

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

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Abstract

Recent methods for trend analysis have been developed that leverage the descriptive potential of long-term time series. Combined with these methods, multi-decadal datasets of water quality in coastal systems can provide valuable opportunities to gain insights into ecosystem properties and drivers of change. This study describes use of an estuarine adaptation of the Weighted Regressions on Time, Discharge, and Season (WRTDS) model to describe water quality trends over four decades in the Delta region of the San Francisco Estuary (SFE). This region is a complex mosaic of inflows that are primary sources of nutrients into the larger Bay. To date, a comprehensive evaluation of flow-normalized trends using the long-term monitoring dataset at multiple stations in the Delta has not been conducted despite the importance of nutrient transport from the region for water quality in the entire bay. The WRTDS technique is data-driven

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13 where the parameterization of the functional model changes smoothly over time fol-
14 lowing dynamic patterns of season and flow. Water quality trends that have not been
15 previously quantified can be described, including variation in flow-normalized concen-
16 trations, frequency occurrence of extreme events, and response to historical changes
17 in the watershed, all of which are important needs for understanding changes in the
18 SFE. Model results from multiple stations in the Delta provided novel descriptions
19 of historical trends and relationships between key species of dissolved inorganic nitro-
20 gen (ammonium, nitrate/nitrite, total). This variation was described in the context of
21 varying contributions of input flows from the Sacramento and San Joaquin rivers, as
22 well as tidal exchange with the central SFE. Conceptual relationships between water
23 quality and drivers of change were used to generate and test hypotheses of mechanistic
24 relationships using selected examples from the trend descriptions. Overall, this analysis
25 provides an ecological and management-based understanding of historical trends in the
26 SFE as a means to interpret potential impacts of recent changes and expected trends in
27 this dynamic system. An argument is also made for more comprehensive evaluations of
28 long-term monitoring datasets to understand relationships between response endpoints
29 and causal mechanisms in coastal waters.

30 1 Introduction

31 Trend analysis is a broad discipline that has been applied to time series for the interpretation
32 of environmentally-relevant changes. Direct evaluation of an observed time series is often
33 insufficient given that a long-term change can be masked by variation at shorter time scales
34 or the observed variation represents the combined effects of many variables.^{1,2} Climate, local,
35 regional, and historical effects may act individually or together to impose a change on time
36 series, such that methods that account for variation at different scales have been used for
37 trend analysis.^{3–6} As a practical approach for water quality evaluation, trend analysis of
38 eutrophication endpoints often focuses on tracking the change in concentrations or loads of

39 nutrients over many years. Indicators of eutrophication can vary naturally with variation
40 in flow conditions and may also reflect long-term effects of management or policy changes.

41 For example, chlorophyll *a* (chl-*a*) concentration as a measure of phytoplankton response {acro:chla}
42 to nutrient inputs can follow seasonal patterns with cyclical variation in temperature and
43 light changes throughout each year, whereas annual trends can follow long-term variation
44 in nutrient inputs to the system.^{7,8} Similarly, nutrient trends that vary with hydrologic
45 loading also vary as a function of utilization rates by primary producers or decomposition
46 processes.^{9–11} Time series analysis of water quality indicators must simultaneously consider
47 effects of processes at multiple scales and interactions between variables of interest to develop
48 a more comprehensive description of system change.

49 Appropriate methods for the analysis of change depend largely on the question of inter-
50 est and on characteristics of the environmental dataset. Trend analyses for aquatic systems
51 have traditionally focused on comparisons between discrete periods of time to estimate a
52 direction and magnitude of a trend using non-parametric tests.^{12,13} Development of these
53 conventional approaches addressed limitations in historical monitoring datasets related to
54 infrequent sampling and relatively few years of continuous data. Increased availability of
55 multi-decadal datasets, particularly for high profile environments, has accelerated recent de-
56 velopment of trend analysis methods that leverage the descriptive potential of long-term
57 time series from continuous monitoring programs.^{6,14} These methods are often data-driven
58 where the parameterization of a simple functional model can change smoothly over time
59 given that relationships between water quality variables and potential drivers are dynamic.

60 The Weighted Regressions on Time, Discharge, and Season (WRTDS) approach was devel- {acro:wrtds}
61 oped under this context and has been used to characterize decadal trends in running-water
62 systems.^{15–19} This method has the potential to provide a spatially and temporally robust
63 description of trends by fitting a dynamic model with parameters that change relative to the
64 domain of interest. More recently, the WRTDS method was adapted for trend analysis in
65 tidal waters, with a focus on chl-*a* trends in Tampa Bay²⁰ and the Patuxent River Estuary,²¹

66 and tidally-influenced time series of dissolved oxygen from continuous sonde measurements.²²
67 These studies have demonstrated potential for the use WRTDS for trend analysis in tidal
68 waters and further application to alternative datasets could provide additional insight into
69 drivers of change in aquatic systems.

70 The San Francisco Estuary (SFE) on the Pacific Coast of the United States is one of the {acro:sfe}
71 most prominent and culturally significant estuaries in the western hemisphere.²³ Background
72 nutrient concentrations in the Bay often exceed those associated with excessive primary
73 production, although eutrophication events have historically been infrequent. Recent changes
74 in response to additional stressors (e.g., variation in freshwater inputs/withdrawals, invasive
75 species, climate change) suggests that Bay condition has not followed past trajectories and
76 more subtle spatial and temporal variation could provide clues that describe underlying
77 properties of this system.²⁴ The unique ecological and social context of the Bay provides a
78 valuable opportunity to gain insight into ecosystem properties of estuaries that define water
79 quality dynamics at different scales. The Delta region of SFE in particular is a mosaic of
80 inflows that receives and processes inputs from the larger watershed to the lower Bay.²⁵⁻²⁷ A
81 comprehensive monitoring dataset has been collected at several fixed locations in the Delta
82 for the last four decades.²⁸ Moreover, nutrient dynamics in the Delta are inherently linked
83 to flow variation from inputs, withdrawal, impoundments, and downstream transport,²⁹
84 suggesting an approach that explicitly considers flow effects is critical for trend analysis. To
85 date, the Delta monitoring dataset is an under-utilized data source and a comprehensive
86 analysis with WRTDS could facilitate an understanding of historical and recent changes in
87 SFE water quality.

88 The goal of this study was to provide a comprehensive description of nutrient trends in
89 the Delta to inform understanding of eutrophication dynamics and potential causes of water
90 quality change in the larger Bay. We applied the newly-adapted method of weighted regres-
91 sion for tidal waters to describe nitrogen trends in different spatial and temporal contexts.
92 The specific objectives were to 1) quantify and interpret trends over four decades at ten

93 stations in the Delta, including annual, seasonal, and spatial changes in nitrogen analytes
94 and response to flow variation, 2) provide detailed descriptions of two case studies in the
95 context of conceptual relationships modelled with WRTDS. The second objective evaluated
96 two specific water quality stations in the Delta to demonstrate complexities with nutrient
97 response to flow, effects of wastewater treatment plant (WWTP) upgrades on water quality,
98 and effects of biological invasion by benthic filter feeders on primary production. Although
99 quantitative descriptions of change can be ends in themselves, the results were expected
100 to have greater impact as a means to more detailed understanding of ecosystem proper-
101 ties. Products derived from WRTDS can be used to inform additional analyses, such as
102 water quality response after removing annual, seasonal, or flow effects. Overall, this analysis
103 is expected to further an ecological and management-based understanding of dynamics in
104 San Francisco Bay, with implications for water quality restoration and protection of this
105 prominent system.

106 2 Materials and Methods

107 2.1 Study system

108 The SFE drains a 200 thousand km² watershed and is the largest bay on the Pacific coast of
109 North America. The watershed provides drinking water to over 25 million people, including
110 irrigation for 18 thousand km² of agricultural land in the Central Valley. Water enters
111 the Bay through the Sacramento and San Joaquin rivers that have a combined inflow of
112 approximately 28 km³ per year, with the Sacramento accounting for 84% of inflow to the
113 Delta. The SFE system is divided into several sub-bays, including Suisun Bay immediately
114 downstream of the Delta, San Pablo Bay to the north, South Bay, and the Central Bay
115 that drains to the Pacific Ocean through the Golden Gate. Water dynamics in SFE are
116 governed by inflows from the watershed, tidal exchange with the Pacific Ocean, and water
117 withdrawals for municipal and agricultural use.²⁵ Seasonally, inflows into SFE peak in the

118 spring and early summer from snowmelt in the upper watershed, whereas consumption,
119 withdrawals, and export have steadily increased from 1960 to present but vary considerably
120 depending on inter-annual climate effects.²⁴ The system is mixed mesotidal and significant
121 exchange with the ocean occurs daily, although the extent of landward saltwater intrusion
122 varies with inflow and annual water use patterns. Notable drought periods have occurred
123 from 1976-1977, 1987-1992, and recently from 2013-2015.²³ Oceanic upwelling and climatic
124 variation are also significant external factors that have influenced water quality dynamics in
125 the Bay.³⁰

