 4 weeks in February and March due to pump maintenance and repairs (Figure 3).

Because \(Q_{gw}\) and \(\Delta S\) could not be separated on a monthly basis, we calculated the combined net water flux (\(Q_{gw} + \Delta S\)) as the difference between monthly hydrological outputs and inputs (Figure 4). Hemond (1980) calculated \(\Delta S\) as the difference in monthly hydrological outputs and inputs for “Thoreau’s Bog,” a nearby uncultivated Sphagnum bog. In our analysis, Thoreau’s Bog served as a reference point to assess sources of variation in monthly values of \(Q_{gw} + \Delta S\) for the cranberry farm. Monthly differences in \(Q_{gw} + \Delta S\) between the cranberry farm and Thoreau’s Bog ranged from −47 to 13 mm year\(^{-1}\) with three exceptions: April (384 mm year\(^{-1}\)), May (89 mm year\(^{-1}\)), and October (−271 mm year\(^{-1}\)). The highly negative difference in \(Q_{gw} + \Delta S\) between May and August was paralleled by increases in \(Q_{gw}\) and \(\Delta S\) due to regional patterns of elevated spring groundwater discharge to the cranberry farm (Li, Rodell, & Famiglietti, 2015). In contrast, the highly negative difference in \(Q_{gw} + \Delta S\) for October was consistent with harvest floodwaters as sources of groundwater recharge (Kennedy, 2015; Kennedy, 2019; Masterson, Carlson, & Walter, 2009).

### 3.2 Nutrient concentrations

Nitrogen concentrations in hydrologic inputs were generally lower than those in surface water discharge, with the exception of particulate forms of N and P (Table 4). Concentrations of TN in hydrological inputs were, on average, composed of 41% DON, 27% NH\(_4\)\(^+\), 26% PON, and 6% NO\(_3\)\(^-\), which were consistent with hydrological inputs to cranberry farms in Massachusetts and Wisconsin (Stackpoole et al., 2011; Neill et al., 2017; Kennedy et al., 2018a). Monthly concentrations of TN in hydrological inputs from the lake ranged from 30 to 1,398 \(\mu\text{g}\) of N L\(^{-1}\), with higher concentrations between May and August (284-1,398 \(\mu\text{g}\) of N L\(^{-1}\)) than September and April (125-265 \(\mu\text{g}\) of N L\(^{-1}\)). Mean concentration of TN in the irrigation pond (mean = 545 \(\mu\text{g}\) of N L\(^{-1}\)) was higher and more variable than the lake (mean = 298 \(\mu\text{g}\) of N L\(^{-1}\); Figure 1), perhaps due to differences in residence (or turnover) time (Oyvind Kaste, Stoddard, & Henriksen, 2003).

### TABLE 3 Hydrological inputs and outputs for the cranberry farm in 2017–2018

| Water flux (mm year\(^{-1}\)) | Input | Output |
|-------------------------------|-------|--------|
| Precipitation, \(Q_p\)        | 1,529 (54) |        |
| Irrigation, \(Q_i\)           | 507 (18)  |        |
| Floodwater, \(Q_f\)          | 819 (29)  |        |
| Surface water, \(Q_{sw}\)    | 2,140 (75) |        |
| Groundwater, \(Q_{gw}\)      | 91 (3)   |        |

