Four decades of water quality change in the upper San Francisco Estuary

Marcus W. Beck[1]*, Thomas W. Jabusch[2], Philip R. Trowbridge[2], David B. Senn[2]

*Corresponding author: marcusb@sccwrp.org

[1]*USEPA National Health and Environmental Effects Research Laboratory*
Gulf Ecology Division, 1 Sabine Island Drive, Gulf Breeze, FL 32561

Current address: Southern California Coastal Water Research Project
3535 Harbor Blvd, Suite 110, Costa Mesa, CA 92626

[2]*San Francisco Estuary Institute*
4911 Central Ave, Richmond, CA 94804

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Abstract

Quantitative descriptions of chemical, physical, and biological characteristics of estuaries are critical for developing an ecological understanding of drivers of change. Historical trends and relationships between key species of dissolved inorganic nitrogen (ammonium, nitrate/nitrite, total) from the Delta region of the San Francisco Estuary were modeled with an estuarine adaptation of the Weighted Regressions on Time, Discharge, and Season (WRTDS). Analysis of flow-normalized data revealed trends that were different from those in the observed time-series. Flow-normalized data exhibited changes in magnitude and even reversal of trends relative to the observed data. Modelled trends demonstrated that nutrient concentrations were on average higher in the last twenty years relative to the earlier periods of observation, although concentrations have been slowly declining since the mid-1990s and early 2000s. We further describe mechanisms of change with two case studies that evaluated 1) downstream changes in nitrogen following upgrades at a wastewater treatment plant, and 2) interactions between biological invaders, chlorophyll, macro-nutrients (nitrogen and silica), and flow in Suisun Bay. WRTDS results for ammonium trends showed a distinct signal as a result of upstream wastewater treatment plant upgrades, with specific reductions observed in the winter months during low-flow conditions. Results for Suisun Bay showed that chlorophyll *a* production in early years was directly stimulated by flow, whereas the relationship with flow in later years was indirect and influenced by grazing pressure. Although these trends and potential causes of change have been described in the literature, results from WRTDS provided an approach to test alternative hypotheses of spatiotemporal drivers of nutrient dynamics in the Delta.

1 Introduction

Understanding drivers of water quality change in estuaries depends on accurate descriptions of source inputs. The Sacramento - San Joaquin River Delta (hereafter 'Delta') is a mosaic of inflows in the upper San Francisco Estuary (SFE) that receives and processes nutrient inputs from primarily urban sources, in addition to agricultural sources (Jassby and Cloern 2000, Jassby et al. 2002, Jassby 2008). Although water quality conditions in the SFE symptomatic of eutrophication have historically been infrequent, recent responses to stressors suggests that ecosystem condition may be changing from past norms. Changes in phytoplankton biomass and composition, water clarity changes from sediment alteration, increases in harmful cyanobacterial blooms (*Microcystis aeruginosa*), increases in non-native macrophytes, and periodic events of low dissolved oxygen have been a recent concern for management of the Delta (Lehman et al. 2005, Santos et al. 2009, Hestir et al. 2013, Lehman et al. 2015, Dahm et al. 2016, ASC 2017). Although these changes are linked to drivers at different spatial and temporal scales, describing inputs from the Delta is critical to understand downstream effects.

Rates of primary production in coastal habitats are often defined by nutrient concentrations, although a simple relationship between enrichment and water quality changes can be difficult to determine (Cloern 2001). Nutrient concentrations are generally non-limiting for phytoplankton growth in the upper SFE, whereas light availability is the primary limiting factor preventing accumulation of phytoplankton biomass (Cole and Cloern 1984, Alpine and Cloern 1988). Grazing pressure from pelagic fishes and benthic invertebrates can also reduce phytoplankton during periods of growth (Nichols 1985, Jassby 2008, Kimmerer and Thompson 2014). Moreover, changes in flow management practices compounded with climate variation have

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altered flushing rates and turbidity as key factors that moderate phytoplankton growth in the system (Alpine and Cloern 1992, Lehman 2000, Wright and Schoellhamer 2004, Canuel et al. 2009). Glibert et al. (2014) attributed recent phytoplankton blooms in Suisun Bay to a drought, during which residence times and nitrification rates increased. Speciation changes in the dominant forms of nitrogen confound a direct interpretation of links with phytoplankton blooms and the relationships between nutrients and primary production in the upper estuary are not completely understood. Although phytoplankton concentrations have been relatively consistent in recent years in Suisun Bay, biomass trends in the Delta are mixed (Jassby 2008, ASC 2017). Descriptions of nitrogen trends over several decades could be used to better understand long-term and more recent changes, particularly in the context of primary production and physical drivers of change (Dahm et al. 2016).

Long-term monitoring data are powerful sources of information that can facilitate descriptions of water quality change. A comprehensive water quality monitoring program has been in place in the upper SFE for several decades (Fig. 1, IEP 2013). Although these data have been used extensively (e.g., Lehman 1992, Jassby 2008, Glibert 2010), water quality trends covering the full spatial and temporal coverage of the Delta have not been systematically evaluated. Quantitative descriptions of nutrient dynamics are challenging given multiple sources and the volume of water that is exchanged with natural and anthropogenic processes. An evaluation using mass-balance models to describe nutrient dynamics in the Delta demonstrated that the majority of ammonium entering the system during the summer is nitrified or assimilated, whereas a considerable percentage of total nitrogen load to the Delta is exported (Novick et al. 2015). Seasonal and annual changes in the delivery of water inflows and water exports directly from the system can also obscure trends (Jassby and Cloern 2000, Jassby 2008). It is important to

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70 consider these variable effects to characterize different trends in nitrogen forms.

71   Formal methods for trend analysis are required to describe water quality changes that vary

72 by space and time. As a practical approach for water quality evaluation, trend analysis of

73 ecosystem response indicators often focuses on tracking the change in concentrations or loads of

74 nutrients over many years. Response indicators can vary naturally with changing flow conditions

75 and may also reflect long-term effects of management or policy changes. Similarly, nutrient

76 trends that vary with hydrologic loading can vary as a function of utilization rates by primary

77 producers or decomposition processes (Sakamoto and Tanaka 1989, Schultz and Urban 2008,

78 Harding et al. 2016). Concentration of chlorophyll *a* (chl-*a*) as a measure of phytoplankton

79 response to nutrient inputs can follow seasonal patterns with cyclical variation in temperature and

80 light changes throughout each year, whereas annual trends can follow long-term variation in

81 nutrient inputs to the system (Cloern 1996, Cloern and Jassby 2010). Describing the relationship

82 of a water quality variable as changing (chemodynamic) or invariant (chemostatic) with flow can

83 isolate components, such as nutrient inputs, for a more direct assessment of causal factors (Wan

84 et al. 2017).

85   The Weighted Regressions on Time, Discharge, and Season (WRTDS) approach was

86 developed in this context and has been used to characterize decadal trends in river systems

87 (Hirsch et al. 2010, Sprague et al. 2011, Medalie et al. 2012, Hirsch and De Cicco 2014, Pellerin

88 et al. 2014, Zhang et al. 2016). The WRTDS method has been adapted for trend analysis in tidal

89 waters, with a focus on chl-*a* trends in Tampa Bay (Beck and Hagy III 2015) and the Patuxent

90 River Estuary (Beck and Murphy 2017). Although the WRTDS method has been effectively

91 applied to describe changes in freshwater systems, use in tidally influenced systems has not been

92 as extensive. Application of WRTDS to describe trends in estuaries could reveal new insights

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given the disproportionate effects of physical drivers, such as flow inputs and tidal exchange, on water quality. The effects of biological drivers may also be more apparent because hydrological effects can be removed by WRTDS. As such, application of WRTDS models for trend analysis could facilitate a broader discusion on the need to focus beyond nutrients to develop integrated plans for water quality management.

The goal of this study was to provide a comprehensive description of nutrient trends in the Delta and Suisun Bay over the last forty years. This information can inform the understanding of ecosystem response dynamics and potential causes of water quality change. The specific objectives were to 1) quantify and interpret trends over four decades at ten stations in the Delta and Suisun Bay, including annual, seasonal, and spatial changes in nitrogen forms and response to flow variation, and 2) provide detailed descriptions of two case studies in the context of conceptual relationships modeled with WRTDS. The second objective evaluated two specific water quality stations to demonstrate complexities with nutrient response to flow, effects of nutrient-related source controls on ambient conditions, and effects of biological invasion by benthic filter feeders on nutrient cycling and primary production. Our general hypothesis was that the results were expected to support previous descriptions of trends in this well-studied system, but that new insight into spatial and temporal variation in response endpoints was expected, particuarly in flow-normalized model predictions.

2   *Materials and Methods*

2.1   Study system

The Delta region drains a 200 thousand km^2 watershed into the SFE, which is the largest estuary on the Pacific coast of North America. The watershed provides water to over 25 million

people and irrigation for 18 thousand km^2 of agricultural land. Water enters the SFE through the Sacramento and San Joaquin rivers that have a combined inflow of approximately 28 km^3 per year, with the Sacramento accounting for 84% of inflow to the Delta. The SFE system includes the Delta and subembayments of San Francisco Bay (Fig. 1). Water dynamics in the SFE and Delta are governed by inflows from the watershed, tidal exchange with the Pacific Ocean, and water withdrawals for municipal and agricultural use (Jassby and Cloern 2000). Seasonally, inflows from the watershed peak in the spring and early summer from snowmelt, whereas consumption, withdrawals, and export have steadily increased from 1960 to present, but vary depending on inter-annual climate effects (Cloern and Jassby 2012). Notable drought periods have occurred from 1976-1977, 1987-1992, and recently from 2013-2015 (Cloern 2015).

Orthophosphate (PO_4^{3-}) and dissolved inorganic nitrogen (DIN) enter the Delta primarily through the Sacramento and San Joaquin rivers and from municipal wastewater treatment plant (WWTP) inputs. Annual nutrient export from the Delta region has been estimated as approximately 30 thousand $kg\,d^{-1}$ of total nitrogen (varying with flow, Novick et al. 2015), with 90% of ammonium (NH_4^+) originating solely from the Sacramento Regional WWTP (Jassby 2008). Although nitrogen and phosphorus inputs are considerable, primary production is relatively low and not nutrient-limited (Jassby et al. 2002, Kimmerer et al. 2012).