126 Nutrient loading in SFE is comparable to other large estuaries that exhibit symptomatic
127 effects of cultural eutrophication (e.g., Chesapeake Bay).³¹ Orthophosphate (PO_4^{3-}) and {acro:din}
128 dissolved inorganic nitrogen (DIN) enter the Bay primarily through riverine sources in the
129 north and municipal WWTP inputs in the densely-populated area immediately surrounding
130 SFE. Annual nutrient export from the Delta region has been estimated as approximately 30
131 thousand kg d⁻¹ of total nitrogen (varying with flow²⁹), with 90% of ammonium (NH_4^+) orig-
132 inating solely from the Sacramento Regional WWTP.²⁷ Although nitrogen and phosphorus
133 inputs are considerable, primary production is relatively low and not nutrient-limited.^{26,32}
134 The resistance of SFE to the negative effects of eutrophication has historically been at-
135 tributed to the unique physical and biological characteristics of the Bay, including strong
136 tidal mixing that limits stratification^{7,33} and limits on phytoplankton growth from high tur-
137 bidity and filter-feeding by bivalve mollusks.^{33,34} However, recent water quality trends have
138 suggested that resistance of the system to nutrient inputs is decreasing given documented
139 changes in chlorophyll biomass,³⁰ increased occurrence of hypoxic conditions,³⁵ and increased
140 abundance of phytoplankton species associated with harmful algal blooms.^{36,37} These recent
141 changes have been attributed to variation in global sea surface temperatures associated with
142 climate change,³⁰ biological invasions,³⁸ and departures from the historical flow record.^{24,39}
143 The role of nutrients in stimulating primary production in SFE has been the focus of several
144 recent investigations.⁴⁰⁻⁴²

The Delta region is of particular interest for understanding historical patterns and potential trajectories of water quality response to nutrient inputs into the Bay (Figure 1). The Delta is a mosaic of linked channels or tracts that receive, process, and transport inflows from the Sacramento and San Joaquin rivers.^{25,27,29} Quantitative descriptions of nutrient dynamics in the Delta are challenging given many nutrients sources and the volume of water that is exchanged through the system with natural and anthropogenic processes. A comprehensive evaluation using mass-balance models to describe nutrient dynamics in the Delta demonstrated that nitrogen enters the system in different forms and is processed at different rates before export or removal.²⁹ For example, a 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 lost. Although, the focus of our analysis is not to quantify sources or sinks of nitrogen species, a quantitative evaluation of long-term trends will provide a more comprehensive historical interpretation to hypothesize the effects of future changes in the context of known dynamics. Nutrients in the Delta also vary with seasonal and annual changes in the delivery of water inflows, including water exports directly from the system.^{25,27} Our analysis also explicitly accounts for the effects of flow changes on nutrient response to better understand variation both within the Delta and potential mechanisms of downstream transport.

2.2 Data sources

Multi-decadal time series of nutrients and flow records were used to develop a quantitative description of nitrogen trends in the Delta. The Interagency Ecological Program (IEP) is a consortium of state and federal agencies that have maintained the Environmental Monitoring Program (EMP) in the Delta region since 1975.⁴³ The EMP collects monthly water quality samples at 19 stations in the Delta, Suisun Bay, and northeastern San Pablo Bay. Water samples were collected using a Van Dorn sample, a submersible pump, or a flow through system depending on site. All samples were processed with standard QA/QC at the California

171 Department of Water Resources Bryte Laboratory in Sacramento.⁴³ Nutrient time series were
172 obtained from the IEP website (<http://water.ca.gov/bdma/meta/Discrete/data.cfm>) at
173 ten discrete sampling stations from 1976 through 2013 (Figure 1). Stations were grouped
174 by location in the study area for comparison: Delta stations C3 (Sacramento inflow), C10
175 (San Joaquin inflow), MD10, P8; middle stations D19, D26, D28; and Suisun stations D4,
176 D6, and D7. These stations were chosen based on continuity of the water quality time series
177 and geographic location for understanding trends. Time series were complete for all stations
178 except for an approximate ten year gap from 1996-2014 for D19. Data were minimally pro-
179 cessed with the exception of averaging replicates that occurred on the same day. The three
180 nitrogen analytes that were evaluated were ammonium, nitrite/nitrate, and DIN (as the sum
181 of the former two). Less than 3% of all observations were left-censored, although variation
182 was observed between analytes and location. The most censored observations were observed
183 for ammonium time series at sites C10 (25.4%), MD10 (18.1%), D28 (17.8%), D19 (12%),
184 and D7 (7.9%).

185 Daily flow estimates for the Delta region were obtained from the Dayflow software pro-
186 gram that provides estimates of average Delta outflow.⁴⁴ Because of the complexity of water
187 inflow, exports, and outflows from the Delta, the Dayflow program combines observations
188 with estimates based on mass balance to reconstruct historical and daily flow estimates.
189 The WRTDS models described below require a matched flow record with the appropriate
190 station to evaluate nutrient trends. Given the complexity of inflows and connectivity of the
191 system, only the inflow estimates from the Sacramento and San Joaquin rivers were used as
192 measures of freshwater influence at each station. Initial analyses indicated that model fit
193 was not significantly improved with flow estimates from locations closer to each station, nor
194 was model fit improved using lagged times series. As such, the Sacramento daily flow time
195 series was used to account for flow effects at C3, D19, D26, and D28, and the San Joaquin
196 time series was used for C10 and P8. The salinity observations at D4, D6, and D7 in Suisun
197 Bay were used as a more appropriate measure of variation in freshwater balance given the

¹⁹⁸ stronger tidal influence at these stations. Salinity has been used as a tracer of freshwater
¹⁹⁹ influence for the application of WRTDS models in tidal waters.²⁰

²⁰⁰ 2.3 Analysis method and application

²⁰¹ A total of thirty WRTDS models were created, one for each nitrogen analyte at each station.

²⁰² The functional form of WRTDS is a simple regression¹⁵ that models the log-transformed
²⁰³ response variable as a function of time, flow, and season:

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

²⁰⁴ where N is one of three nitrogen analytes, time t is a continuous variable as decimal time
²⁰⁵ to capture the annual or seasonal trend, and Q is the flow variable (either flow or salinity
²⁰⁶ depending on station). The seasonal trend is modelled as a sinusoidal component to capture
²⁰⁷ periodicity between years. The WRTDS model is a moving window regression that fits
²⁰⁸ a unique set of parameters at each observation point in the time series. A unique set of
²⁰⁹ weights is used for each regression to control the relevance of observations used to fit the
²¹⁰ model to the observation at the center of the window. The weights are based on a scaled
²¹¹ Euclidean distance to estimate the differences of all points from the center in relation to
²¹² annual time, season, and flow. The final vector used to fit the model at each point weights
²¹³ observations more similar to the center of the window with more importance. The complete
²¹⁴ model for the time series contains a parameter set for every time step that considers the
²¹⁵ unique context of the data. As such, predictions from WRTDS are more precise than those
²¹⁶ from more conventional models that fit a single parameter set to the entire time series.^{20,45}

²¹⁷ The WRTDS model applied to the Delta time series was based on a tidal adaptation of the
²¹⁸ original method.²⁰ The WRTDS models were fit to describe the conditional mean response
²¹⁹ using a weighted Tobit model for left-censored data.⁴⁶ Previous adaptations of WRTDS to
²²⁰ tidal waters have used quantile regression to describe trends in the conditional quantiles,

such as changes in the frequency of occurrence of extreme events. The application to the Delta data focused only on the conditional mean models to establish a baseline response which has not been previously quantified. All analyses used the WRTDStidal package for R.^{47,48}

A hallmark of the WRTDS approach is the description of flow-normalized trends that are independent of variation from freshwater inflows. Flow-normalized trends have value for the interpretation of changes that are potentially caused by drivers other than flow, such as WWTP upgrades or phytoplankton grazing by benthic filter-feeders.²⁰ Although variation in nutrients is caused by the combined effects of several variables acting at different temporal and spatial scales, flow-normalization provides a basis for further exploration by removing a critical confounding variable that could affect the interpretation of trends. A flow-normalized value is the average of predictions at a given observation using all flow values that are expected to occur for the relevant month across years in the record. Flow-normalized trends for each analyte at each station were used to describe long-term changes in different annual and seasonal periods. Specifically, 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. These annual and seasonal groupings were chosen for continuity with similar comparisons reported in Ref. 28 and as approximate twenty year midway points in the time series.