Note: Percent of total inputs or outputs are in parentheses.
Mean concentration of TN in surface water was 28% to 223% higher than floodwater and irrigation, respectively, suggesting that the farm was a source of N to surface water. The composition of TN in surface water was 42% NH$_4^+$, 37% DON, 15% PON, and 5% NO$_3^-$, which differed from high-NO$_3^-$ drainage waters from row crop farms such as wheat, corn, and soybeans (Pellerin et al., 2006; Goolsby & Battaglin, 2001) but was similar to relatively low NO$_3^-$ concentrations in surface water discharge from other cranberry farms (Niell et al., 2017; Howes & Teal, 1995). Seasonal variation consisted of relatively high TN in summer and autumn (mean: 1.013 and 716 μg of N L$^{-1}$, respectively) and low TN in winter and spring (mean: 259 and 300 μg of N L$^{-1}$, respectively) surface water discharges. Kennedy et al. (2018a) also observed elevated summer TN due to the timing of N fertilizer applications in the first year of a 2-year study. In this study, peak levels of TN in surface water coincided with past fertilizer applications in early July (Figure 3). We suspect that the summer "spike" in TN concentration was, at least partly, related to fertilizer additions of urea N, which is the most common and cost-effective N fertilizer used in cranberry agriculture. If so, the forms of N (i.e., 62% NH$_4^+$, 28% DON, 5% PON, and 5% NO$_3^-$) would imply incomplete mineralization of urea N and minimal nitrification of ammonium N. Phosphorus concentrations in hydrological inputs were highly variable with 5–17 times higher TP concentrations in irrigation water than incoming floodwaters (Table 4). TP concentrations of hydrological inputs were predominantly composed of organic P (85–95%, DOP + TPP), with small but significant levels of DRP (5–15%). Mean TP and DRP concentrations of surface water discharge were 2 and 7 times higher than hydrological inputs, respectively. These results aligned with those from focused investigations of the winter and harvest floods, which showed higher TP concentrations in surface water discharge compared with incoming floodwaters (Kennedy, 2019; Kennedy et al., 2015). Seasonally, the highest concentrations of P, particularly DRP, were observed in autumn and associated with the cranberry harvest (Figure 3). The pattern of increasing concentration of DRP with discharge of the harvest flood was consistent with the physical leaching of P released from anoxic sediments to soil pore water, which is a well-documented mechanism in both wetland and agricultural environments (Patrick & Mahapatra, 1968; DeMoranville, Howes, Schlezinger, & White, 2009; Kinsman-Costello et al., 2014; Kennedy
Summer fertilizer additions also appeared to increase P concentrations in surface water (Figure 3). For instance, the July fifth "spike" in TP concentration was composed of 36% DRP, 35% TPP, and 29% DOP. These forms of P are somewhat at odds with the application of inorganic P fertilizers, possibly due to rapid sorption and biological uptake of phosphate in agricultural soils (Jarvie et al., 2005; McDowell, Sharpley, & Folmar, 2003). Nevertheless, summer peaks of P and N concentrations in surface water occurred when fertilizers were commonly applied to the cranberry farm (Figure 3).

3.3 Nutrient loads

Annual TN loads in hydrological inputs were 101.6, 30.7, 23.8, and 9.0 kg of N year\(^{-1}\) for atmospheric deposition, irrigation water, harvest floodwater, and winter floodwater, respectively, and were 44% to 85% in the form organic N (DON + PON) with the exception of atmospheric N, which was 72% DIN (Figure 5). In total, hydrologic inputs added 165.2 kg of N year\(^{-1}\) compared with 190.2 kg of N year\(^{-1}\) discharged by surface water (Figure 5). The N load in surface water discharge was composed of 41% NH\(_4^+\), 30% DON, 22% PON, and 6% NO\(_3^-\). Seasonally, TN loads in surface water discharge were highest in summer (71.6-kg N), lowest in winter (15.5-kg N), and roughly mid range in spring (45.8-kg N) and autumn (57.2-kg N). Within season, N forms in surface water loads were highly variable, with DON and NH\(_4^+\) collectively representing 82% of summer and autumn TN export and DON and PON accounting for about half of winter TN export (Figure 5).

Seasonal patterns in TN export could be broadly tied to climate variation and regional trends in agricultural management (Figure 6). We observed a generally weak relationship between seasonal TN export and surface water discharge, with the highest value of TN export occurring in the driest season (\(R^2 = .13, p = .64; N = 4\)). However, when seasonal values of TN export were regressed against the respective TN concentrations, a more significant linear relationship was observed (\(R^2 = .77; p = .12; N = 4\)). A first-order interpretation of these results is that seasonal differences in TN export are tied to increases in N pools that are susceptible to transport. Increases in more mobile forms of N may be related to warmer summer temperatures that facilitate N transformation processes in the unsaturated zone, particularly mineralization of organic N to NH\(_4^+\) (Davenport & DeMoranville, 2004). Given the timing of fertilizer applications in July, which alone accounted for 32% of the annual TN load export, fertilizer management was also likely a source of elevated N loss.

### TABLE 4

|          | N  | NH\(_4^+\) | NO\(_3^-\) | DON | PON | TN  | DRP | DOP | TPP | TP |
|----------|----|------------|------------|-----|-----|-----|-----|-----|-----|----|
| Precipitation\(^a\) | n.a. | 162 | 116 | 99 | 7 | 384 | 14 | n.d. | n.d. | 14 |
| Irrigation | 16 | 91 | 15 | 199 | 164 | 468 | 39 | 79 | 136 | 254 |
| Harvest flood | 2 | 102 | 13 | 61 | 29 | 206 | 2 | 5 | 9 | 15 |
| Winter flood | 3 | 31 | 22 | 88 | 23 | 164 | 2 | 45 | 7 | 54 |
| Surface water | 66 | 252 | 31 | 224 | 91 | 598 | 96 | 96 | 62 | 254 |

Abbreviations: DON, dissolved organic N; DOP, dissolved organic P; DRP, dissolved reactive P; N, sample size; n.a., not applicable; n.d., not determined; PON, particulate organic N; TN, total N; TP, total P; TPP, total particulate P.

