



San Luis & Delta-Mendota Water Authority

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March 10, 2014

Delivered Via E-mail to robert.schlipf@waterboards.ca.gov

Mr. Robert Schlipf
California Regional Water Quality Control Board
San Francisco Bay Region
1515 Clay Street, Suite 1400
Oakland, CA 94612

Subject: Comments on the Tentative Order for Municipal Wastewater Discharges of Nutrients to San Francisco Bay, NPDES Permit

Dear Mr. Schlipf:

The State Water Contractors (SWC) and San Luis & Delta-Mendota Water Authority (Authority)¹ (collectively, "Public Water Agencies") appreciate the opportunity to provide comments on the San Francisco Bay Regional Water Quality Control Board's (Regional Water Board) Tentative Order for Municipal Wastewater Discharges of Nutrients to San Francisco Bay (tentative permit). The Public Water Agencies have a significant interest in the health of the San Francisco Bay/Sacramento-San Joaquin River Delta (Bay-Delta) ecosystem and in the development of management actions to address stressors on the Bay-Delta, including water quality impairment. With respect to municipal wastewater discharges, we support the development of nutrient water quality objectives and the implementation of management actions to address nutrient impacts in the Bay-Delta ecosystem, and we are active participants in the Regional Water Board's Nutrient Management Strategy.

The Public Water Agencies commend the Regional Water Board for its efforts to address nutrients discharged in municipal wastewater in a comprehensive manner for the San Francisco Bay region through the development of the tentative permit. We support the efforts of the Regional Water Board to collect additional nutrient data, require evaluation of treatment options for nutrient discharge reduction and require the municipal wastewater treatment agencies (Dischargers) to support science studies to implement the San Francisco Bay Nutrient Management Strategy. Effluent and receiving water monitoring will further characterize nutrient concentrations in receiving waters and provide data for modeling studies. The evaluation of treatment optimization and treatment upgrade options to reduce nutrient

¹ For a description of the SWC and San Luis & Delta-Mendota Water Authority see Attachment 1.

discharges will provide additional information that the Regional Water Board can use to make decisions on nutrient management actions. We support the consideration of side-stream treatment as one of the optimization options for wastewater treatment plants, as we believe in some circumstances it is an important and ready tool for incremental and prompt nutrient load reduction while the Board develops a comprehensive Nutrient Management Strategy.

The Public Water Agencies also support the requirement for the Dischargers to fund the development and implementation of nutrient studies. The studies and modeling will support the implementation of the Nutrient Management Strategy and the development of nutrient management actions, including needed nutrient load reductions. However, the Monitoring, Modeling and Embayment Studies section of the tentative permit requires additional detail to be effective. Accordingly, the Public Water Agencies recommend the tentative permit be revised to include more detail on the required studies to be implemented, including a detailed schedule for conducting the studies. Further, we also recommend that the tentative permit be revised to provide opportunities for interested parties to provide input to the monitoring, modeling and nutrient studies as they are implemented.

That said, and despite the value of conducting additional monitoring and collecting additional data as contemplated by the tentative permit, the Public Water Agencies believe there is already more than enough scientific justification to require nutrient load reductions for wastewater treatment plant facilities now. As the State Water Resources Control Board (SWRCB) found when upholding the discharge permit issued to the Sacramento Regional County Sanitation District, "the historical resilience of the Bay-Delta ecosystem to excess anthropogenic nutrient loading is weakening and may be nearing an irreversible tipping point." (SWRCB Order WQ 2012-0013 at 31) "Additionally, there is enough evidence in the record of cyanobacteria blooms in the Delta, and other phytoplankton blooms in the San Francisco Bay (including blooms of *Heterosigma akashiwo*) to demonstrate that excessive biostimulation is occurring, even if diatom populations in Suisun Bay are not experiencing bloom conditions" (SWRCB Order WQ 2012-0013 at 31). And, "[e]vidence is present in the record for each of these indicators of cultural eutrophication in the Delta and San Francisco Bay with the current nutrient loads. The Northern San Francisco Bay, specifically Suisun Bay, has undergone significant changes in ecosystem structure. These changes are presently being attributed to ecosystem perturbations over the past several decades resulting from changes in nutrient ecosystem stoichiometry" (SWRCB Order WQ 2012-0013 at 32).