Trends within 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.⁴⁹ 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

²⁴⁸ median value of the response variable in the period of observation.²⁷ Trends within annual
²⁴⁹ groupings were based on all monthly observations within relevant years, whereas seasonal
²⁵⁰ groupings were based only on the relevant months across years. Seasonal Kendall tests were
²⁵¹ also used to describe trends in the model predictions for the observed data. These trends were
²⁵² compared with those based on the flow-normalized trends to evaluate the improved ability of
²⁵³ WRTDS to describe trends that are independent of flow. Functions in the EnvStats package
²⁵⁴ in R were used for the seasonal Kendall tests.⁵⁰

²⁵⁵ 3 Results and Discussion

²⁵⁶ 3.1 Observed Data

²⁵⁷ The observed time series for the ten Delta stations had substantial variation in scale among
²⁵⁸ the nitrogen analytes and differences in apparent seasonal trends (Figure 2). In general, long-
²⁵⁹ term (inter-annual) trends were not easily observed from the raw data. DIN for most stations
²⁶⁰ was dominated by nitrite/nitrate, whereas ammonium was a smaller percentage of the total.
²⁶¹ However, C3 had a majority of DIN composed of ammonium and other stations (e.g., P8,
²⁶² D16) had higher concentrations of ammonium during winter months when phytoplankton
²⁶³ assimilation is lower.²⁹ By location, observed concentrations of DIN for the entire time
²⁶⁴ series were higher on average for the upper Delta stations (C3, C10, MD10, P8; maximum
²⁶⁵ likelihood estimation of mean \pm standard error: $1.04 \pm 0.03 \text{ mg L}^{-1}$) and similar for the
²⁶⁶ middle (D19, D26, D28, 0.43 ± 0.01) and Suisuan Bay stations (D4, D6, D7, 0.44 ± 0.01).
²⁶⁷ Average concentrations were highest at P8 ($1.63 \pm 0.05 \text{ mg L}^{-1}$) and lowest at C3 (0.4 ± 0.01)
²⁶⁸ for DIN, highest at P8 (0.28 ± 0.02) and lowest at D28 (0.05 ± 0.003) for ammonium, and
²⁶⁹ highest at C10 (1.4 ± 0.04) and lowest at C3 (0.15 ± 0.004) for nitrite/nitrate. Mean observed
²⁷⁰ concentrations were also higher later in the time series for all analytes. For example, average
²⁷¹ DIN across all stations was $0.61 \pm 0.01 \text{ mg L}^{-1}$ for 1976-1995, compared to 0.7 ± 0.01 for 1996-
²⁷² 2013. Seasonal changes across all years also suggested that nitrogen concentrations were

²⁷³ lower in the summer and higher in the winter. However, observed seasonality patterns were
²⁷⁴ inconsistent between sites. For example, site MD10 had distinct seasonal spikes for elevated
²⁷⁵ DIN in the winter, whereas other stations had less prominent variation between years (D6,
²⁷⁶ D7, Figure 2).

²⁷⁷ 3.2 Trends

²⁷⁸ Application of seasonal Kendall tests to evaluate trends in observed data provided explicit in-
²⁷⁹ formation on the direction, magnitude, and statistical significance of changes between years.
²⁸⁰ Trends estimated from the observed data for 1976-1995 and 1996-2013 varied considerably
²⁸¹ between sites and analytes (Figure 3). Significant trends were observed from 1976-1995 for
²⁸² eight of ten sites for DIN (seven increasing, one decreasing), eight sites for ammonium (six
²⁸³ increasing, two decreasing), and six sites for nitrite/nitrate (five increasing, one decreasing).
²⁸⁴ More sites had decreasing trends for the observed data from 1996-2013. Eight sites had
²⁸⁵ significant trends for DIN (four increasing, four decreasing), seven sites for ammonium (five
²⁸⁶ increasing, two decreasing), and eight sites for nitrite/nitrate (four increasing, four decreas-
²⁸⁷ ing). Trends by location (upper Delta, middle, and Suisun stations) were not apparent,
²⁸⁸ suggesting individual sites had trends that differed independent of relative location. For
²⁸⁹ example, P8 had a relatively large decrease in ammonium (-8.3% change per year) for the
²⁹⁰ second annual period compared to all other sites. Trends by season were similar such that
²⁹¹ increases were generally observed in all seasons from 1976-1995 (Figure S1) and decreases
²⁹² were observed for 1996-2013 (Figure S2). Trends for the seasonal comparisons were noisier
²⁹³ and significant changes were less common compared to the annual comparisons.

²⁹⁴ Relationships between flow and observed water quality are complex and can change signif-
²⁹⁵ icantly through space and time.^{15,19} These principles have been demonstrated for monitoring
²⁹⁶ data in the Delta region,²⁷⁻²⁹ suggesting that trend analyses using the observed time series
²⁹⁷ are confounded by flow effects. Change over for time in the observed reflect could reflect
²⁹⁸ mobilization or dilution effects of flow on concentration rather than an empirical system

299 response to changes in nutrient sources. As a proof of concept, Figure 4 demonstrates use of
300 WRTDS to isolate a flow-normalized time series from the observed data. Raw data are pre-
301 sented in Figure 4a and the annual results by water year (October through September) from
302 WRTDS are shown in Figure 4b. In addition to removing the seasonal component, Figure 4b
303 shows the flow-normalized component (solid line) independent of the model predictions. The
304 difference between the two is shown in Figure 4c such that years with predictions greater
305 or less than the flow-normalized values correspond with long-term trends in flow shown in
306 Figure 4d. For example, 1984 is a period of high flow and a large, negative difference between
307 prediction and flow-normalized concentration, suggesting a dilution effect of increased flow
308 on nutrient concentration. Further, Figure 4e shows seasonal variation in the relationships
309 of DIN with flow throughout the period of record. Increases in flow (y-axis) were associated
310 with an increase in DIN (colors) for flow values within the observed range. Seasonal patterns
311 also differed throughout the time period with a wider range of DIN within a growing season
312 in the early 2000s relative to the 1980s.

313 A comparison of trends with flow-normalized results from WRTDS relative to observed
314 data is justified because flow and nutrient concentrations were linked at many of the stations
315 in the study area, similar to Figure 4. These comparisons are made relative to changes in the
316 magnitude, significance, and direction of trends, all of which have important implications
317 for decision-making. For all sixty trend comparisons in Figure 3 regardless of site, nitrogen
318 analyte, and time period, thirteen comparisons had trends that were insignificant with the
319 observed data but significant with flow-normalized results. The magnitude of all comparisons
320 changed

321 Overall, the differences in apparent trends underscore the critical importance of consid-
322 ering flow effects in the interpretation of environmental changes, where flow in the Delta
323 is affected by complex relationships between river inputs, water withdrawals, and tidal ex-
324 change with the larger Bay.

³²⁵ **3.3 Selected examples**

³²⁶ Two stations were chosen for closer evaluation to demonstrate use of WRTDS to develop a
³²⁷ more comprehensive description of decadal trends in the Delta. The stations were chosen to
³²⁸ address ecological and management-based questions that have relevance outside of the region,
³²⁹ having importance for the understanding of estuarine processes that influence eutrophication
³³⁰ trends over several years. The selected case studies focused on 1) effects of wastewater
³³¹ treatment upgrades upstream of P8, and 2) effects of biological invasion on nutrient dynamics
³³² in Suisun Bay. Each case study is built around hypotheses that results from WRTDS models
³³³ were expected to support, both as a general description and for additional testing with
³³⁴ alternative methods.

³³⁵ **3.3.1 Effects of wastewater treatment**

³³⁶ Wastewater treatment plants upstream of and within the Delta are a major source of nutrient
³³⁷ loading to the system. As noted in,²⁷ the Sacramento Regional WWTP alone contributes
³³⁸ 90% of the ammonium load to the region. Significant efforts have been made in recent years
³³⁹ to reduce nitrogen loading from regional WWTPs given the disproportionate contribution
³⁴⁰ of nutrients relative to other sources (e.g., watershed agricultural load, sediment flux, etc.,
³⁴¹).^{29,51} Several WWTPs have recently been or are planned to be upgraded to include tertiary
³⁴² filtration and nitrification to convert biologically available ammonium to nitrate. The City
³⁴³ of Stockton WWTP was upgraded in 2006 and is immediately upstream of station P8.²⁸
³⁴⁴ Therefore, a modal response of nutrient concentrations at P8 centered around 2006 is ex-
³⁴⁵ pected as a result of upstream WWTP upgrades, and water quality should exhibit 1) a
³⁴⁶ shift in load contributions before/after upgrade, 2) a flow-normalized annual trend at P8
³⁴⁷ to show a change concurrent with WWTP upgrades, and 3) different nitrogen species will
³⁴⁸ have different changes depending on change in load outputs. The use of WRTDS to describe
³⁴⁹ downstream effects of WWTP upgrades could reveal flow-independent trends that have not
³⁵⁰ been previously described.

351 Overall reduction in total nitrogen load was observed as a result of reduction in ammonium (Figure S3). Nitrate is the primary constituent of total nitrogen after 2007. Organic
352 nitrogen is a larger percentage of the total after nitrification. What was reduction in ammonium starting in 2002?

355 Nitrogen trends at P8 shifted in response to upstream WWTP upgrades (Figure 6), with ammonium showing the largest reduction. Interestingly, nitrite/nitrate concentrations also
356 showed a similar but less dramatic decrease. Percent changes are shown in Table 4, where both nitrogen species shows large percent increases prior to WWTP upgrades followed by decreases after upgrades with ammonium showing the largest percentage. Seasonally, increases
357 prior to upgrades were most apparent in the July-August-September (JAS) months for both analytes. Seasonal reductions post-upgrades were also largest in JAS for nitrite/nitrate,
358 whereas percent reductions were similar across all monthly groupings for ammonium.
359

360 Relationships of nitrogen with flow showed the typical inverse flow/concentration dynamic {acro:jas}
361 with flushing at high flow, although patterns differed by nitrogen species. Seasonal variation
362 was more apparent for ammonium, although both typically had the highest concentrations in the winter. Additionally, strength of the flow/nutrient relationship changed throughout
363 the time series the year where the strongest relationship differed by analyte. Nitrite/nitrate
364 typically had the strongest relationship flow later in the time series, whereas ammonium had
365 the strongest relationship with flow in the early 2000s.