2.2   Data sources

Nutrient time series of monthly observations from 1976 to 2013 were obtained for ten active sampling stations in the Delta (Fig. 1 and Table 1, IEP 2013). Stations were grouped by location in the study area for comparison: *peripheral* Delta stations C3 (Sacramento inflow), C10 (San Joaquin inflow), MD10, and P8; *interior* Delta stations D19, D26, and D28; and *Suisun*

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stations D4, D6, and D7. These stations cover all of the major inflows and outflows to the Delta and were selected for analysis based on the continuity of the period of observation (Jabusch and Gilbreath 2009). Although many other stations are available for the region, the stations were chosen because they represent a consistent long-term dataset generated by a single program and they capture dominant seasonal and annual modes of nitrogen variability characteristic of the region (Jabusch et al. 2016). Time series were complete for all stations except for an approximate ten year gap from 1996-2004 for D19. Data were minimally processed, with the exception of averaging replicates that occurred on the same day. The three nitrogen forms that were evaluated were ammonium, nitrite/nitrate, and DIN (as the sum of the former two). Less than 3% of all observations were below the detection limit (left-censored), although variation was observed between analytes and location. The ammonium time series had the most censored observations at sites C10 (25.4% of all observations), MD10 (18.1%), D28 (17.8%), D19 (12%), and D7 (7.9%).

WRTDS models require flow data paired with nutrient data. At the Delta stations, daily flow estimates were matched with the corresponding sample dates for the nutrient data. Daily flow estimates were obtained from the Dayflow software program (IEP 2016). The Sacramento daily flow time series was used to account for flow effects at C3, D19, D26, D28, and MD10, and the San Joaquin time series was used for C10 and P8 based on station proximity to each inflow. Given the complexity of inflows and connectivity of the system, only the inflow estimates from the Sacramento and San Joaquin rivers were used as measures of freshwater influence at each station. Initial analyses indicated that model fit was not significantly improved with flow estimates from locations closer to each station, nor was model fit improved using lagged times series. Salinity was used as a proxy for flow at the Suisun Bay sites D4, D6, and D7 where tidal influences were much stronger. Salinity has been used as a tracer of freshwater influence for the

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application of WRTDS models in tidal waters (Beck and Hagy III 2015). Models were evaluated

using salinity at the Delta stations, but performance was reduced relative to models that used daily

flow estimates.

2.3   Analysis method and application

A total of thirty WRTDS models were created, one for each nitrogen form at each station.

The functional form of WRTDS is a simple regression (Hirsch et al. 2010) that models the

log-transformed response variable as a function of time, flow, and season:

$$\ln{(N)} = \beta_0 + \beta_1 t + \beta_2 \ln{(Q)} + \beta_3 \sin{(2\pi t)} + \beta_4 \cos{(2\pi t)} \tag{1}$$

where N is one of three nitrogen analytes, time t is a continuous variable as decimal time to

capture the annual (β_1) or seasonal (β_3, β_4) trend, and Q is the flow variable (either flow or

salinity depending on station).

The WRTDS model is a moving window regression that fits unique parameters (i.e., β_0,

..., β_4) at each observation point in the time series (n ranging from 433 at D19 to 571 at C3).

Rather than fitting a global model to the entire time series, one regression is fit to every

observation. Observations within a window for each regression are weighted relative to annual,

seasonal, and flow distances from the observation at the center of the window. Observations with

distances farther from the center (i.e., greater time and different flow values from the center) have

less weight during parameter estimation for each regression. This approach allows for a type of

smoothing where the observed fit is specific to the data characteristics within windows. Models

applied herein were based on a tidal adaptation of the original method that can use either flow or

salinity estimates as nutrient predictors (Beck and Hagy III 2015). All models were fit to describe

the conditional mean response using a weighted Tobit model for left-censored data below the

detection limit (Tobin 1958). Model predictions were evaluated as monthly values or as annual

values that averaged monthly results within each water year (October to September). All analyses

used the WRTDStidal package for the R statistical programming language (Beck 2017, RDCT (R

Development Core Team) 2017). The default model fitting procedures were used that set

half-window widths as six months for seasonal weights, ten years for annual weights, and half the

range of salinity or flow in the input data for Q weights.

A hallmark of the WRTDS approach is the description of flow-normalized trends that are

independent of variation from freshwater inflows (Hirsch et al. 2010) or tidal variation (Beck and

Hagy III 2015). Flow-normalized trends for each analyte at each station were used to describe

long-term changes in different annual and seasonal periods. Flow-normalization predictions for

each month of each year were based on the average of predictions for flow values that occur in the

same month across all years, weighted within each specific month and year for every observation.

Flow-normalized trends in each analyte were summarized as both medians and percent changes

from the beginning to end of annual groupings from 1976-1995 and 1996-2013, and seasonal

groupings of March-April-May (spring), June-July-August (summer),

September-October-November (fall), and December-January-February (winter) within each

annual grouping. Annual groupings were chosen as approximate twenty year midpoints in the

time series and seasonal groupings were chosen to evaluate inter-annual changes while keeping

season constant.

Trends in each annual and seasonal grouping were based on seasonal Kendall tests of the

flow-normalized predictions. This test is a modification of the non-parametric Kendall test that

accounts for variation across seasons in the response variable (Hirsch et al. 1982, Millard 2013).

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Results from the test can be used to evaluate the direction, magnitude, and significance of a monotonic change within the period of observation. The estimated rate of change per year is also returned as the Theil-Sen slope and was interpreted as the percent change per year when divided by the median value of the response variable in the period of observation (Jassby 2008). Trends in annual groupings were based on all monthly observations within relevant years, whereas seasonal groupings were based only on the relevant months across years. Seasonal Kendall tests were also used to describe trends in the observed data. These trends were compared with those based on the flow-normalized trends to evaluate potential differences in conclusions caused by flow effects.

2.4   Selected examples

Two stations were chosen to demonstrate use of WRTDS to develop a more comprehensive description of decadal trends in the Delta. The selected case studies focused on 1) effects of wastewater treatment upgrades upstream of P8, and 2) effects of biological invasion on nutrient dynamics in Suisun Bay using observations from D7. Each case study is built around hypotheses that results from WRTDS models were expected to support, both as a general description and for additional testing with alternative methods.

2.4.1   Effects of wastewater treatment

Significant efforts have been made in recent years to reduce nitrogen loading from regional WWTPs given the disproportionate contribution of nutrients relative to other sources (Cornwell et al. 2014, Novick et al. 2015). Several WWTPs in the Delta have recently been or are planned to be upgraded to include tertiary filtration and nitrification to convert biologically available ammonium to nitrate. The City of Stockton WWTP was upgraded in 2006 and is immediately upstream of station P8 (Jabusch et al. 2016), which provides a valuable opportunity

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225 to assess how nutrient or nutrient-related source controls and water management actions have

226 changed ambient concentrations downstream. A decrease of ammonium concentrations at P8

227 after 2006 is expected as a result of upstream WWTP upgrades, and water quality should exhibit

228 1) a shift in the ratio of the components of DIN from the WWTP before/after upgrade, and 2) a

229 flow-normalized annual trend at P8 to show a change concurrent with WWTP upgrades.

230 *2.4.2 Effects of biological invasions*

231 Invasion of the upper SFE by the Asian clam *Potamocorbula amurensis* in 1986 caused

232 severe changes in phytoplankton abundance and species composition. Reduction in phytoplankton

233 biomass has altered trophic networks in the upper SFE and is considered an important mechanism

234 in the decline of the protected delta smelt (*Hypomesus transpacificus*) and other important

235 fisheries (Feyrer et al. 2003, Mac Nally et al. 2010). Changes in the physical environment have

236 also occurred, particularly increased water clarity from a reduction of particle transport and

237 erodible sediment supply (Jassby 2008, Schoellhamer 2011, Cloern and Jassby 2012), although

238 decreases in phytoplankton by clam biofiltration may have also increased clarity (Mac Nally et al.

239 2010). The clams are halophilic such that drought years are correlated with an increase in biomass

240 and further upstream invasion of the species (Parchaso and Thompson 2002, Cloern and Jassby

241 2012). We hypothesized that results from WRTDS models would show 1) a decline in annual,

242 flow-normalized chlorophyll concentrations over time coincident with an increase in abundance

243 of invaders, and 2) variation in the chlorophyll/clam relationship through indirect or direct

244 controls of flow. Although the relationship between phytoplankton and clams in the upper SFE

245 has been well described (Kimmerer and Thompson 2014), we use WRTDS to develop additional

246 evidence that an increase in DIN was facilitated in part by clam invasion and the relationship of

247 phytoplankton with clam abundance was mediated by flow and climatic variation in recent years.

3 Results

3.1 Observed data and modelled trends

The observed time series for the ten Delta - Suisun Bay stations had substantial variation in scale among the nitrogen analytes and differences in apparent seasonal trends (Fig. 2). DIN for most stations was dominated by nitrite/nitrate, whereas ammonium was a smaller percentage of the total. However, C3 had a majority of DIN composed of ammonium and other stations (e.g., P8, D26) had higher concentrations of ammonium during winter months when assimilation rates are lower (Novick et al. 2015). By location, observed concentrations of DIN for the entire time series were higher on average for the peripheral Delta stations (C3, C10, MD10, P8; mean $\pm$ s.e.: 1.04 ± 0.03 mg L^{-1}) and similar for the interior Delta (D19, D26, D28, 0.43 ± 0.01) and Suisun Bay stations (D4, D6, D7, 0.44 ± 0.01). Average concentrations were highest at P8 (1.63 ± 0.05 mg L^{-1}) and lowest at C3 (0.4 ± 0.01) for DIN, highest at P8 (0.28 ± 0.02) and lowest at D28 (0.05 ± 0.003) for ammonium, and highest at C10 (1.4 ± 0.04) and lowest at C3 (0.15 ± 0.004) for nitrite/nitrate. Mean observed concentrations were also higher later in the time series for all analytes. For example, average DIN across all stations was 0.61 ± 0.01 mg L^{-1} for 1976-1995, compared to 0.7 ± 0.01 for 1996-2013. Seasonal changes across all years showed that nitrogen concentrations were generally lower in the summer and higher in the winter, although observed patterns were inconsistent between sites. For example, site MD10 had distinct seasonal spikes for elevated DIN in the winter, whereas other stations had less prominent seasonal maxima (e.g., C3, D7, Fig. 2).

Relative to the observed data, long-term trends between stations for the different nitrogen

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analytes were apparent from the modelled results (Fig. 3). Although each station varied in the

overall concentrations, patterns within the three regions (peripheral Delta, interior Delta, and

Suisun Bay) were observed. Concentrations for all nitrogen analytes were highest in the

peripheral stations. Ammonium concentrations were highest at P8 and C3 and showed a

consistent increase over time, followed by a reduction beginning in the early 2000s, whereas

ammonium concentrations at C10 and MD10 were low and gradually decreasing throughout the

period of record. By contrast, DIN and nitrite/nitrate concentrations at the peripheral Delta

stations showed increases at P8 and C10 followed by a decline in the early 2000s, whereas

concentrations at C3 and MD10 were lower and did not show any noticeable trends. Trends at the

interior Delta stations showed a gradual increase in ammonium followed by a gradual decrease

beginning in the early 1990s, particularly for D26. Trends in DIN and nitrite/nitrate for the

interior Delta stations showed a reduction early in the time series, followed by a slight increase

beginning in the mid-1980s, and finally a reduction beginning in the late 1990s. These trends

were similar for the Suisun Bay stations, although the reduction in the late 1990s did not occur.