Those wastewater treatment plants discharging to Suisun Bay present additional concerns. As noted on page F-17 of the tentative permit, Suisun Bay is characterized as having extremely low phytoplankton biomass and a highly altered phytoplankton community composition. The attached Nutrient Science Summary Technical Memorandum (Attachment 3) documents nutrient impacts on aquatic life in the Bay-Delta Estuary, including Suisun Bay. Published studies have repeatedly shown, among other things, that the total loads, the forms, and the relative proportions of nutrients have changed over time. These changes – due in large measure to the untreated ammonia and nitrogen discharges from wastewater treatment plants – have impacted the Bay-Delta Estuary, resulting in declines in primary production and changes in phytoplankton community composition. Ultimately these changes impact lower trophic levels of the Bay-Delta food web dependent on phytoplankton.

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The Public Water Agencies' detailed comments on specific sections of the tentative permit are provided in Attachment 2. We appreciate the opportunity to provide comments on the tentative permit and look forward to continuing to work with the Regional Water Board on nutrient issues. If you have any questions concerning these comments, please contact Lynda Smith, Metropolitan Water District of Southern California, at 916-650-2632 or lsmith@mwdh2o.com.

Sincerely,



Daniel G. Nelson
Executive Director
San Luis & Delta-Mendota Water Authority



Terry L. Erlewine
General Manager
State Water Contractors

Attachments

Attachment 1

San Luis & Delta-Mendota Water Authority: The San Luis & Delta Mendota Water Authority (Authority) is a joint powers authority, established under California's Joint Exercise of Powers Act. (Gov. Code, § 6500 et seq.). The Authority is comprised of 29 member agencies, 27 of which hold contractual rights to water from the federal Central Valley Project (CVP). The Authority member agencies have historically received up to 3,100,000 acre-feet annually of CVP water for the irrigation of highly productive farm land primarily along the San Joaquin Valley's Westside, for municipal and industrial uses, including within California's Silicon Valley, and for publicly and privately managed wetlands situated in the Pacific Flyway. The areas served by the Authority's member agencies span portions of seven counties encompassing about 3,300 square miles, an area roughly the size of Rhode Island and Delaware combined. The Authority's members are: Banta-Carbona Irrigation District; Broadview Water District; Byron Bethany Irrigation District (CVPSA); Central California Irrigation District; City of Tracy; Columbia Canal Company (a Friend); Del Puerto Water District; Eagle Field Water District; Firebaugh Canal Water District; Fresno Slough Water District; Grassland Water District; Henry Miller Reclamation District #2131; James Irrigation District; Laguna Water District; Mercy Springs Water District; Oro Loma Water District; Pacheco Water District; Pajaro Valley Water Management Agency; Panoche Water District; Patterson Irrigation District; Pleasant Valley Water District; Reclamation District 1606; San Benito County Water District; San Luis Water District; Santa Clara Valley Water District; Tranquillity Irrigation District; Turner Island Water District; West Side Irrigation District; West Stanislaus Irrigation District; Westlands Water District.

State Water Contractors, Inc. (SWC): The SWC organization is a nonprofit mutual benefit corporation that represents and protects the common interests of its 27 member public agencies in the vital water supplies provided by California's State Water Project ("SWP"). Each of the member agencies of the State Water Contractors holds a contract with the California Department of Water Resources ("DWR") to receive water supplies from the SWP. Collectively, the SWC members deliver water to more than 25 million residents throughout the state and more than 750,000 acres of agricultural lands. SWP water is served from the San Francisco Bay Area, to the San Joaquin Valley and the Central Coast, to Southern California. The SWC's members are: Alameda County Flood Control and Water Conservation District Zone 7; Alameda County Water District; Antelope Valley-East Kern Water Agency; Casitas Municipal Water District; Castaic Lake Water Agency; Central Coastal Water Authority; City of Yuba City; Coachella Valley Water District; County of Kings; Crestline-Lake Arrowhead Water Agency; Desert Water Agency; Dudley Ridge Water District; Empire-West Side Irrigation District; Kern County Water Agency; Littlerock Creek Irrigation District; Metropolitan Water District of Southern California; Mojave Water Agency; Napa County Flood Control and Water Conservation District; Oak Flat Water District; Palmdale Water District; San Bernardino Valley Municipal Water District; San Gabriel Valley Municipal Water District; San Geronio Pass Water Agency; San Luis Obispo County Flood Control & Water Conservation District; Santa Clara Valley Water District; Solano County Water Agency; and Tulare Lake Basin Water Storage District.