370 3.3.2 Effects of biological invasions

371 The San Francisco Estuary is considered one of the most invaded ecosystems in the world
372 with an estimated 234 exotic species by the turn of the century, half of which have been
373 reported after 1965.³⁸ The invasion of benthic grazers as ecosystem engineers is one of the
374 more notable events that has been characterized by dramatic shifts in primary production
375 of the Bay's trophic network.^{34,52-54} In particular, invasion of the upper estuary by the
376 Asian clam *Potamocorbula amurensis* in 1986 caused dramatic changes in phytoplankton

abundance and species composition with increased grazing. Reduction in phytoplankton biomass has altered trophic networks in the Bay and is considered a primary mechanism in the decline of the protected delta smelt and other important fisheries.^{55,56} Changes in the physical environment have also occurred with the most notable effect being increased water clarity following a reduction of phytoplankton.⁵⁶ The clams are halophilic such that drought years are generally correlated with an increase in biomass and further upstream invasion of the species.^{24,57}

We hypothesized that WRTDS models applied to water quality observations in the upper estuary would show 1) a decline in annual, flow-normalized chlorophyll concentrations over time coincident with an increase in abundance of invaders, 2) changes in ratios of limiting nutrients (nitrogen, SiO₂) suggesting different uptake rates by grazers with a shift in community composition, and 3) seasonal shifts in limiting nutrients based on changes in community composition and relative abundances with seasonal succession. The application of WRTDS to water quality observations at station D7 in Suisun Bay and comparison with clam abundance and biomass data from³⁴ was expected to reveal the competing effects of inflow on phytoplankton and benthic grazers.

Data from^{27,34} describes phytoplankton community changes in the upper estuary, including chlorophyll response to flow. Figure 10 in Ref. 27 showed that chlorophyll generally decreased with flow in 1980 but increased with flow in 2000.

Note the decrease in Potamocorbula abundance in 2011, 2012. These are wet years where abundance/biomass of the clams is driven down by lower salinity. Contrasted with the annual chlorophyll trends in the same years, the predicted values are above the flow-normalized trend suggesting an increase in chlorophyll with higher flow. The potential mechanism is therefore a decrease in clam abundance with high flow that releases phytoplankton from filtration pressure. This also explains the positive association of chlorophyll with flow in recent years (bottom right dynaplot). See suggestions in Ref. 57,58 regarding flow/grazer relationships in the Bay.

404 Further, chlorophyll trends early in the time series generally show a decrease with high
405 flow with a distinct maximum at moderate flow. This may suggest stratification events
406 at moderate flow contributed to phytoplankton blooms early in the time series. Water
407 withdrawals later in the time series could have also altered environmental conditions to
408 reduce the frequency occurrence of stratification events. Look into this more...

409 What about biomass/density relationships for Potamocorbula? Although clam density
410 increases throughout the period, What about initial decrease in chlorophyll prior to clam
411 invasion? Is this related to water withdrawals (i.e., decrease in stratification events at mod-
412 erate flow)?

413 Figure 7, Table 5

414 3.4 Implications

415 Implications of different flow varialbes - refer to,²⁹ a hybrid approach would be best for
416 complex systems like the delta.

417 Second case study showed typical inverse relationships between nutrients and flow, more
418 flow means greater flushing and dilution of nutrient concentrations. Conversely, low flow
419 means less flushing and higher nutrient concentrations, although this may not always be
420 observed if the available nutrients are biologically available. Low-flow events during warmer
421 months show the lowest ammonium concentrations, which corresponds to seasonal max-
422 ima in chlorophyll concentration. A similar but weaker relationship was observed with
423 nitrite/nitrate where increased flow was related to decreased concentration and lower con-
424 centrations overall were observed in the summer. However, low-flow events still had higher
425 concentrations than high-flow events in July, as compared to ammonium which was low re-
426 gardless of flow. This suggests that ammonium concentrations are driving phytoplankton
427 production at P8. Annual trends in chlorophyll concentration (not shown) showed an overall
428 decrease from the 1970s to present, although a slight peak is observed in the 2000s. This peak
429 is likely related to the maximum ammonium concentration shown in Figure 6. Moreover,

430 flow/chlorophyll relationships have generally been constant throughout the period of record
431 such that a change in flow has not been related to a change in phytoplankton production.
432 This suggests that nutrient loads that contribute to production at P8 are primarily from
433 point sources at WWTP outflows as a change in flow does not affect the load output. But
434 what are watershed loads?

435 What do nitrogen trends mean? Have to interpret relative to trends in other variables.
436 A decrease in nitrogen or constant nitrogen does not mean nitrogen inputs have stayed the
437 same, they might actually be increasing if nitrogen. A change in chlorophyll relative to
438 change in nitrogen could be informative, and even moreso, a change in silica relative to
439 change in chlorophyll suggests diatom biomass has changed. However, there are mismatches
440 in these trends that suggest other processes are at play, e.g., residence times and flow in-
441 puts, etc. Trends in Suisun relative to trends in Delta provide an example, e.g., Suisun is
442 decrease in chlorophyll, increase in silica, increase in nitrogen, delta is decrease in silica, in-
443 crease/decrease in DIN (depending on time period/season), decrease in chlorophyll, what's
444 going on? See Senn slide 14 (from burial?). The WRTDS model lets us at least address
445 trends in the context of season, time, and flow. This allows for more improved interpretation
446 relative to observing raw data. Also explain more information by looking at ammonium,
447 nitrate/nitrite, relative to DIN. What about other variables (light level as suspended par-
448 ticulate matter, temperature)?

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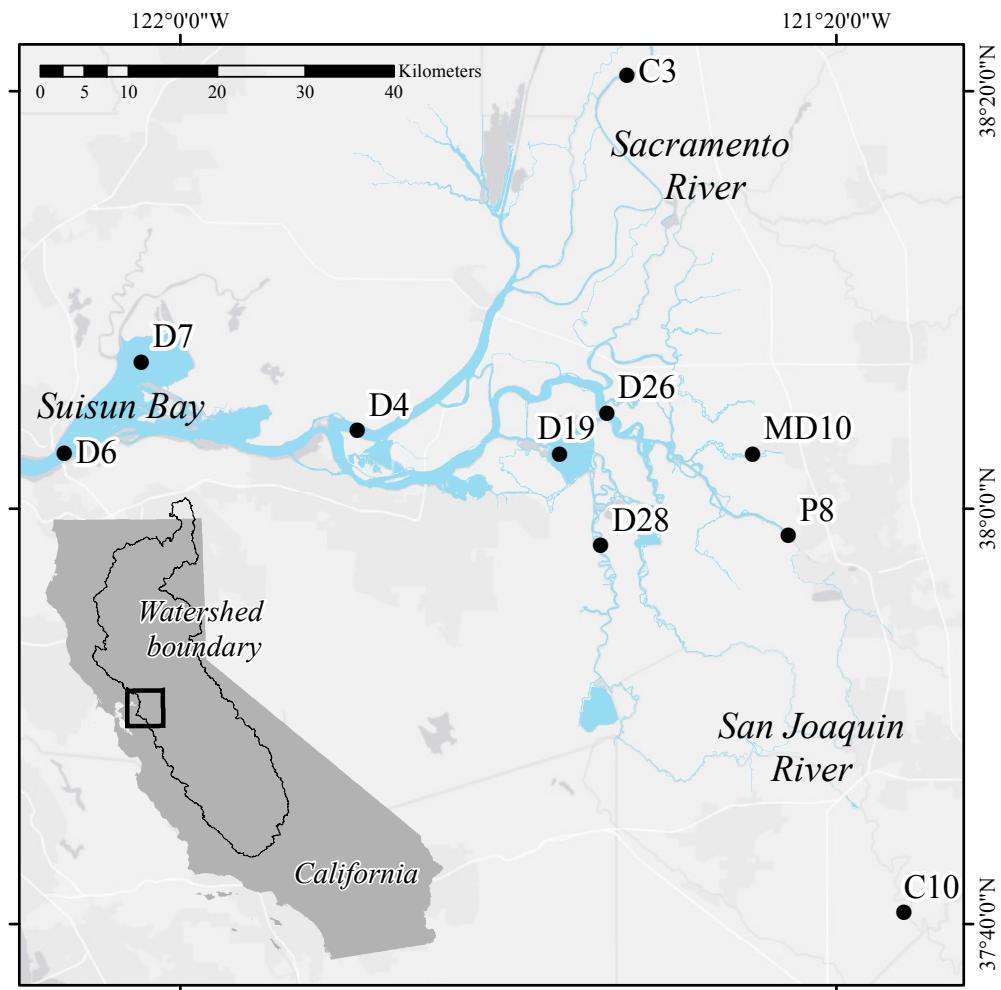


Figure 1: The San Francisco Estuary Delta and monitoring stations used for analysis. The Delta drains the combined watershed from the Sacramento and San Joaquin rivers (bottom left). All data were obtained from the Interagency Ecological Program website (<http://water.ca.gov/bdma/meta/Discrete/data.cfm>).⁴³