By contrast, ammonium concentrations were low in Suisun Bay but a gradual increase over the

period of record was observed.

3.2   Trend tests

Estimated trends from Seasonal Kendall tests on the raw time series varied considerably

between sites and analytes (Fig. 4). Significant trends were observed from 1976-1995 for eight of

ten sites for DIN (seven increasing, one decreasing), eight sites for ammonium (six increasing,

two decreasing), and six sites for nitrite/nitrate (five increasing, one decreasing). Decreasing

trends were more common for the observed data from 1996-2013. Eight sites had significant

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₂₉₁ trends for DIN (four increasing, four decreasing), seven sites for ammonium (five increasing, two

₂₉₂ decreasing), and eight sites for nitrite/nitrate (four increasing, four decreasing). P8 had a

₂₉₃ relatively large decrease in ammonium (-8.3% change per year) for the second annual period

₂₉₄ compared to all other sites. Trends by season were similar such that increases were generally

₂₉₅ observed in all seasons from 1976-1995 (Fig. S1) and decreases were observed for 1996-2013

₂₉₆ (Fig. S2). Trends for the seasonal comparisons were noisier and significant changes were less

₂₉₇ common compared to the annual comparisons.

₂₉₈   A comparison of flow-normalized results from WRTDS relative to observed data

₂₉₉ identified changes in the magnitude, significance, and direction of trends. For all sixty trend

₃₀₀ comparisons in Fig. 4 (flow-normalized values in Table 2) regardless of site, nitrogen analyte, and

₃₀₁ time period (annual or seasonal aggregations), thirteen comparisons had trends that were

₃₀₂ insignificant ($p > 0.05$) with the observed data but significant with flow-normalized results,

₃₀₃ whereas only one trend changed to insignificant. This suggests that time series that include flow

₃₀₄ effects had sufficient noise to obscure or prevent identification of an actual trend of a water

₃₀₅ quality parameter. Further, changes in the magnitude of the estimated percent change per year

₃₀₆ were also apparent for the flow-normalized trends, such that fourteen comparisons showed an

₃₀₇ increase in magnitude (more negative or more positive) and twenty five had a decrease (less

₃₀₈ positive or less negative) compared to observed trends. Eleven comparisons showed a trend

₃₀₉ reversal from positive to negative estimated change, nine sites went from no change to negative

₃₁₀ estimated change, and one site went from no change to a positive trend for the flow-normalized

₃₁₁ results. Differences by season in the observed relative to flow-normalized trends from WRTDS

₃₁₂ were also apparent (Figs. S1 and S2 and Tables S1 and S2). The most notable change in the

₃₁₃ flow-normalized results was an overall decrease (less positive trend or greater negative trend) in

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concentrations for most sites in the summer and fall seasons for 1996-2013. More statistically

significant trends were also observed with the flow-normalized results.

3.3 Selected examples

3.3.1 *Effects of wastewater treatment*

Effluent measured from 2003 to 2009 from the Stockton WWTP had similar DIN

concentrations before and after upgrades, whereas ammonium concentrations were greatly

reduced (Fig. 5). Ammonium and nitrate concentrations were comparable prior to 2006, whereas

nitrate was a majority of total nitrogen after the upgrade, with much smaller percentages from

ammonium and nitrite. As expected, flow-normalized nitrogen trends at P8 shifted in response to

upstream WWTP upgrades (Fig. 6a), with ammonium showing an increase from 1976 followed

by a large reduction in the 2000s. Nitrite/nitrate concentrations at P8 also showed a similar but

less dramatic decrease despite an increase in the WWTP effluent concentrations following the

upgrade (Fig. 5). Percent changes from seasonal Kendall tests on flow-normalized results

(Table 3) showed that both nitrogen species increased prior to WWTP upgrades (2% per year for

nitrite/nitrate, 2.8% for ammonium), followed by decreases after upgrades (-1.9% for

nitrite/nitrate, -16.6% for ammonium). Seasonally, increases prior to upgrades were highest in

the summer for nitrite/nitrate (2.4%) and in the fall for ammonium (4.9%). Similarly, seasonal

reductions post-upgrade were largest in the summer for nitrite/nitrate (-4.3%) and largest for

ammonium in the winter (-26.7%).

Nitrogen concentrations varied with flow, although relationships depended on season and

year. Relationships of nitrite/nitrate with flow described by WRTDS showed flushing or dilution

at higher flow (Fig. 6b). Seasonal variation was even more apparent for ammonium, although

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both nitrite/nitrate and ammonium typically had the highest concentrations at low flow in the winter (January). Additionally, strength of the flow/nutrient relationship changed between years. Nitrite/nitrate typically had the strongest relationship with flow later in the time series (i.e., larger negative slope), whereas ammonium had the strongest relationship with flow around 2000 in January.

3.3.2   Effects of biological invasions

Invasion in the 1980s showed a clear reduction of *Corbicula fluminea* and increase of *P. amurensis* (Fig. 7a), where biomass of the latter was negatively associated with flow from the Sacramento river (Fig. 7b). The increase in clam abundance was associated with a notable decrease in annually-averaged chl-*a* from WRTDS results (Fig. 7c), as expected if WRTDS is adequately capturing flow variation and identifying the well-established phytoplankton decrease beginning in the 1980s. A seasonal shift in the flow-normalized results was also observed such that chl-*a* concentrations were generally highest in July/August prior to invasion, whereas a spring maximum in April was more common in recent years (Fig. 7f). An increase in annually-averaged silicon dioxide (Fig. 7d) was coincident with the chl-*a* decrease, with the largest increases occuring in August (Fig. 7g). Further, DIN trends were similar to silicon-dioxide in both annual and seasonal changes (i.e., Figures 7e and 7h compared to 7d and 7g), such that an increase in both nutrients earlier in the time series corresponded with the decrease in chl-*a*.

The relationship of chl-*a* with clam biomass was significant (Fig. 7i), with lower chl-*a* associated with higher biomass. However, the effect of flow on both clams and phytoplankton as a top-down or bottom-up control changed throughout the time series. The chl-*a*/flow relationship showed that increasing flow (decreasing salinity) was associated with a slight increase in chl-*a* followed by a decrease early in the time series (Fig. 7j), whereas overall chl-*a* was lower but a

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359 positive association with flow (negative with salinity) was observed later in the time series.

360 Following clam invasion, chl-*a* concentrations were reduced by grazing but showed a positive and

361 monotonic relationship with increasing flow. The increase in clam abundance was concurrent with

362 decline in chl-*a* concentration, although variation in abundance between years was also observed.

363 Clam abundance was reduced during high flow years in the late 1990s, 2006, and 2011 (7a). In

364 the same years, WRTDS predictions for chl-*a* were higher than the flow-normalized component

365 (Fig. 7c), which further suggests a link between increased flow and phytoplankton production.

4   Discussion

367 Water quality conditions in the Delta-Suisun Bay region are dynamic and not easily

368 characterized from observed time series. Annual aggregations of WRTDS modelled results and

369 application of formal trend analyses provided insight into the spatial and temporal variation of

370 nitrogen analytes in three distinct regions that is not possible with raw observations. A general

371 conclusion is that nitrogen concentrations have showed a consistent decrease beginning in the

372 mid-1990s and early 2000s, although average concentrations remain above those observed earlier

373 in the period of record. These results are confirmed visually from WRTDS (Fig. 3) and through

374 significant results from trend tests (Fig. 4, Table 2). Although the overall trends suggest a

375 system-wide reduction, considerable differences by location and analyte were characterized by

376 the analysis and are important independent of overall trends. Nutrient concentrations were highest

377 at peripheral Delta stations (C3, MD10, P8, and C10) that monitor inflows from the Sacramento

378 and San Joaquin rivers. The highest concentrations among all nitrogen analytes, ammonium in

379 particular, were observed at P8 in the early 2000s as a direct consequence of WWTP inputs

380 upstream prior to infrastructure upgrades. Elevated ammonium concentrations were also observed

at C3 as a measure of upstream contributions from the Sacramento WWTP. By contrast, nitrite/nitrate concentrations were highest at C10 as a measure of contributions from the San Joaquin River to the south that drains a predominantly agricultural watershed.

Differing magnitudes of nitrogen analytes between stations as a function of source type can have an effect on the relationship between flow and nutrients. Both Hirsch et al. (2010) and Beck and Hagy III (2015) used WRTDS results to demonstrate variation between flow and nutrient dynamics depending on pollutant sources. In particular, a chemodynamic (i.e., changing) response of nutrients with flow variation is common if nutrients originate primarily from the watershed through diffuse sources (Thompson et al. 2011, Wan et al. 2017). Increased flow may induce a change in nutrient concentrations, such that reduction may occur with flushing or an increase may occur through mobilization. By contrast, nutrient loads are relatively chemostatic or invariant with changes in flow if point-sources are the dominant contributor. These relationships are modelled particularly well with WRTDS, which can provide a means of hypothesizing unknown sources or verifying trends in response to management actions. As noted above, C10 at the inflow of the San Joaquin River is dominated by nitrite/nitrate consistent with diffuse, agricultural inputs from the watershed. A logical expectation is that trends from observed data may vary considerably from trends with modelled results that are flow-normalized. Accordingly, trend analysis of nitrate/nitrate by year and season showed that percent changes at C10 were typically underestimated with the observed data during the recent period from 1996-2013 (Tables 2 and S2). This is consistent with an expected effect of flow on raw time series, particularly for chemodynamic behavior at locations that drain highly altered watersheds (Wan et al. 2017).

Our results underscore the importance of considering flow effects in the evaluation of

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water quality trends. There were important differences in trends between observed and flow-normalized data. These differences ranged from a simple change in the magnitude and significance of a trend, to more problematic changes where the flow-normalized trend could demonstrate a complete reversal relative to the observed (e.g., DIN trends for all Suisun stations from 1996-2013, Fig. 4). Differences in apparent trends underscore the importance of considering flow effects in the interpretation of environmental changes, particularly if trend evaluation is used to assess the effects of nutrients on ecosystem health or the effectiveness of past nutrient management actions. An alternative evaluation of flow in the Delta demonstrated that flow contributions from different sources vary considerably over time at each station (Novick et al. 2015). For example, flow at MD10 represents seasonally changing inputs from the Sacramento and San Joaquin rivers and other sources. For simplicity, our analysis considered only the dominant water source and used the total daily inflow estimates from either the Sacramento River or the San Joaquin River at any of the Delta sites. Given that substantial differences with flow-normalized results were apparent from relatively coarse estimates of flow contributions, more precise differences could be obtained by considering the influence of multiple flow components at each location. Output from the Dayflow software program (IEP 2016) provides a complete mass balance of flow in the Delta that could be used to develop a more comprehensive description. Our analysis is the first attempt to model nutrient dynamics related to flow in the entire Delta, such that additional work should focus on improving the characterization of flow signals at each station.