\(^a\)Calculated as the modelled atmospheric deposition (see text) divided by the measured annual precipitation.
High autumn N export was consistent with seasonal N losses from a riparian cranberry farm (Figure 6), which were linked to enhanced foliar leaching, disturbance, and anoxic release of N during the harvest flood in October (Howes & Teal, 1995). Notably, 38% of the annual DON export occurred in October, possibly as a consequence of the cranberry harvest.

Hydrological inputs of TP were predominantly derived from irrigation water, which accounted for 72% of all P inputs to the cranberry farm. In contrast, Kennedy et al. (2018a) observed that P inputs were mostly derived from incoming floodwaters, which were transferred from adjacent cranberry fields rather than a lake. In total, hydrological inputs added 28.26 kg of P year\(^{-1}\) to the cranberry farm as 46% TPP, 29% DOP, and 25% DRP. Surface water export of 89.64 kg of P year\(^{-1}\) was about three times higher than hydrological inputs of TP, indicating leaching of soil P to surface water. Of this amount, 39% was DOP, 37% was DRP, and 24% was TPP. In general, monthly variation in TP export from the cranberry farm closely mirrored that for TN, with the exception of the months of July and October (Figure 7), which may be related to higher fertilizer additions of N compared with P in July and greater anaerobic dissolution of P than N compounds in October (Table 1). Thus, the factors that contributed to TN export generally appeared to have a similar effect on TP, with the exceptions of fertilizer application and the harvest flood (Figure 6).

### 3.4 Forms of dissolved nitrogen export

Total dissolved N was the dominant fraction of TN export from the cranberry farm, accounting for 78% of the TN load in surface water (Figure 5). Of TDN export, 54% was NH\(_4^+\), 39% was DON, and 7% was NO\(_3^-\). Based on TDN loading rates for five cranberry farms, mean TDN export from cranberry agriculture consists of 46% NH\(_4^+\), 41% DON, and 13% NO\(_3^-\) (Table 5), which is consistent with field observations of low soil NO\(_3^-\) in cranberry farms (Stackpoole et al., 2008; Ballentine et al., 2017). In most croplands, NO\(_3^-\) is the dominant form of TDN export (Pellerin et al., 2006), in part because it is the most mobile form of N in the environment (Böhlke, 2002). In a study of four agricultural watersheds within the Mississippi River Basin, mean TDN export was 75% NO\(_3^-\), 22% DON, and 3% NH\(_4^+\) (Goolsby & Battaglin, 2001), which differed from cranberry farms for NO\(_3^-\) and NH\(_4^+\) but not for DON (Figure 8). In the Midwest, about 90% of synthetic fertilizers are applied as NH\(_4^+\) (Cao, Lu, & Yu, 2018), indicating high rates of nitrification relative to NH\(_4^+\) transport through the unsaturated zone (Böhlke, 2002).

In contrast, low rates of net nitrification have been observed in peatlands (Devito, Westbrook, & Schiff, 1999; Hayden & Ross, 2005; Roswall & Granhall, 1980), at least partly due to rapid uptake of NO\(_3^-\) (Westbrook & Devito, 2004). For example, mean TDN export was 58% DON, 25% NH\(_4^+\), and 17% NO\(_3^-\) for peatlands in the United States and United Kingdom (Verry & Timmons, 1982; Hemond, 1980, 1983; Worrall et al., 2012; Hill et al., 2016), suggesting minimal production or rapid consumption of NO\(_3^-\).
compared with row crop farms such as wheat, corn, and soybeans (Figure 8). Although N transformation processes are still poorly understood in cranberry farms, our results suggest that they may resemble those in peatlands, despite the management effects of fertilization and artificial drainage.

### 3.5 Management implications

Accurate estimates of N loading rates from cranberry agriculture are essential for comprehensive N management plans to protect coastal water quality in southeastern Massachusetts (Howes et al., 2014; Williamson et al., 2017). We calculated TN values of $F_{\text{net}}$ that ranged from 6.8 to 23.9 kg of N ha$^{-1}$ year$^{-1}$ (mean = 14.0 kg of N ha$^{-1}$ year$^{-1}$; N = 6), or were equal to 23.9 kg of N ha$^{-1}$ year$^{-1}$ for riparian cranberry farms and 12.1 kg of N ha$^{-1}$ year$^{-1}$ for nonriparian cranberry farms (mean; N = 5). We found that the two highest values of TN export were related to less retention of N fertilizer in the vadose zone of riparian cranberry farms or nonriparian cranberry farms that receive high groundwater inputs (Table 5). By comparison, Howes et al. (2014) cite the nonriparian farm N loading rate of 7.0 kg of N ha$^{-1}$ year$^{-1}$, a value that does not include farms that receive high groundwater inputs (i.e., $Q_{\text{gw}} < 0.2$ m year$^{-1}$ or <6% of all hydrological inputs; see DeMoranville, 2015; DeMoranville & Howes, 2005).