{fig:delt_m}

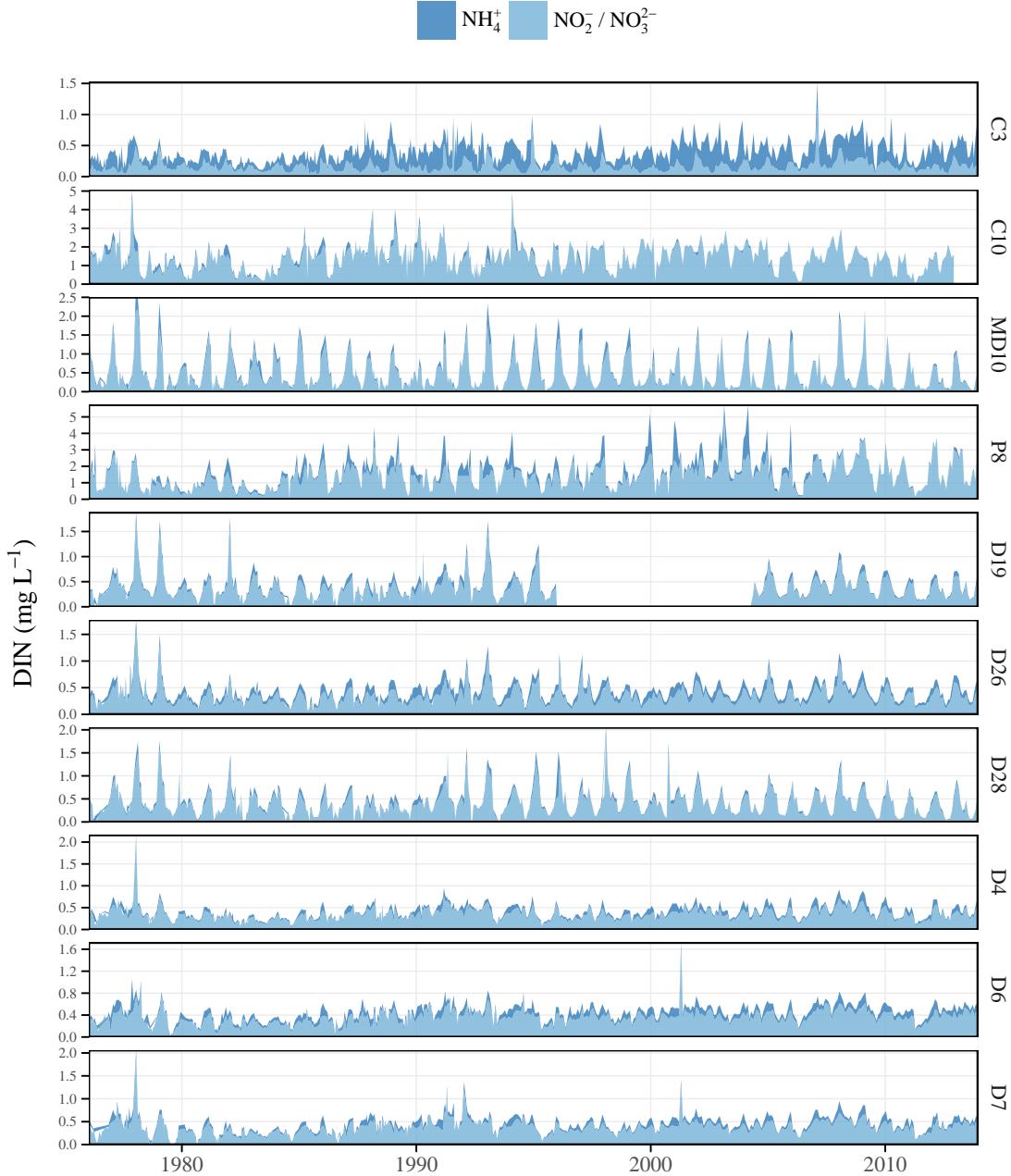


Figure 2: Observed DIN ($\text{NH}_4^+ + \text{NO}_2^-/\text{NO}_3^{2-}$) from ten stations in the upper SFE Delta. Data were collected monthly and evaluated with WRTDS models from 1976 to 2013. Note different y-axis scales. See Figure 1 for station locations.

{fig:obsdat}

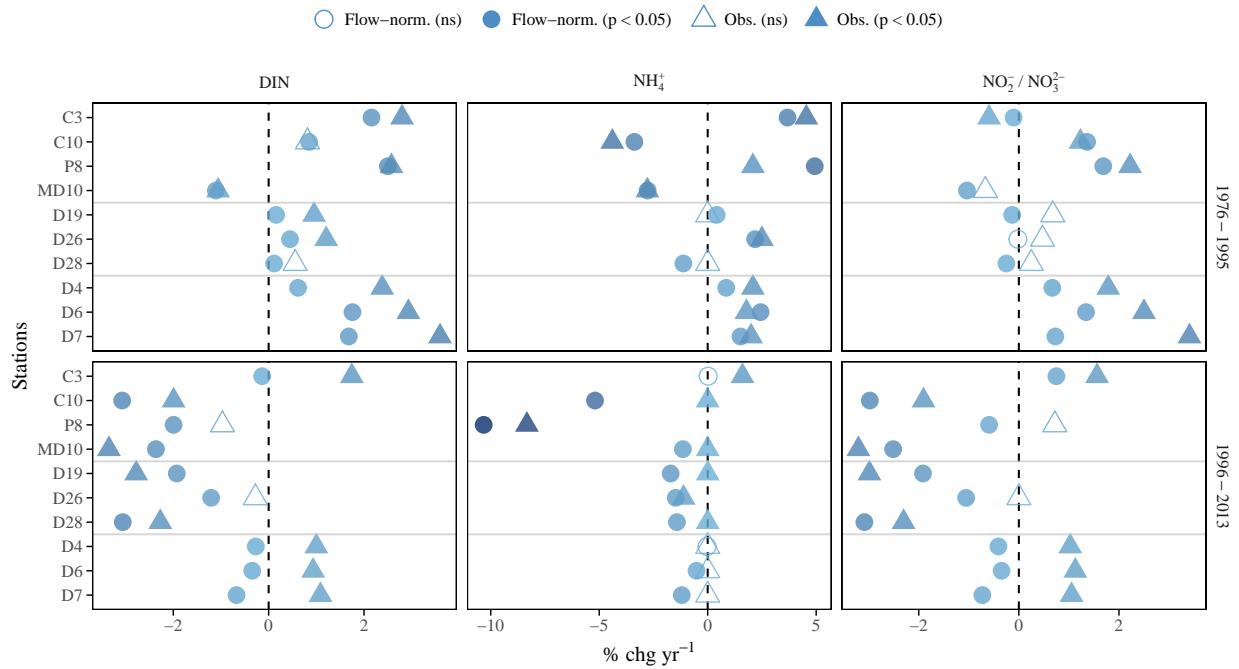


Figure 3: Results from seasonal Kendall tests on observed data (triangles) and flow-normalized predictions (circles) from WRTDS for nitrogen analytes. Results are shown as the percent change per year as the estimated Theil-Sen slope divided by the median for a given aggregation period (significance evaluated at $\alpha = 0.05$, based on τ). Trends are shown separately for different annual groupings. See Figures S1 and S2 for seasonal groupings.