Long-term trends in nutrient and phytoplankton concentrations in Suisun Bay have also been the focus of intense study for many years (Cloern et al. 1983, Lehman 1992, Dugdale et al. 2007, Jassby 2008, Glibert et al. 2014). Our results demonstrated an overall increase in

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ammonium from the late 1970s to 2000, with initial flow-normalized values of annual averages

estimated as approximately 0.05 mg L^{-1} for the Suisun stations in 1976 to a maximum ranging

from 0.08 mg L^{-1} (station D4) to above 0.1 mg L^{-1} (D6) in 2000 (Fig. 3). Trends from

1996-2005 evaluated in (Jassby 2008) showed a similar increase in ammonium. However, a

reversal of trends in recent years may also be occuring in Suisun Bay, as model estimates suggest

either relatively constant concentrations or even a decrease at some stations beginning around

2000 (e.g., D6, D7, Fig. 4). Combined with the shift towards a dominant spring peak in chloropyll

growth (Fig. 7f), changing nitrogen ratios continue to be a concern for the management of

production in the upper SFE.

4.1 Interpretation of case studies

Seasonal timings of water quality improvements and the link to flow changes are difficult

to characterize from the observed time series. WRTDS models were used to characterize these

changes at P8 and to develop additional hypotheses of factors that influence nutrient

concentrations. A general conclusion is that ammonium reductions were concurrent with WWTP

upgrades, but the reduction was most apparent at low-flow in January. Estimated ammonium

concentrations in July were low for all flow levels, which suggests either nitrogen inputs were low

in the summer or nitrogen was available but uptake by primary consumers and bacterial

processing were high. Seasonal patterns in the relationship between flow and nitrite/nitrate were

not as dramatic as compared to ammonium, and in particular, low-flow events in July were

associated with higher concentrations. This could suggest that ammonium concentrations at P8

are driving phytoplankton production at low flow during warmer months, and not nitrite/nitrate

given the higher estimated concentrations in July at low flow. As such, these simple observations

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provide quantitative support of cause and effect mechanisms of nutrient impacts on potentially

adverse environmental conditions as they relate to nutrient-related source controls upstream.

Additional research could investigate these hypotheses to better describe mechanisms of change

as a basis for more informed management.

The results for Suisun Bay provide additional descriptions of change in production as it

relates to flow, grazing, and nitrogen ratios. In general, clam biomass was associated with a

decrease in chl-*a* concentration, as shown by others (Alpine and Cloern 1992, Thompson et al.

2008). Our results also suggested that diatoms were the dominant genera early in the time series,

particularly in late summer, whereas the spring peak observed in later years represents a shift to

an earlier seasonal maxima. This supports past research that showed a decrease in silica uptake by

diatoms following invasion (Cloern 1996, Kimmerer 2005). A nontrivial portion of the DIN

increase could be related to the decrease in a major 'sink', i.e., decreased DIN uptake by

phytoplankton due to top down grazing pressure from *P. amurensis*. Flow effects on

phytoplankton production have also changed over time. In the absence of benthic grazing prior to

invasion, chl-*a* production was limited at low flow as less nutrients were exported from the Delta,

stimulated as flow increases, and reduced at high flow as either nutrients or phytoplankton

biomass are exported to the estuary (Fig. 7j). Recent years have shown a decrease in overall

chlorophyll, with particularly low concentrations at low flow (high salinity). As such, chl-*a*

production in early years is directly related to flow, whereas the relationship with flow in later

years is indirect as increased flow reduces clam abundance and releases phytoplankton from

benthic grazing pressure. These relationships have been suggested by others (Cloern et al. 1983,

Alpine and Cloern 1992, Parchaso and Thompson 2002, Jassby 2008), although the precise

mechanisms demonstrated by WRTDS provide additonal quantitative evidence of factors that

472 drive water quality in the Delta.

4.2 Conclusions

474   As demonstrated by both case studies and the overall trends across all stations, water

475 quality dynamics in the Delta are complex and driven by multiple factors that change through

476 space and time. WRTDS models can facilitate descriptions of change by focusing on high-level

477 forcing factors that explicitly account for annual, seasonal, and flow effects on trend

478 interpretations. We have demonstrated the potential for imprecise or inaccurate conclusions of

479 trend tests that focus solely on observed data and emphasize that flow-normalized trends have

480 more power to quantify change. The results from WRTDS are also consistent with described

481 trends of nutrient loads from point sources (e.g., Sacramento WWTP increases and exports to

482 Suisun Bay, Jassby 2008, Novick and Senn 2014), demonstrating that these changes are not

483 unexpected. Consequently, we are not detracting from the potential implications of such

484 increases. The important conclusion is that the physical/hydrological and biogeochemical factors

485 that influence nutrient cycling and ambient concentrations in the Bay-Delta, and changes to those

486 factors, are substantial enough that they can be comparable in magnitude to anthropogenic load

487 increases or comparable to the effects of management actions to decrease nutrient levels.

488 Therefore, methods that adjust for the effects of these factors are critical when studying long-term

489 records to assess the impacts or effectiveness of load increases or management actions,

490 respectively.

491   Combined with additional data, our results can support hypotheses that lead to a more

492 comprehensive understanding of ecosystem dynamics. Additional factors to consider include the

493 effects of large-scale climatic patterns, more detailed hydrologic descriptions, and additional

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ecological components that affect trophic interactions. For example, a more rigorous matching of flow time series with water quality observations at each station that considers varying source contributions over time could provide a more robust description of flow effects. Alternative methods could also be used to address a wider range of questions, particularly those with more generic structural forms that can explicitly include additional variables (e.g., generalized additive models, Beck and Murphy 2017). Overall, quantitative interpretations of multiple factors can provide a more comprehensive understanding of relationships between nutrients and primary production, including adverse effects on ecosystem condition.

Acknowledgments

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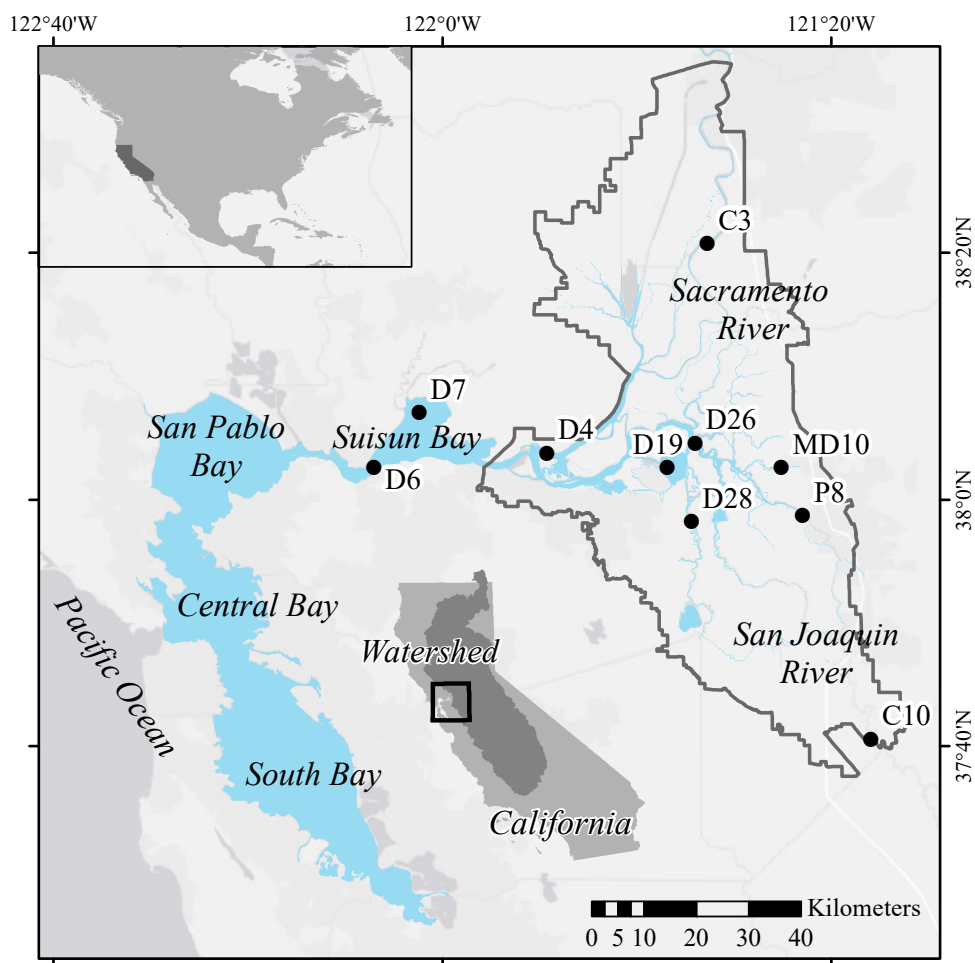Fig. 1: The San Francisco Estuary and Delta region with monitoring stations used for analysis. The Delta drains the combined watersheds of the Sacramento and San Joaquin rivers (inset). The grey outline is the legal boundary of the Delta. All data were obtained from the Interagency Ecological Program website (http://water.ca.gov/bdma/meta/Discrete/data.cfm, IEP (2013)). See Table 1 for station descriptions.
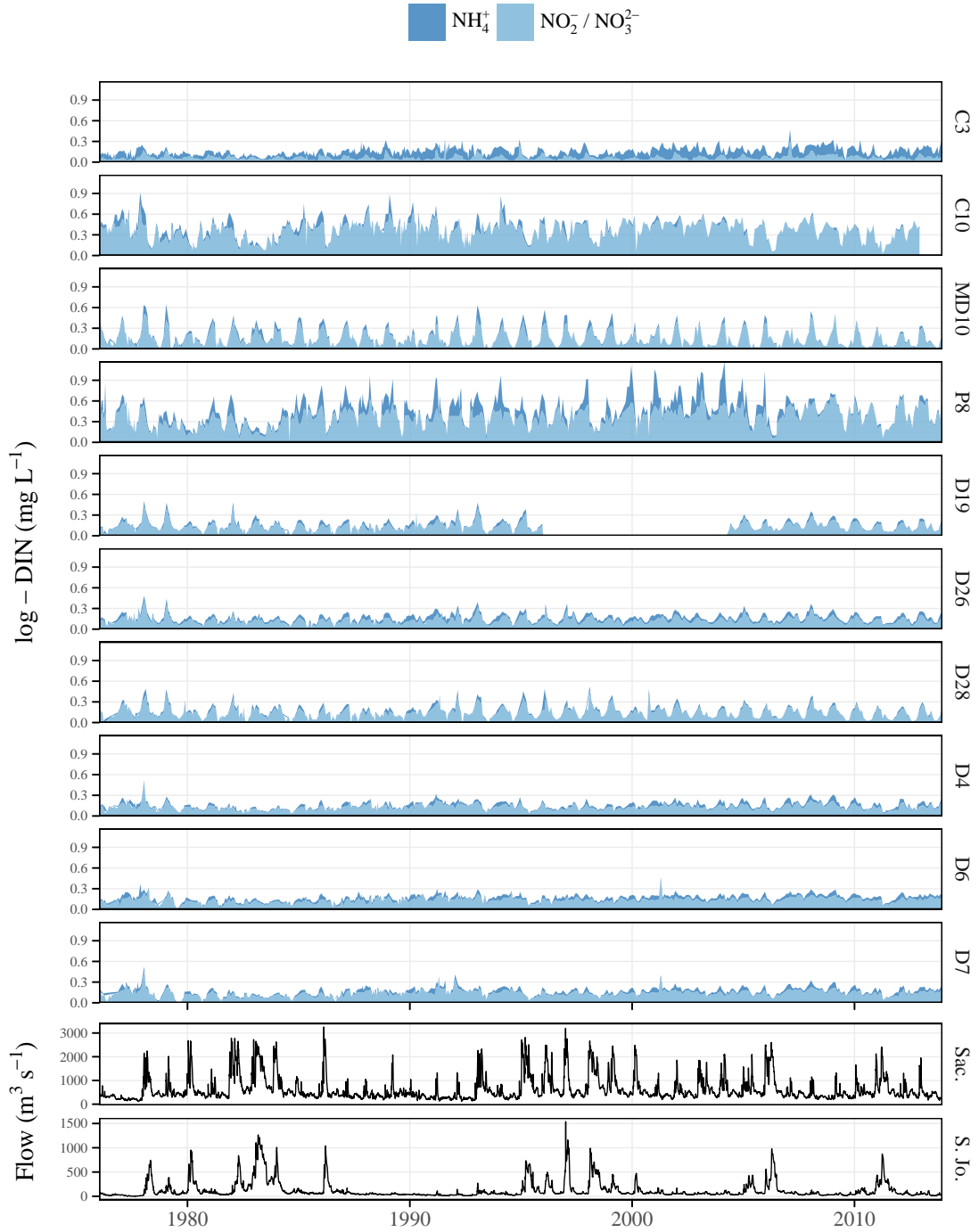
Fig. 2: Observed DIN ($NH_4^+ + NO_2^-/NO_3^{2-}$) from ten stations in the upper SFE Delta and flow from the Sacramento and San Joaquin rivers. Data were collected monthly and evaluated with WRTDS models using daily flow estimates from 1976 to 2013. Note different y-axis scales. See Fig. 1 for station locations.