Based on these results, we recommend that future N TMDLs,
especially those developed for the major agricultural watersheds in southeastern Massachusetts (i.e., Sippican River, Weweantic River, and Wareham River), classify cranberry farm N loading rates based on riparian and nonriparian hydrology (Hoekstra et al., 2020) and the extent of groundwater exchanges in nonriparian farms (Neill et al., 2017).

Unlike most agricultural crops, cranberry farms export high amounts of DON and NH₄⁺ relative to NO₃⁻ (Figure 8). Recently, a state-funded programme was established to support wetland restoration of retired cranberry farms (MA DER, 2019). The programme was motivated, in part, by the potential to enhance denitrification through wetland restoration of retired cranberry farms (Ballentine et al., 2017). However, the amount of NO₃⁻ removed through wetland restoration of retired cranberry farms has not been quantified. As a first-order estimate, we calculated gross NO₃⁻ export from five cranberry farms (i.e., \( F_{\text{net}} \) in Equation 4), with values ranging from 0.2 to 6.2 kg of N ha⁻¹ year⁻¹ (mean ± SD = 2.2 ± 2.5 kg of N ha⁻¹ year⁻¹; Neill et al., 2017; Howes & Teal, 1995; Table 5). Although the microbial capacity to consume NO₃⁻ may limit the amount of NO₃⁻ available for denitrification (Devito and Westbrook, 2004; Stark & Hart, 1997), as suggested by low NO₃⁻ export, some cranberry farms represent seepage faces for high-NO₃⁻ groundwater inflows, particularly riparian cranberry farms located in low-lying areas of watersheds where septic effluent is a major N input to groundwater (e.g., Coonamessett Bogs on Cape Cod). As such, state-funded wetland restoration projects should target retired riparian cranberry farms that receive high groundwater inputs of NO₃⁻, which appears to be the principal limiting factor of denitrification in these peatland systems.

In addition to coastal watershed N management plans, P regulations have been implemented to decrease agricultural P loadings to White Island Pond by 86% (MA DEP 2009a), with our estimate of \( F_{\text{net}} \) (3.04 kg of P ha⁻¹ year⁻¹) within 11% of the TP loading rate used to establish the White Island Pond P TMDL (Mattson 2015; MA DEP 2009a). Since the implementation of the White Island Pond P TMDL in 2010, drainage waters have been diverted away from the lake to an upland leaching field with two notable exceptions: brief (<6 hr) June discharges to the lake in 2014 and 2017. For the month of June, we measured TP export of 8.8-kg P from the cranberry farm (Figure 6). Although this does not account for the ~6-hr June discharge to the lake, it does suggest that most June discharges from the cranberry farm are below the TMDL allocation of 10-kg P year⁻¹ (MA DEP 2009a), including those that occurred in 2014 and 2017.

Wet (flood) harvesting is practiced by ~90% of cranberry growers in Massachusetts (DeMoranville, 2008), but is also a significant source of P loss from cranberry farms. In this study, we estimate that wet harvesting contributes up to 32% of the annual TP load in surface water. Although growers are unlikely to replace wet harvesting, cranberry production is uniquely adapted to remedial strategies along the lines of containment, treatment, and trapping (Aslan & Cakici, 2007; Blowes, Robertson, Ptacek, & Merkley, 1994). Given that about two thirds of the harvest flood P loss was in the form of organic P (Figure 5), surface applications of P-sorbing materials that promote flocculation, such as aluminium sulfate, are well suited to sequester P from the harvest flood (Kennedy et al., 2017b). In addition, the iron-rich soils of cranberry farms may function as natural filter beds, depending on the P concentration and infiltration rate of the harvest flood (Kennedy, 2019).

4 | CONCLUSIONS

We find that cranberry farms export significantly less NO₃⁻ than would be expected from agricultural crops that receive regular additions of N fertilizers. In uncultivated peatlands, low NO₃⁻ export has been connected to rapid biological consumption of NO₃⁻ and NH₄⁺ (Westbrook & Devito, 2004), which would also serve to reduce nitrification rates. We believe similar soil biogeochemical processes may occur in cranberry farms, but detailed studies on N cycling in cranberry farms are lacking. Field measurements of gross nitrification and mineralization rates will fill a critical gap in our understanding of the fate of N in cranberry farms.

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DATA AVAILABILITY STATEMENT

The data that support the findings of this study are available from the corresponding author upon reasonable request.

ORCID

Casey D. Kennedy https://orcid.org/0000-0003-0707-8436

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