Attachment 2

Public Water Agencies Detailed Comments on the Tentative Order for Municipal Wastewater Discharges of Nutrients to San Francisco Bay, NPDES Permit

Page 4, section VI. C. 2, Evaluation of Potential Discharge Reduction by Treatment Upgrades or Other Means. The tentative permit needs to specify nutrient load reduction targets (e.g. 90%, 95%, 99%) or effluent nutrient concentration targets, and specify which nutrient constituents (e.g., nitrate, ammonia, total nitrogen, phosphate, total phosphorus) for the evaluation of treatment upgrades.

Page 6, section VI. C. 3. a, Support Science Plan Development and Implementation. The tentative permit should include more detail on the science studies required to implement the San Francisco Bay Nutrient Management Strategy, and specifically provide for a process that allows interested parties to provide input on the implementation plan and schedule, and the details of the studies.

Page 7, section VI. C. 3. b, Support Receiving Water Monitoring for Nutrients. While the tentative permit includes detail on the effluent monitoring requirements, the receiving water monitoring requirements are very general and require more detail.

- The tentative permit should include more detail on what is required by a network of monitoring stations. How will this network be determined? How often will monitoring occur and at what depths? Will there be opportunity for input from interested parties? How will the data be made available for review by the public in a timely fashion and a usable format in order to ensure a transparent process and allow for meaningful public participation?
- The tentative permit should specify which nutrients will be monitored (e.g. does this include all N and P species including urea?). Is it the same list that is specified for effluent monitoring in Attachment E? Also, the tentative permit should specify method and detection levels.
- The tentative permit receiving water monitoring requirements should also include phytoplankton speciation in order to evaluate whether and how species composition has shifted in response to nutrients.
- The tentative permit needs to include more detail on the harmful algae bloom studies. How will these studies be determined? Is there an approval step by the Regional Water Board Executive officer prior to study initiation? This is another area where it is important to provide an opportunity for input from interested parties. At a minimum, the tentative permit should indicate how many monitoring stations, how frequently the monitoring needs to occur, and which toxins will be monitored at what detection levels.

Page E-2, Table E-2, Effluent Monitoring Requirements. The tentative permit should specify analytical method and detection level for each of these constituents, or reference where these conditions are specified.

Page F-8 and F-9, I. C. The tentative permit presents a very long time period for moving from nutrient monitoring, studies and treatment evaluations specified in this tentative permit, to requiring implementation of additional nutrient management actions in 2024 and 2029 permit reissuances. This is an excessively lengthy time period for management actions, and does not reflect the state of the science on nutrient impacts. As stated earlier in these Public Water Agencies comments, there is already more than enough scientific justification *now* to require nutrient load reductions for wastewater treatment plant facilities, especially those discharging to Suisun Bay. The schedule presented in the tentative permit appears to pre-judge the outcome of ongoing nutrient studies as not justifying implementation of nutrient management actions. The Public Water Agencies request that the language in the tentative permit be modified to provide for implementation of nutrient management actions by no later than the 2019 permit reissuance.