Fig. 3: Model results (points) and flow-normalized predictions (lines) for ten stations grouped by nitrogen analyte and geographic location in the Delta region (locations in Fig. 1). Results are annually-averaged for each water year from October to September.

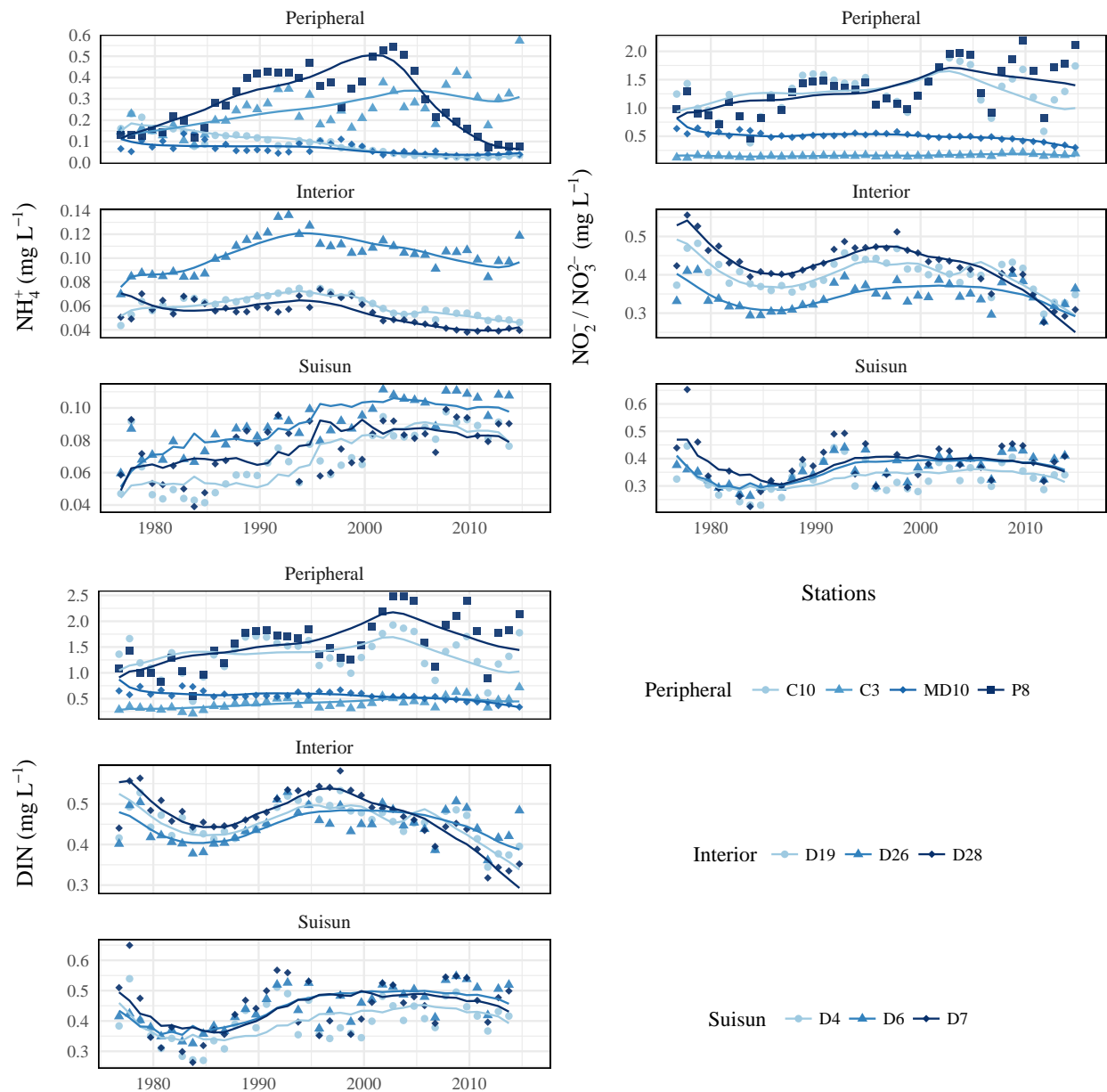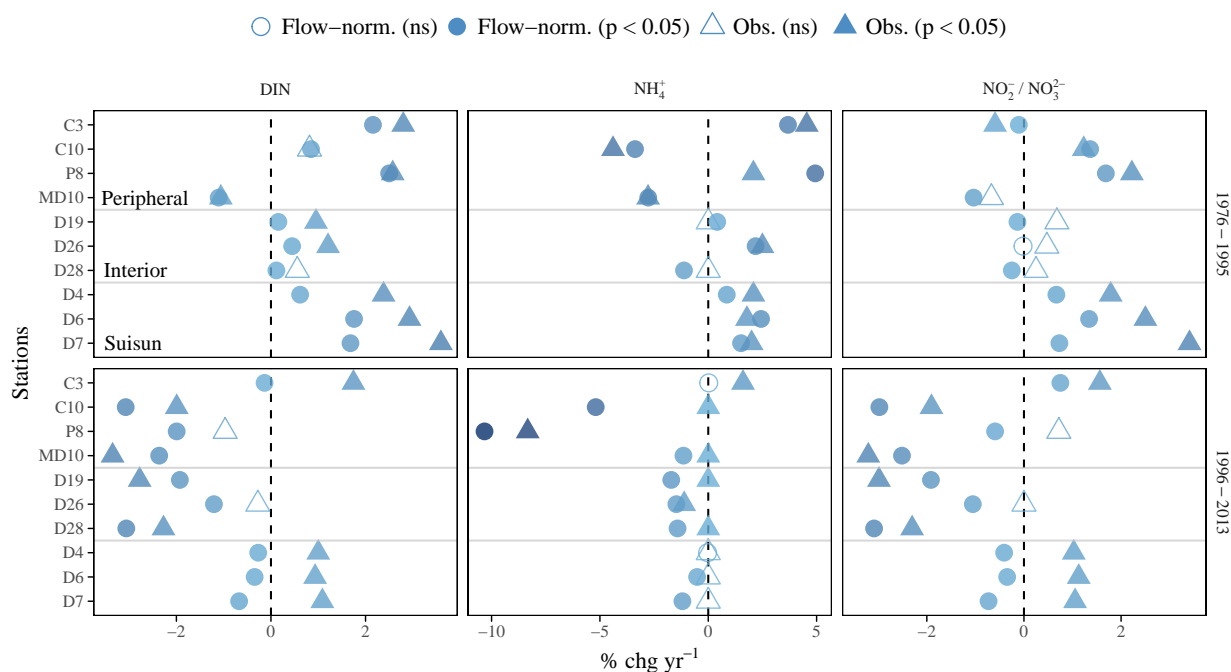Fig. 4: Results from seasonal Kendall tests on observed data (triangles) and flow-normalized predictions (circles) from WRTDS for nitrogen analytes. Results are shown as the percent change per year as the estimated Theil-Sen slope divided by the median for a given aggregation period (significance evaluated at $\alpha = 0.05$, based on τ). Trends are shown separately for different annual groupings from 1976 to 1995 (top) and 1996 to 2013 (bottom). See Figs. S1 and S2 for seasonal groupings.
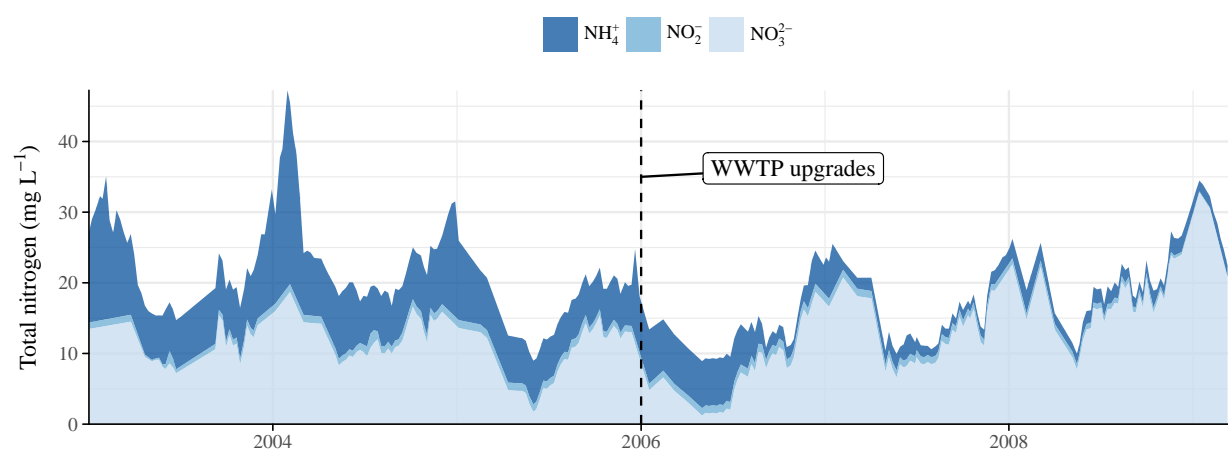
Fig. 5: Nitrogen concentration measurements (mg L^{-1}) from the City of Stockton Wastewater Treatment Plant, San Joaquin County. Wastewater discharge requirements were implemented in 2006 to convert ammonium to nitrate.
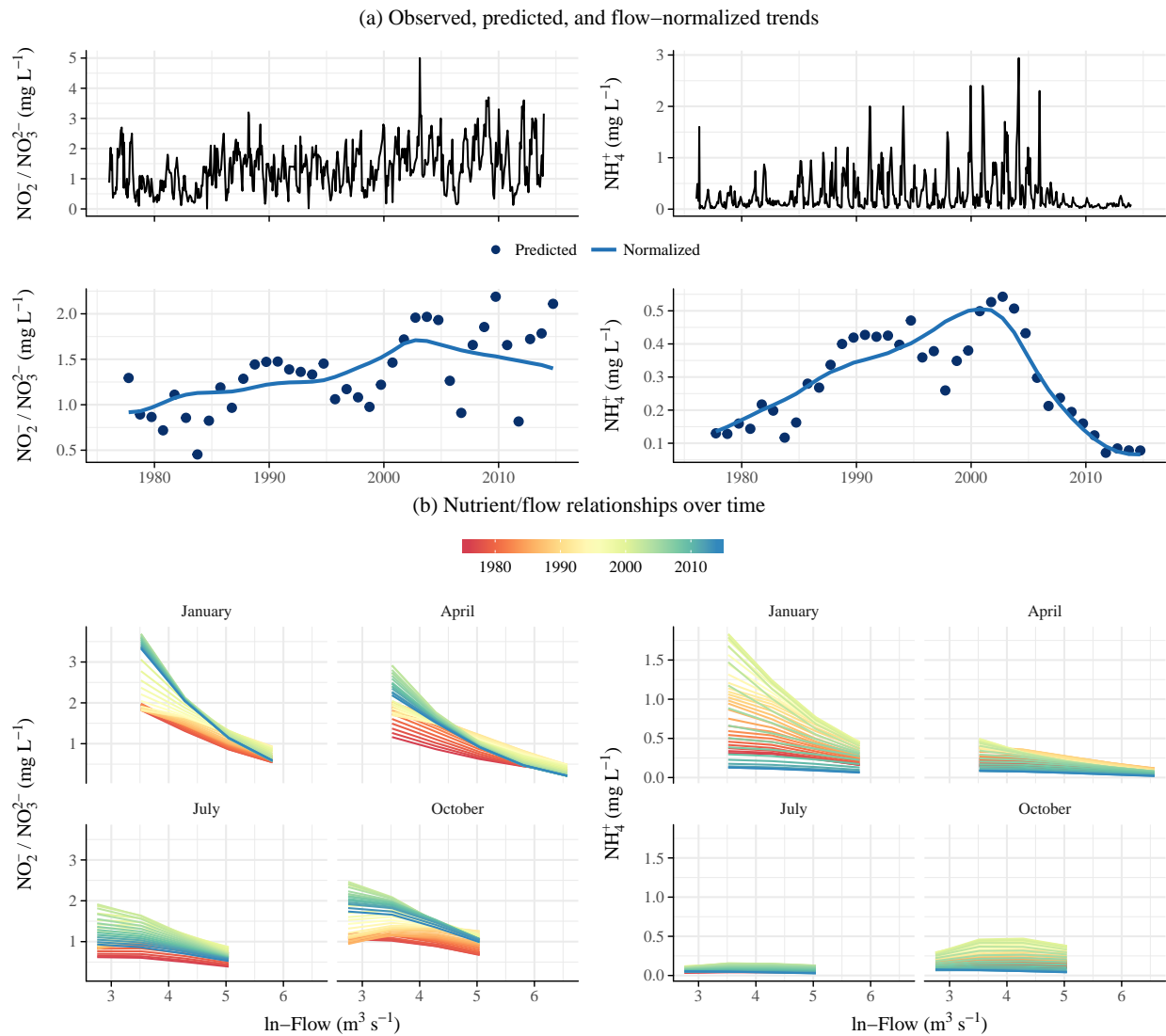