{fig:trndcc}

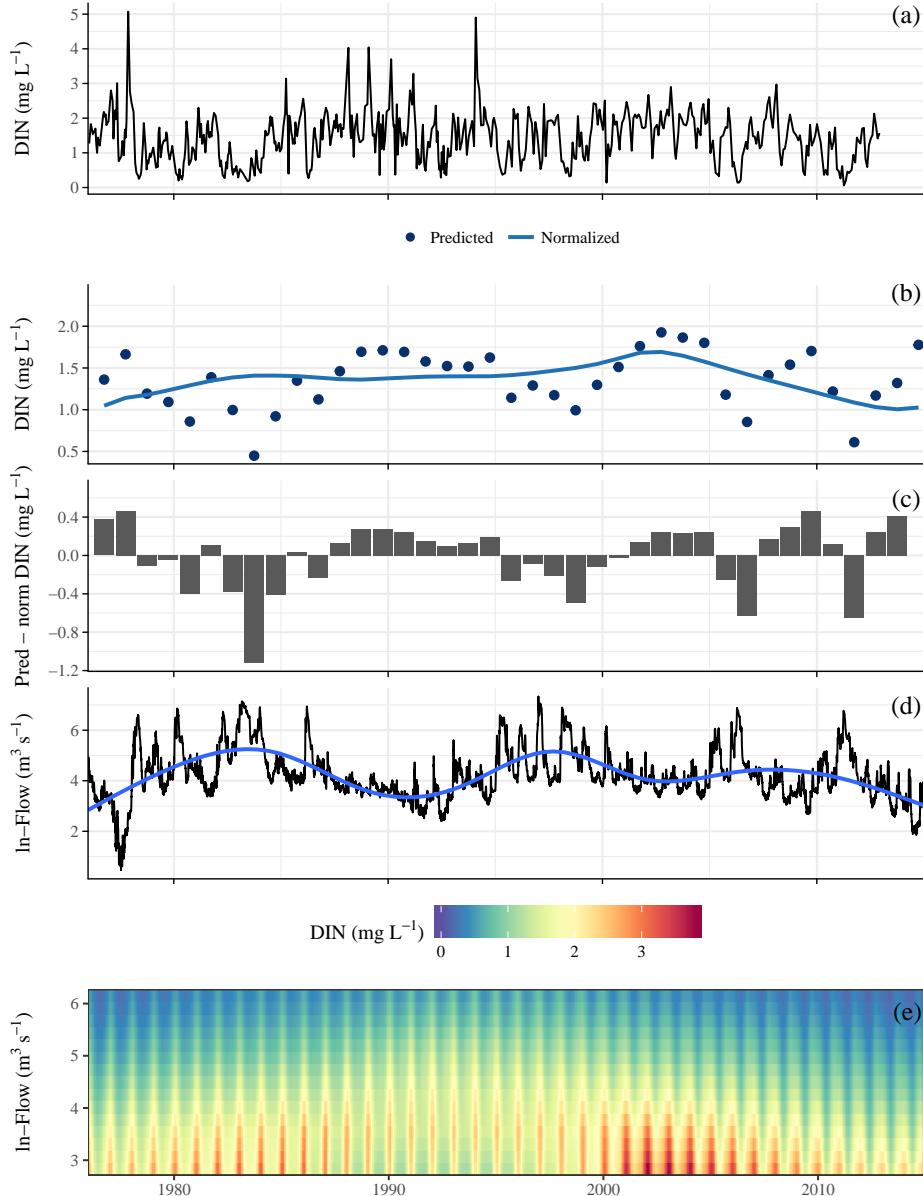


Figure 4: Time series of DIN and flow at station C10. Subfigure (a) shows the observed DIN time series and subfigure (b) shows the annual (water year starting in October) predictions from WRTDS for the conditional mean response. The points in subfigure (b) are predictions of observed DIN and the lines are flow-normalized predictions. Subfigure (c) shows the difference between the model predictions and flow-normalized predictions. Subfigure (d) shows the flow time series of the San Joaquin River with a locally-estimated (loess) smooth to emphasize the long-term trend. Subfigure (e) shows the modelled relationships between DIN, flow, and time. Flow values in (e) are truncated by 5th and 95th percentiles of observed. {fig:dinc10}

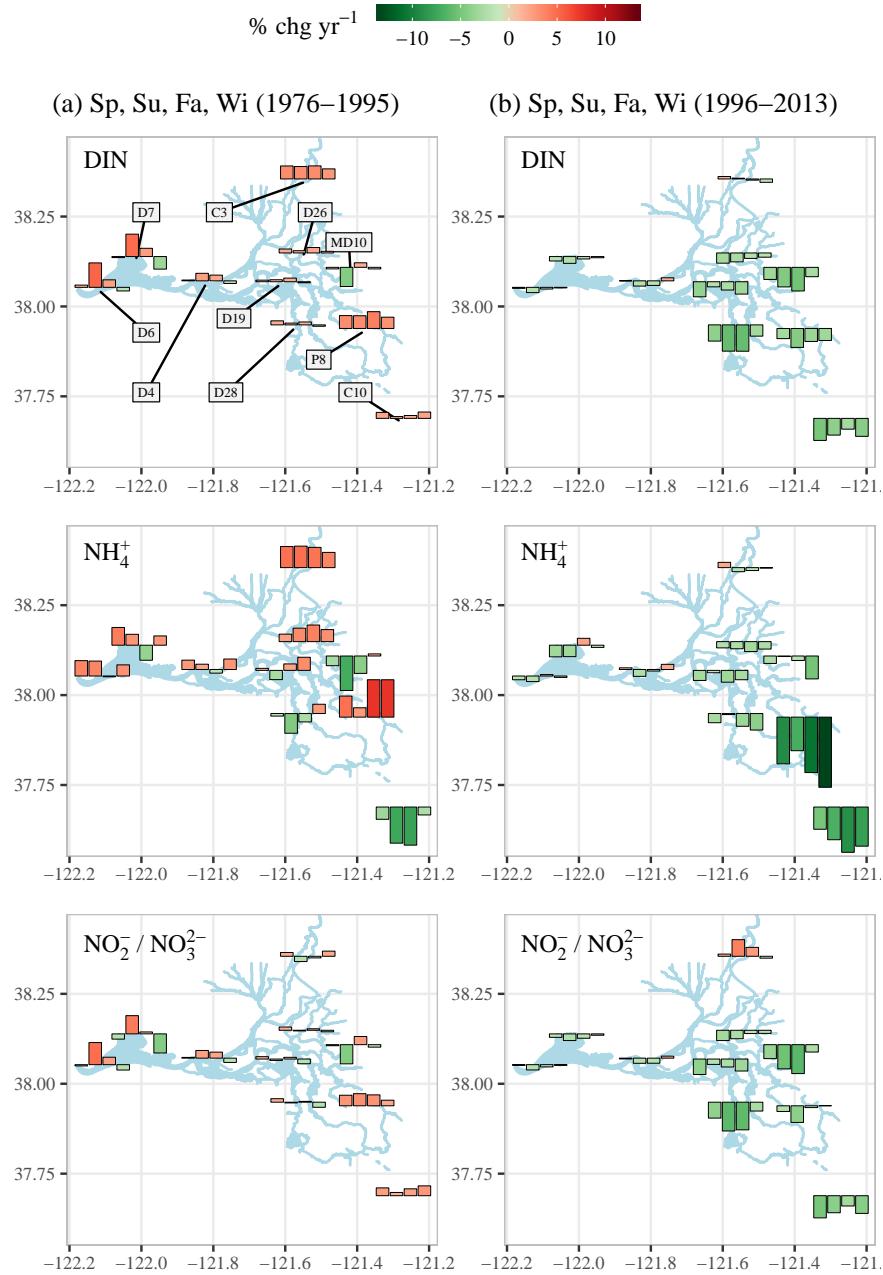


Figure 5: Percent change per year in nitrogen analytes for aggregations by seasons from (a) 1976-1995 and (b) 1996-2013. Changes are based on seasonal Kendall tests of flow-normalized results within each time period. Station names are shown in the top left panel. Station locations have been jittered to reduce overlap. Months for each season are Spring: MAM, Summer: JJA, Fall: SON, Winter: DJF.

{fig:trndma}

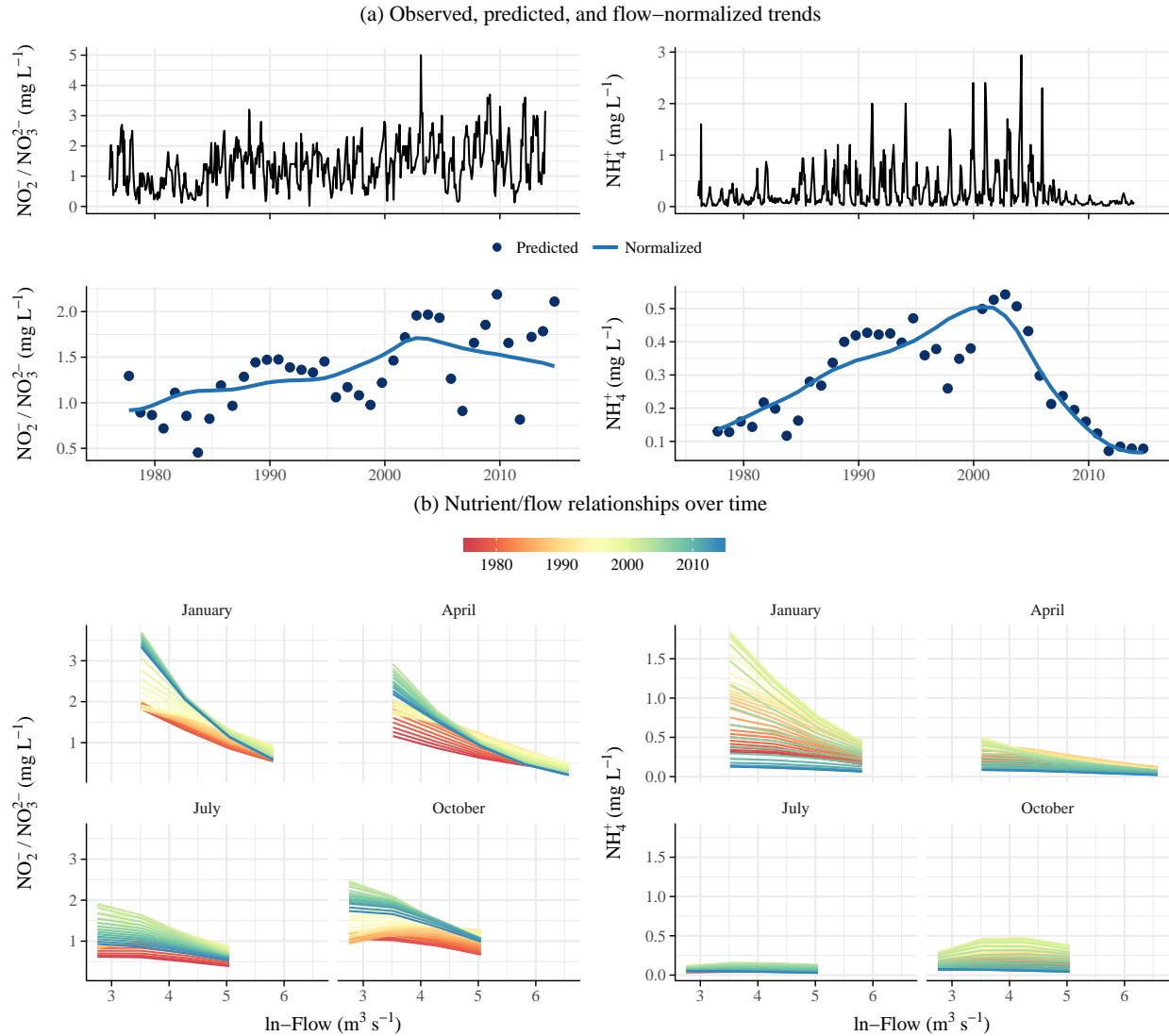


Figure 6: Nitrogen trends at P8 as observed (a, top), predicted and flow-normalized estimates from WRTDS (a, bottom), and relationships with flow over time (b). Nitrite/nitrate trends are on the left and ammonium trends are on the right. Wastewater treatment plant upgrades at the City of Stockton (San Joaquin County) were completed in 2006 (Figure S3), coincident with a dramatic decrease in ammonium at P8.