Page F-13, section IV, Rationale for Discharge Prohibitions and Specifications. The Public Water Agencies respectfully disagree with the Regional Water Board statement that there is insufficient evidence to conclude that nutrients cause or contribute to excursions of the narrative water quality objective for biostimulatory substances. As stated earlier in these comments, consistent with the findings made by the State Water Resources Control Board – supported by a wide range of federal environmental, fish and wildlife agencies - there is already more than enough scientific justification now to require nutrient load reductions for wastewater treatment plant facilities, especially those discharging to Suisun Bay. As noted on page F-17 of the tentative permit, Suisun Bay is characterized as having extremely low phytoplankton biomass and a highly altered phytoplankton community composition. The attached Nutrient Science Summary Technical Memorandum (Attachment 3) summarizes nutrient impacts on aquatic life in the Bay-Delta Estuary, including Suisun Bay, from the published literature. Published studies have shown that the total loads, the forms, and the relative proportions of nutrients have changed over time. These changes have impacted the Bay-Delta Estuary, resulting in declines in primary production and changes in phytoplankton community composition. Ultimately these changes impact lower trophic levels of the Bay-Delta food web dependent on phytoplankton.

Relevant narrative objectives that need to be considered in determining nutrient impairment include the following:

3.3.3 Biostimulatory Substances

- Waters shall not contain biostimulatory substances in concentrations that promote aquatic growths to the extent that such growths cause nuisance or adversely affect beneficial uses. Changes in chlorophyll a and associated phytoplankton communities follow complex dynamics that are sometimes associated with a discharge of biostimulatory substances. Irregular and extreme levels of chlorophyll a or phytoplankton blooms may indicate exceedance of this objective and require investigation. (2011 WQCP p3-4)

3.3.8 Population and Community Ecology

- All waters shall be maintained free of toxic substances in concentrations that are lethal to or that produce significant alterations in population or community ecology or receiving water biota. In addition, the health and life history characteristics of aquatic organisms

in waters affected by controllable water quality factors shall not differ significantly from those for the same waters in areas unaffected by controllable water quality factors. (2011 WQCP p 3-5)

3.3.18 Toxicity

- All waters shall be maintained free of toxic substances in concentrations that are lethal to or that produce other detrimental responses in aquatic organisms. Detrimental responses include, but are not limited to, decreased growth rate and decreased reproductive success of resident or indicator species. (2011 WQCP, page 3-7)

Page F-16, section VI. C. 3. Monitoring, Modeling and Embayment Studies. The tentative permit states that the Bay Area Clean Water Agencies (BACWA) has identified \$880,000 each permit year as a collective level of effort from the Dischargers, and further states that the Regional Water Board finds this amount to be appropriate. This determination about adequacy of the Discharger funding needs to be more transparent. There is no information in the tentative permit to support the statement that \$880,000 is adequate. The tentative permit should include more detail on monitoring, modeling and nutrient studies, a schedule and proposed budget that demonstrates the required studies will have sufficient funding for full implementation.

Page F-17, section VI. C. 3. Monitoring, Modeling and Embayment Studies. The tentative permit states that low phytoplankton biomass and a highly altered phytoplankton community composition have characterized the Suisun Bay system since 1987. Published work indicates that changes in Suisun Bay phytoplankton composition began before the invasive clam *Potamocorbula* became established (Glibert et al., 2011 p.371)

Page F-17, footnote 7. It is preferable to cite the underlying research rather than a draft report. See Wilkerson et al 2006; Dugdale et al 2007; Dugdale et al 2012; Dugdale et al 2013; Glibert et al 2011; and Glibert et al 2013. Specific references are provided in Attachment 3.

Technical Memorandum Nutrient Science Summary

February 1, 2014

Historically, scientists have described primary productivity in the Bay-Delta as not being limited by nutrients (Cloern 2001; Lopez et al. 2006) and not experiencing signs of classic eutrophication (Cloern 2001). Such classic eutrophication effects include increased chlorophyll *a* (chl-*a*) in the water column (i.e., more algal blooms), development of hypoxia or anoxia (low or no dissolved oxygen, respectively), loss of native submerged aquatic vegetation, increased proportion of those algae that are considered harmful algal blooms (HABs), and changes in biodiversity, including loss of certain fisheries (e.g., Cloern 2001; Anderson et al. 2002). However, we now recognize that changes in nutrient loads are affecting ecosystem dynamics in complex ways that extend beyond our historic understanding of the process of eutrophication.