Fig. 6: Nitrogen trends at P8 as (a, top) observed, (a, bottom) predicted and flow-normalized estimates from WRTDS, and (b) relationships with flow over time from WRTDS. Nitrite/nitrate trends are on the left and ammonium trends are on the right. Wastewater treatment plant upgrades at the City of Stockton (San Joaquin County) were completed in 2006 (Fig. 5).
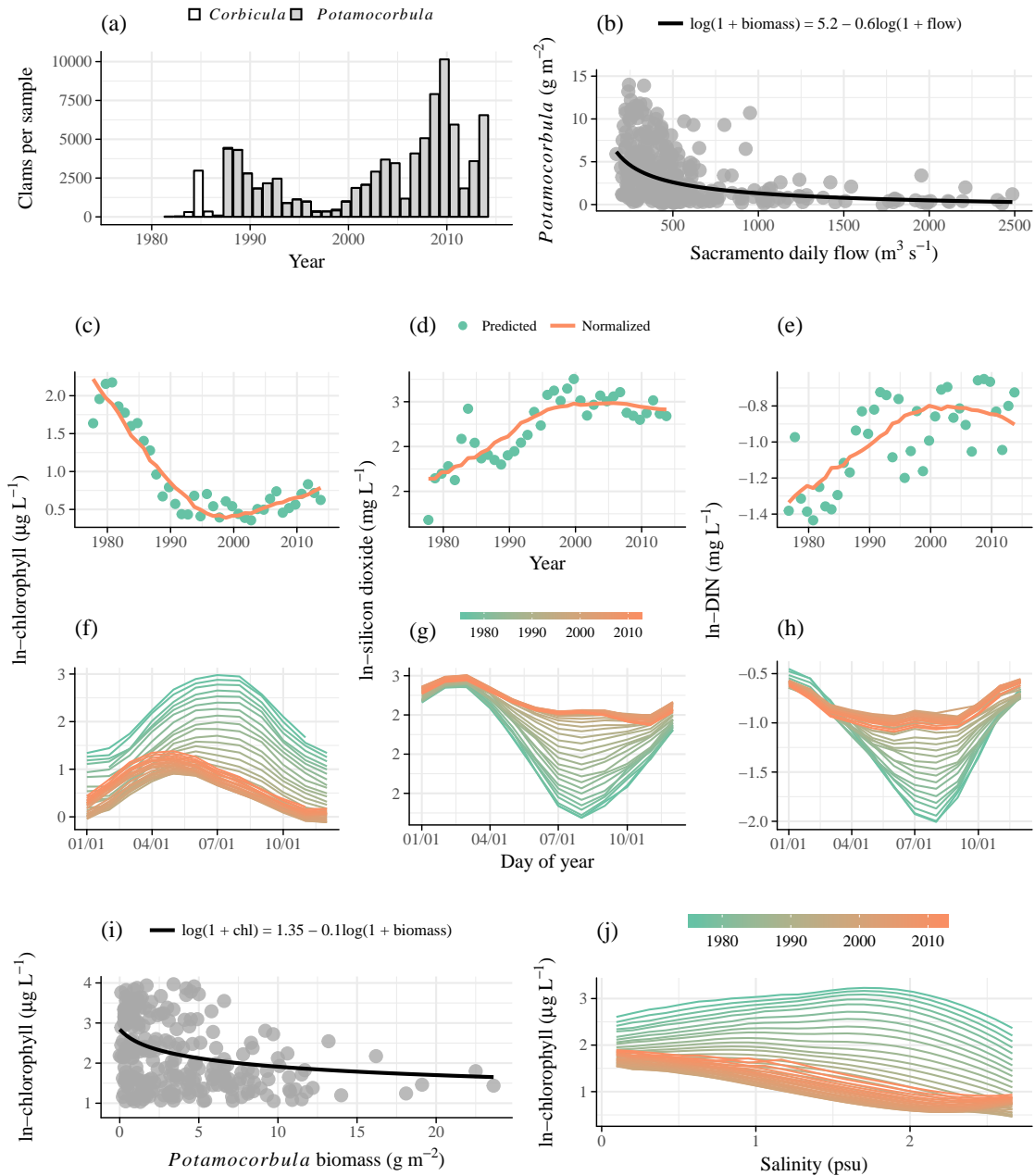
Fig. 7: Trends in clam abundance and chl-*a* concentration from 1976 to 2013 at station D7 in Suisun Bay. Invasion by *Potamocorbula amurensis* clams in the late 1980s and reduction of *Corbicula fluminea* was shown by changes in clam density (a, annual means), with biomass linked to salinity (b). A decrease in chl-*a* concentration was also observed by changes in annual (c) and seasonal trends (f) based on WRTDS results. Reductions in chl-*a* concentration were coincident with an increase in SiO_2 and DIN concentrations (d, e), with the greatest increases in August (g, h). A significant ($p < 0.001$) relationship between clam biomass and chl-*a* concentration is shown in subfigure (i). Flow relationships with chl-*a* concentration shown by WRTDS have also changed over time (j, observations from June).

Table 1: Monitoring stations in the upper San Francico Estuary used to evaluate nitrogen trends. Records from 1976 to 2013 were evaluated using the total observations (n) at each station. Median values (mg L^{-1}) are reported for the entire period of record.