{fig:p8trnd}

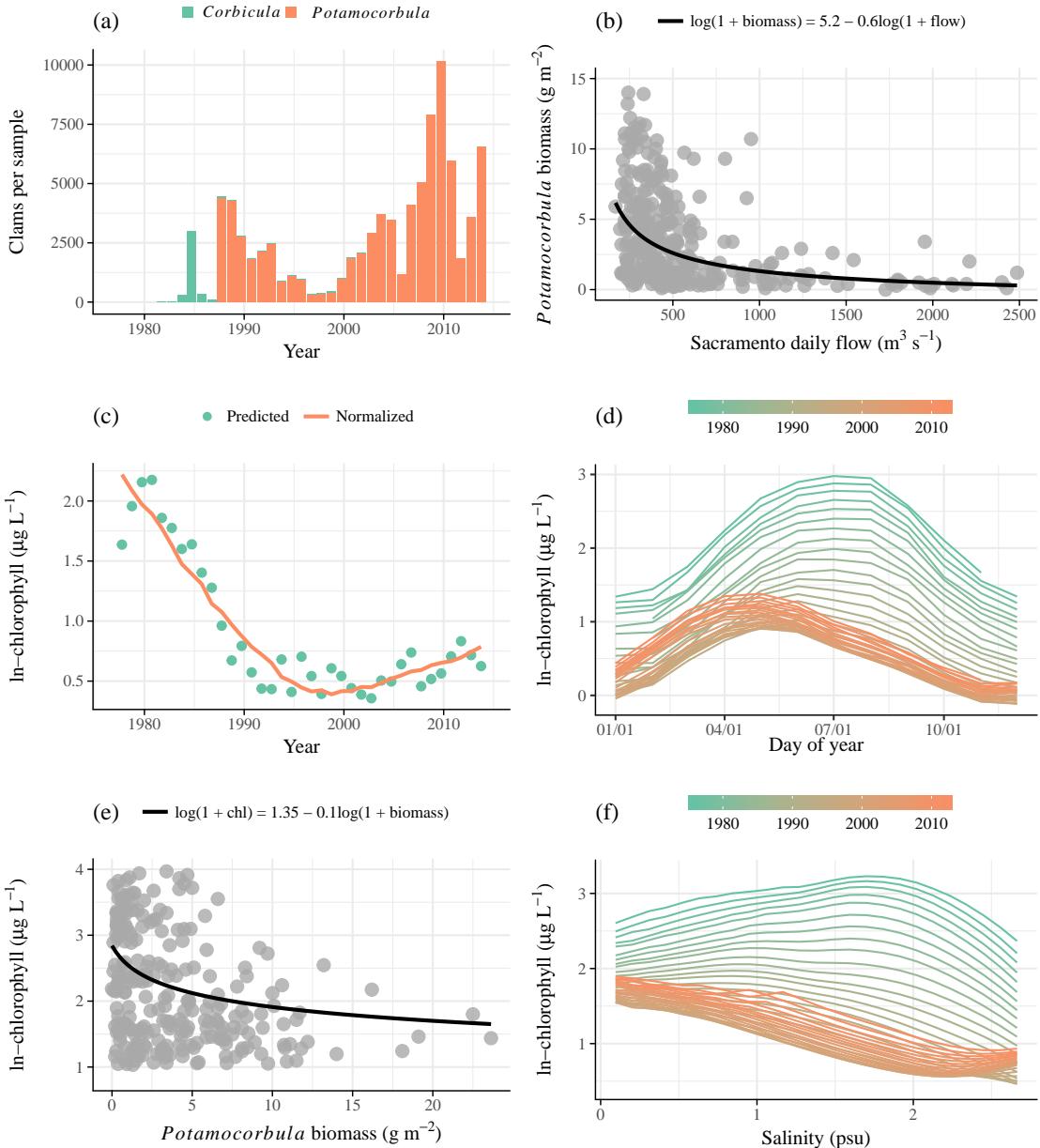


Figure 7: Trends in clam abundance and chl-*a* concentration from 1976 to 2013 at station D7 in Suisun Bay. Invasion by *Potamocorbula amurensis* clams in the late 1980s and displacement of *Corbicula fluminea* was shown by changes in clam density (a, annual means), with biomass linked to salinity (b). A coincident decrease in chl-*a* concentration was also observed by changes in annual (c) and seasonal trends(d). A significant ($p < 0.001$) relationship between clam biomass and chl-*a* concentration is shown in subfigure (e). Flow relationships with chl-*a* concentration have also changed over time (f, observations from June).

{fig:clmchl}

Table 1: 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-italic). Changes and significance estimates are based on seasonal Kendall tests of flow-normalized results within each time period. See Figure 5 for a summary of spatial trends.

{tab:trndsa}

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)**
NH₄⁺		
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)**
NO₂⁻/NO₃²⁻		
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)**

* $p < 0.05$; ** $p < 0.005$

Table 2: 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-italic). Changes and significance estimates are based on seasonal Kendall tests of flow-normalized results within each time period. See Figure 5 for a summary of spatial trends. Months for each season are Spring: MAM, Summer: JJA, Fall: SON, Winter: DJF.

{tab:trndsb}

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)**
NH₄⁺				
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)**
NO₂⁻/NO₃²⁻				
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)**

* $p < 0.05$; ** $p < 0.005$

Table 3: 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-italic). Changes and significance estimates are based on seasonal Kendall tests of flow-normalized results within each time period. See Figure 5 for a summary of spatial trends. Months for each season are Spring: MAM, Summer: JJA, Fall: SON, Winter: DJF.

{tab:trndsa}

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)**
NH₄⁺				
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)**
NO₂⁻/NO₃²⁻				
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)

* $p < 0.05$; ** $p < 0.005$

Table 4: 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, Figure S3). Summaries are medians and percent change per year in parentheses (increasing in bold-italic). 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.

{tab:p8chg}

Period	$\text{NO}_2^-/\text{NO}_3^{2-}$		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**

* $p < 0.05$; ** $p < 0.005$

Table 5: Summaries of flow-normalized trends in dissolved inorganic nitrogen (mg L^{-1}), chlorophyll ($\mu\text{g L}^{-1}$), and silicon dioxide (mg L^{-1}) concentrations for different time periods at station D7. Summaries are medians and percent change per year in parentheses (increasing in bold-italic). 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.

{tab:d7ch}

Period	DIN		Chl-a		SiO ₂	
	Median	% change	Median	% change	Median	% change
All						
1976-2013	0.4	0.6**	3	-6.7**	12.7	0.7**
Annual						
1976-1985	0.4	-2.1**	8.8	-10.7**	10.2	-0.2
1986-1994	0.4	3.6**	2.6	-13.5**	11.9	2.3**
1995-2003	0.5	-0.1	1.8	1.9**	13.3	0.7**
2004-2013	0.5	-1.3**	2.1	2.9**	13.1	-0.3**
Seasonal						
Spring	0.5	-0.1	3.4	-1	14.7	0.1**
Summer	0.4	1.5**	3.4	-8.8**	12.2	1.2**
Fall	0.4	0.6**	1.7	-8.8**	12.1	1**
Winter	0.6	-0.2	1.4	-3.1**	14.5	0.3**

* $p < 0.05$; ** $p < 0.005$

⁶¹⁰ **Supporting Information Available**

⁶¹¹ The following files are available free of charge.