Station			DIN		NH$_4^+$		NO$_2^-$/NO$_3^{2-}$	
	Lat	Lon	n	Med.	n	Med.	n	Med.
Peripheral								
C3	38.35	−121.55	569	0.36	569	0.22	571	0.13
C10	37.68	−121.27	539	1.44	539	0.04	558	1.40
MD10	38.04	−121.42	548	0.31	548	0.03	570	0.28
P8	37.98	−121.38	556	1.46	556	0.12	563	1.20
Interior								
D19	38.04	−121.61	433	0.35	435	0.04	462	0.31
D26	38.08	−121.57	556	0.38	556	0.09	565	0.29
D28	37.97	−121.57	529	0.38	535	0.03	555	0.33
Suisun								
D4	38.06	−121.82	546	0.38	546	0.05	565	0.32
D6	38.04	−122.12	534	0.45	534	0.08	562	0.35
D7	38.12	−122.04	535	0.44	535	0.06	561	0.36

Table 2: Summaries of flow-normalized trends in nitrogen analytes for all stations and annual aggregations. Summaries are medians (mg L^{-1}) and percent change per year in parentheses (increasing in bold). Changes and significance estimates are based on seasonal Kendall tests of flow-normalized results within each time period. *$p < 0.05$

Analyte/Station	Annual	
	1976-1995	1996-2013
DIN		
C10	1.3 (**0.8**)*	1.4 (-3.1)*
C3	0.3 (**2.2**)*	0.5 (-0.1)*
D19	0.4 (**0.2**)*	0.4 (-1.9)*
D26	0.4 (**0.4**)*	0.5 (-1.2)*
D28	0.4 (**0.1**)*	0.4 (-3.1)*
D4	0.3 (**0.6**)*	0.4 (-0.3)*
D6	0.4 (**1.8**)*	0.5 (-0.3)*
D7	0.4 (**1.7**)*	0.5 (-0.7)*
MD10	0.4 (-1.1)*	0.3 (-2.4)*
P8	1.3 (**2.5**)*	1.7 (-2)*
NH$_4^+$		
C10	0.1 (-3.4)*	0 (-5.2)*
C3	0.2 (**3.7**)*	0.3 (**0**)
D19	0 (**0.4**)*	0 (-1.7)*
D26	0.1 (**2.2**)*	0.1 (-1.5)*
D28	0 (-1.1)*	0 (-1.4)*
D4	0 (**0.9**)*	0.1 (**0**)
D6	0.1 (**2.4**)*	0.1 (-0.5)*
D7	0.1 (**1.5**)*	0.1 (-1.2)*
MD10	0.1 (-2.8)*	0 (-1.1)*
P8	0.2 (**4.9**)*	0.1 (-10.3)*
NO$_2^-$/NO$_3^{2-}$		
C10	1.2 (**1.4**)*	1.4 (-3)*
C3	0.1 (-0.1)*	0.2 (**0.7**)*
D19	0.4 (-0.1)*	0.4 (-1.9)*
D26	0.3 (**0**)	0.4 (-1.1)*
D28	0.4 (-0.2)*	0.4 (-3.1)*
D4	0.3 (**0.7**)*	0.3 (-0.4)*
D6	0.3 (**1.3**)*	0.4 (-0.3)*
D7	0.4 (**0.7**)*	0.4 (-0.7)*
MD10	0.4 (-1)*	0.3 (-2.5)*
P8	1.2 (**1.7**)*	1.5 (-0.6)*

Table 3: Summaries of flow-normalized trends in nitrite/nitrate and ammonium (mg L^{-1}) concentrations before and after WWTP upgrades upstream of station P8. Upgrades were completed in 2006 at the City of Stockton WWTP (San Joaquin County, Fig. 5). Summaries are medians and percent change per year in parentheses (increasing in bold). Changes and significance estimates are based on seasonal Kendall tests of flow-normalized results within each time period. Increasing values are in bold-italics. Months for each season are Spring: MAM, Summer: JJA, Fall: SON, Winter: DJF. *$p < 0.05$

Period	NO_2^-/NO_3^{2-}		NH_4^+	
	Median	% change	Median	% change
Annual				
1976-2006	1.3	**2***	0.2	**2.8***
2007-2013	1.4	-1.9*	0.1	-16.6*
Seasonal, pre				
Spring	1.2	**1.6***	0.2	**1.4***
Summer	1	**2.4***	0.1	**3.3***
Fall	1.3	**2.2***	0.2	**4.9***
Winter	1.5	**2.1***	0.7	**4.8***
Seasonal, post				
Spring	1.3	-1.6*	0.1	-16.2*
Summer	0.9	-4.3*	0.1	-15.7*
Fall	1.5	-1.7*	0.1	-19.3*
Winter	2.2	-0.8*	0.2	-26.7*

Fig. S1: Results from seasonal Kendall tests on observed data (triangles) and flow-normalized predictions (circles) from WRTDS for nitrogen analytes. Results are shown as the percent change per year as the estimated Theil-Sen slope divided by the median for a given aggregation period (significance evaluated at $\alpha = 0.05$, based on τ). Trends are shown separately for different seasonal groupings from 1976-1995. Months for each season are Spring: MAM, Summer: JJA, Fall: SON, Winter: DJF. See Fig. 4 for annual comparisons.
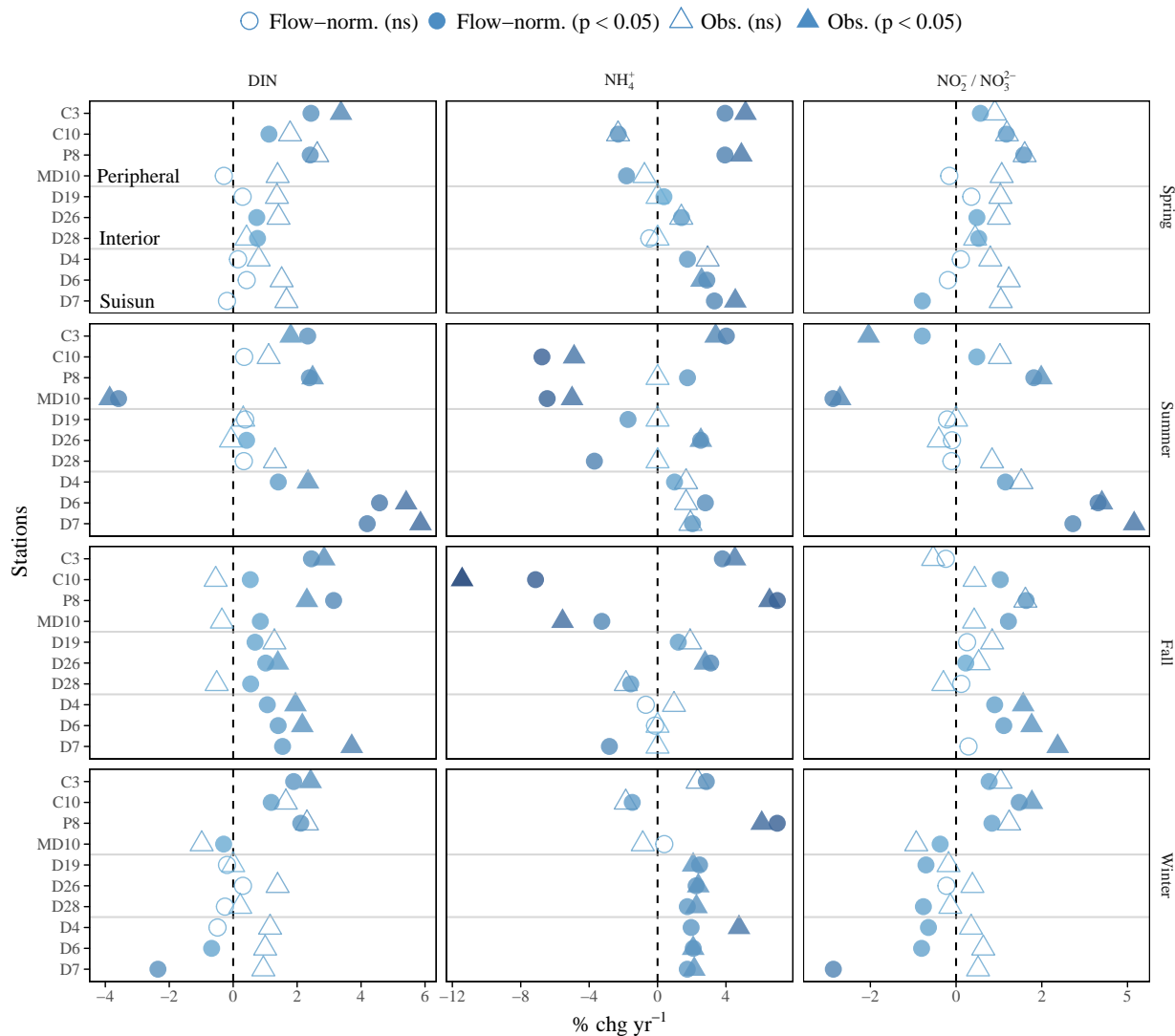
Fig. S2: Results from seasonal Kendall tests on observed data (triangles) and flow-normalized predictions (circles) from WRTDS for nitrogen analytes. Results are shown as the percent change per year as the estimated Theil-Sen slope divided by the median for a given aggregation period (significance evaluated at $\alpha = 0.05$, based on τ). Trends are shown separately for different seasonal groupings from 1996-2013. Months for each season are Spring: MAM, Summer: JJA, Fall: SON, Winter: DJF. See Fig. 4 for annual comparisons.
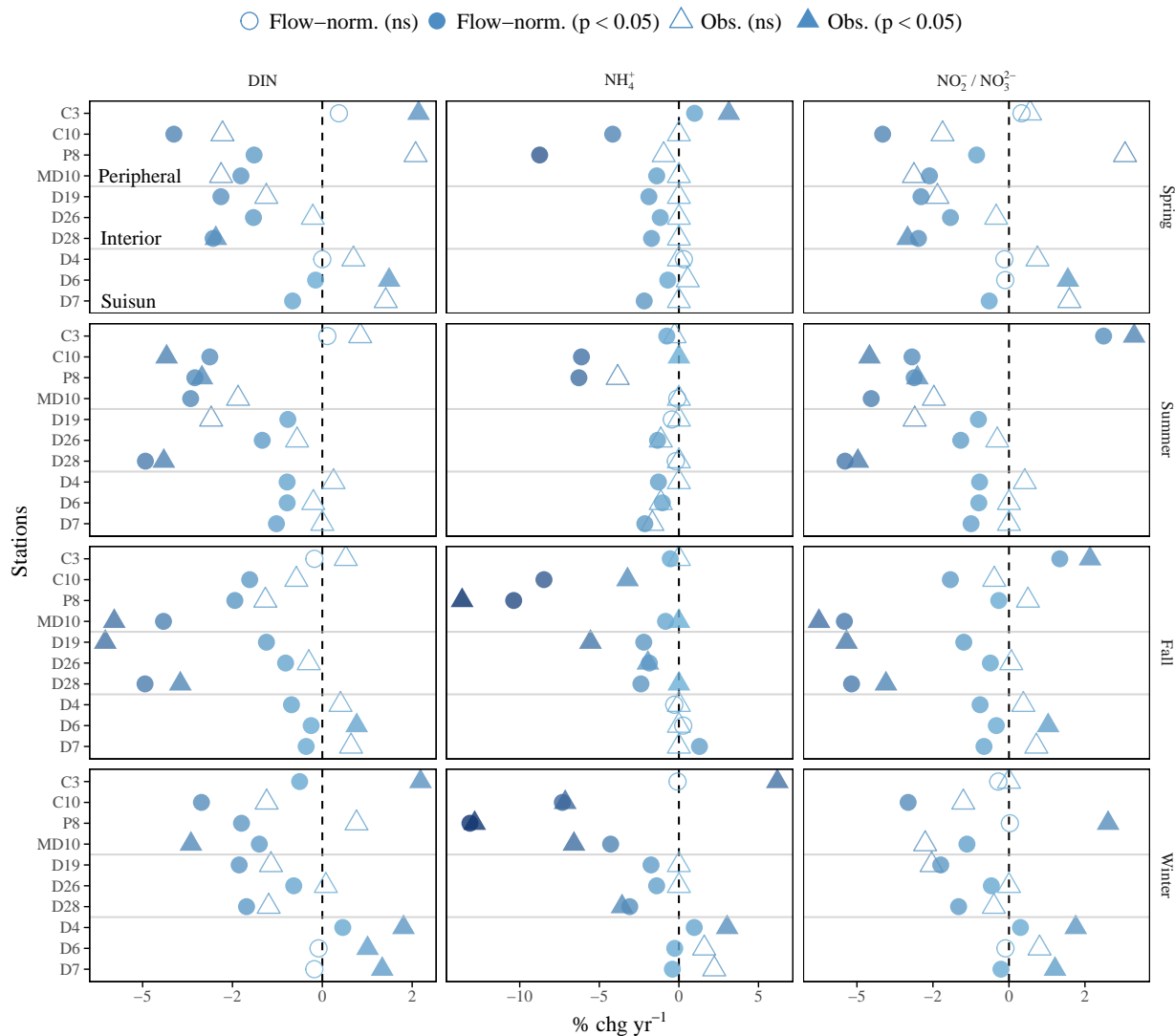
Table S1: Summaries of flow-normalized trends in nitrogen analytes for all stations and seasonal aggregations from 1976-1995. Summaries are medians (mg L^{-1}) and percent change per year in parentheses (increasing in bold). Changes and significance estimates are based on seasonal Kendall tests of flow-normalized results within each time period. Months for each season are Spring: MAM, Summer: JJA, Fall: SON, Winter: DJF. *$p < 0.05$