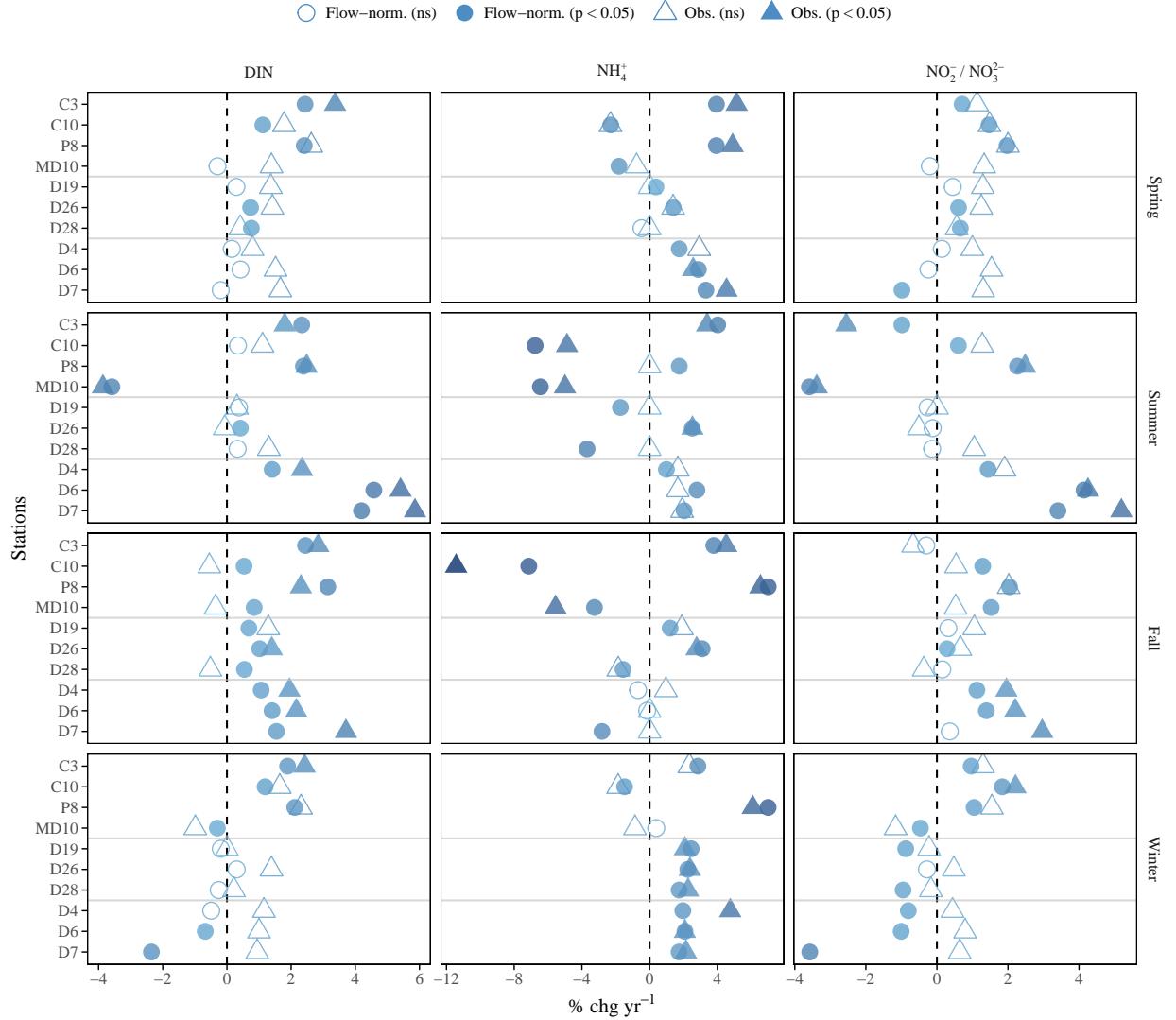


Figure S1: Results from seasonal Kendall tests on observed data (triangles) and flow-normalized predictions (circles) from WRTDS for nitrogen analytes. Results are shown as the percent change per year as the estimated Theil-Sen slope divided by the median for a given aggregation period (significance 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 Figure 3 for annual comparisons.

{fig:trndcc}

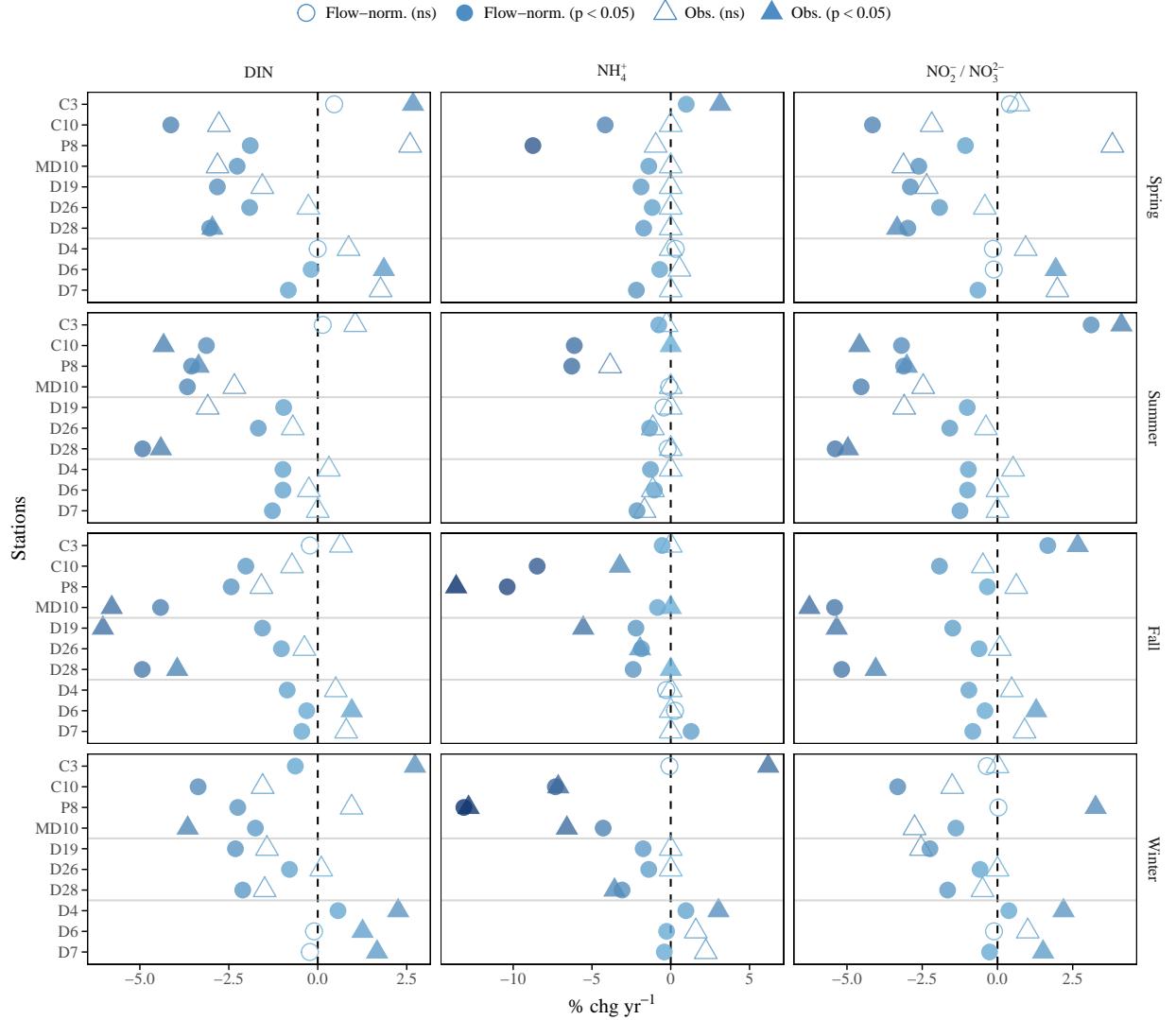


Figure S2: Results from seasonal Kendall tests on observed data (triangles) and flow-normalized predictions (circles) from WRTDS for nitrogen analytes. Results are shown as the percent change per year as the estimated Theil-Sen slope divided by the median for a given aggregation period (significance 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 Figure 3 for annual comparisons.

{fig:trndcc}

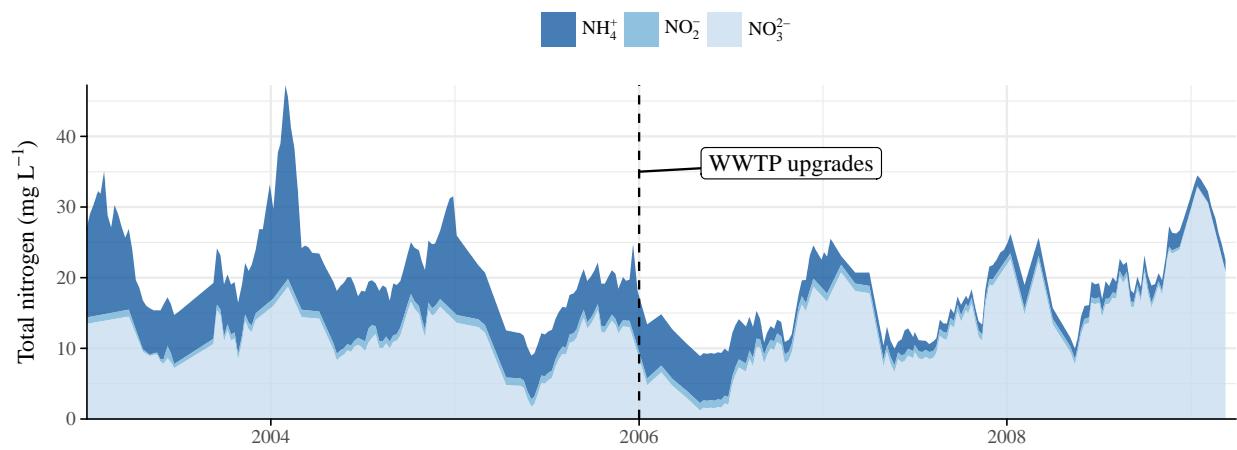


Figure S3: Nitrogen concentration measurements (mg L^{-1}) from the City of Stockton Wastewater Treatment Plant, San Joaquin County. Wastewater discharge requirements were implemented in 2006 for nitrification/denitrification and tertiary filtration to convert ammonium to nitrate.

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