Analyte/Station	Seasonal, 1976-1995			
	Spring	Summer	Fall	Winter
DIN				
C10	1.2 (**1.1**)*	1.2 (**0.3**)	1.3 (**0.5**)*	1.7 (**1.2**)*
C3	0.3 (**2.4**)*	0.3 (**2.3**)*	0.4 (**2.4**)*	0.4 (**1.9**)*
D19	0.5 (**0.3**)	0.2 (**0.4**)	0.3 (**0.7**)*	0.7 (-0.2)
D26	0.4 (**0.7**)*	0.3 (**0.4**)*	0.4 (**1**)*	0.6 (**0.3**)
D28	0.5 (**0.8**)*	0.2 (**0.3**)	0.3 (**0.5**)*	0.8 (-0.3)
D4	0.4 (**0.2**)	0.3 (**1.4**)*	0.3 (**1.1**)*	0.5 (-0.5)
D6	0.4 (**0.4**)	0.3 (**4.6**)*	0.4 (**1.4**)*	0.5 (-0.7)*
D7	0.4 (-0.2)	0.3 (**4.2**)*	0.4 (**1.5**)*	0.6 (-2.4)*
MD10	0.6 (-0.3)	0.2 (-3.6)*	0.3 (**0.8**)*	1.3 (-0.3)*
P8	1.3 (**2.4**)*	0.9 (**2.4**)*	1.3 (**3.1**)*	1.9 (**2.1**)*
NH$_4^+$				
C10	0.1 (-2.3)*	0 (-6.8)*	0.1 (-7.1)*	0.3 (-1.5)*
C3	0.2 (**3.9**)*	0.2 (**4**)*	0.3 (**3.8**)*	0.2 (**2.9**)*
D19	0.1 (**0.4**)*	0 (-1.7)*	0 (**1.2**)*	0.1 (**2.5**)*
D26	0.1 (**1.4**)*	0.1 (**2.5**)*	0.1 (**3.1**)*	0.1 (**2.3**)*
D28	0.1 (-0.5)	0 (-3.7)*	0 (-1.6)*	0.1 (**1.7**)*
D4	0.1 (**1.7**)*	0 (**1**)*	0 (-0.7)	0.1 (**2**)*
D6	0.1 (**2.9**)*	0.1 (**2.8**)*	0.1 (-0.1)	0.1 (**2.1**)*
D7	0.1 (**3.3**)*	0 (**2**)*	0.1 (-2.8)*	0.1 (**1.7**)*
MD10	0.1 (-1.8)*	0 (-6.5)*	0 (-3.3)*	0.2 (**0.4**)
P8	0.2 (**3.9**)*	0.1 (**1.8**)*	0.2 (**7**)*	0.6 (**7**)*
NO$_2^-$/NO$_3^{2-}$				
C10	1.1 (**1.5**)*	1.2 (**0.6**)*	1.2 (**1.3**)*	1.5 (**1.8**)*
C3	0.2 (**0.7**)*	0.1 (-1)*	0.1 (-0.3)	0.2 (**1**)*
D19	0.4 (**0.4**)	0.2 (-0.3)	0.3 (**0.3**)	0.6 (-0.9)*
D26	0.4 (**0.6**)*	0.2 (-0.1)	0.3 (**0.3**)*	0.5 (-0.3)
D28	0.5 (**0.7**)*	0.2 (-0.1)	0.3 (**0.2**)	0.7 (-1)*
D4	0.3 (**0.1**)	0.3 (**1.4**)*	0.3 (**1.1**)*	0.4 (-0.8)*
D6	0.4 (-0.2)	0.3 (**4.1**)*	0.3 (**1.4**)*	0.4 (-1)*
D7	0.4 (-1)*	0.3 (**3.4**)*	0.4 (**0.4**)	0.4 (-3.6)*
MD10	0.5 (-0.2)	0.2 (-3.6)*	0.2 (**1.5**)*	1.2 (-0.5)*
P8	1.2 (**2**)*	0.9 (**2.3**)*	1.1 (**2**)*	1.4 (**1**)*

Table S2: Summaries of flow-normalized trends in nitrogen analytes for all stations and seasonal aggregations from 1996-2013. Summaries are medians (mg L^{-1}) and percent change per year in parentheses (increasing in bold). Changes and significance estimates are based on seasonal Kendall tests of flow-normalized results within each time period. Months for each season are Spring: MAM, Summer: JJA, Fall: SON, Winter: DJF. *$p < 0.05$

Analyte/Station	Seasonal, 1996-2013			
	Spring	Summer	Fall	Winter
DIN				
C10	1.1 (-4.1)*	1.3 (-3.1)*	1.6 (-2)*	1.7 (-3.4)*
C3	0.5 (**0.5**)	0.4 (**0.1**)	0.6 (-0.2)	0.5 (-0.6)*
D19	0.5 (-2.8)*	0.2 (-1)*	0.3 (-1.6)*	0.7 (-2.3)*
D26	0.5 (-1.9)*	0.3 (-1.7)*	0.4 (-1)*	0.6 (-0.8)*
D28	0.5 (-3)*	0.2 (-4.9)*	0.2 (-4.9)*	0.7 (-2.1)*
D4	0.4 (**0**)	0.4 (-1)*	0.4 (-0.9)*	0.5 (**0.6**)*
D6	0.5 (-0.2)*	0.5 (-1)*	0.5 (-0.3)*	0.5 (-0.1)
D7	0.5 (-0.8)*	0.4 (-1.3)*	0.4 (-0.4)*	0.6 (-0.2)
MD10	0.4 (-2.3)*	0.2 (-3.7)*	0.2 (-4.4)*	1 (-1.8)*
P8	1.5 (-1.9)*	1.2 (-3.5)*	1.8 (-2.4)*	2.7 (-2.2)*
NH$_4^+$				
C10	0 (-4.2)*	0 (-6.1)*	0 (-8.5)*	0.1 (-7.3)*
C3	0.3 (**1**)*	0.3 (-0.8)*	0.4 (-0.5)*	0.2 (-0.1)
D19	0 (-1.9)*	0 (-0.4)	0 (-2.2)*	0.1 (-1.8)*
D26	0.1 (-1.2)*	0.1 (-1.3)*	0.1 (-1.9)*	0.1 (-1.4)*
D28	0 (-1.7)*	0 (-0.2)	0 (-2.4)*	0.1 (-3.1)*
D4	0.1 (**0.3**)	0 (-1.3)*	0.1 (-0.3)	0.1 (**1**)*
D6	0.1 (-0.7)*	0.1 (-1)*	0.1 (**0.3**)	0.1 (-0.3)
D7	0.1 (-2.2)*	0 (-2.1)*	0.1 (**1.3**)*	0.1 (-0.4)*
MD10	0 (-1.4)*	0 (-0.1)	0 (-0.8)*	0.1 (-4.3)*
P8	0.2 (-8.7)*	0.1 (-6.3)*	0.2 (-10.4)*	0.5 (-13.1)*
NO$_2^-$/NO$_3^{2-}$				
C10	1.1 (-4.2)*	1.2 (-3.2)*	1.6 (-1.9)*	1.6 (-3.3)*
C3	0.2 (**0.4**)	0.1 (**3.1**)*	0.2 (**1.7**)*	0.2 (-0.4)
D19	0.4 (-2.9)*	0.2 (-1)*	0.3 (-1.5)*	0.6 (-2.2)*
D26	0.4 (-1.9)*	0.2 (-1.6)*	0.3 (-0.6)*	0.5 (-0.6)*
D28	0.5 (-3)*	0.2 (-5.4)*	0.2 (-5.2)*	0.7 (-1.7)*
D4	0.3 (-0.1)	0.3 (-1)*	0.3 (-1)*	0.4 (**0.4**)*
D6	0.4 (-0.1)	0.4 (-1)*	0.4 (-0.4)*	0.4 (-0.1)
D7	0.4 (-0.6)*	0.4 (-1.2)*	0.4 (-0.8)*	0.4 (-0.3)*
MD10	0.4 (-2.6)*	0.1 (-4.5)*	0.2 (-5.4)*	1 (-1.4)*
P8	1.3 (-1.1)*	1.1 (-3.1)*	1.6 (-0.3)*	2.2 (**0**)