Nutrient effects on aquatic systems are far more complex than those normally associated with eutrophication and can occur across the full spectrum of nutrient limitation to nutrient super-saturation (Glibert et al 2013). Even relatively small changes in nutrient supply – even when those nutrients are not limiting for primary production - are being shown to have large consequences on many important properties of ecosystems (Nielsen, 2003). Two important properties of nutrient supply that have effects beginning at the level of the physiology of primary producers (algae) and propagating through the food web are the form in which nutrients are supplied (chemical state, oxidized vs. reduced, organic vs. inorganic, dissolved vs. particulate) and the proportion of different elements (including carbon (C), nitrogen (N), phosphorous (P), and silicon (Si), among others). In the Bay-Delta, the total loads, the forms, and the relative proportions of nutrients have been changing over time. These changes have had profound effects on ecosystem structure of this system, as documented below. This technical memorandum describes the effects of two such changes: the proportion of ammonium to nitrate ($\text{NH}_4:\text{NO}_3$) and the proportion of nitrogen to phosphorus (N:P).

Ammonium:Nitrate

For decades, researchers have explored the relative use – or relative preference for – different forms of nitrogen (N) by phytoplankton. Ammonium (NH_4) is generally considered to be the form of nitrogen preferred by phytoplankton due to the more favorable energetics associated with its assimilation compared to that of nitrate (NO_3). Ecological consequences of dependence on NO_3 vs. NH_4 have also been well recognized for decades. Based on the concept of “new” and “regenerated” production (Dugdale and Goering 1967, Eppley and Peterson 1979), phytoplankton dependence on NH_4^+ leads to production that is cycled within the microbial loop, whereas that based on NO_3^- more often leads to production that supports a food web leading to secondary production as well as export out of the euphotic zone (Dugdale and Goering, 1967; Eppley and Peterson, 1979). Shifts in nitrogen (N) form from NO_3^- to NH_4^+ have been shown in numerous systems to lead to community shifts away from plankton communities dominated by diatoms to those dominated by flagellates, cyanobacteria, and bacteria, in turn, resulting in a shift in composition of higher food webs (e.g., Legendre and Rassoulzadegan, 1995; Glibert, 1998; Mousseau, 2001; Heil et al., 2007). Until recently, most of the data illustrating such changes resulting from shifts in nutrient form were derived from systems for which N was the limiting nutrient. An important question in this context is whether the physiological and ecological consequences of dependence on NO_3^- versus NH_4^+ remain the same under nutrient rich conditions as under conditions of nutrient deficiency. A classic assumption is that when cells are growing at maximal growth rates (set by environmental conditions of light, temperature, etc.), it is fully expected that the total N taken up by cells will be the same whether they are provided NO_3^- or NH_4^+ (or urea or other forms of N). In some nutrient rich (but not classically

eutrophic) estuaries receiving high loads of NH_4 there is evidence for reduced, rather than increased, rates of primary productivity, compared to systems receiving comparable N loads in oxidized forms; such systems have been termed High Nutrient Low Growth (HNLG) systems (Yoshiyama and Sharp 2006).

Although nutrient effects have generally not been considered controlling factors in San Francisco Estuary, the more subtle ecological impacts of NH_4^+ loading and the importance of changes in $\text{NO}_3^-:\text{NH}_4^+$ in phytoplankton succession are beginning to be considered as important factors that may have contributed to historical changes seen in the food web (e.g., Dugdale et al., 2007; 2012, 2013; Glibert, 2010; 2012; Glibert et al., 2011; Parker et al., 2012a,b). Not only have dominant species changed in this system, but rates of primary production have also declined over the course of the past few decades (e.g., Jassby et al., 2002; Kimmerer et al., 2012). Without question, the Bay Delta is receiving high loads of NH_4 and these loads have increased over the recent decades (Fig. 1). A question receiving considerable attention is whether these concentrations of NH_4 have contributed to the tilting of the Bay-Delta away from a productive ecosystem to one that has HNLG characteristics. Based solely on previously reported ambient chl *a* in Suisun Bay values that tend to be $<5 \mu\text{g L}^{-1}$ on average (occasional blooms excepted), combined with ambient nutrient concentrations, this system is characteristic of an HNLG region (Yoshiyama and Sharp, 2006; Dugdale et al., 2007).

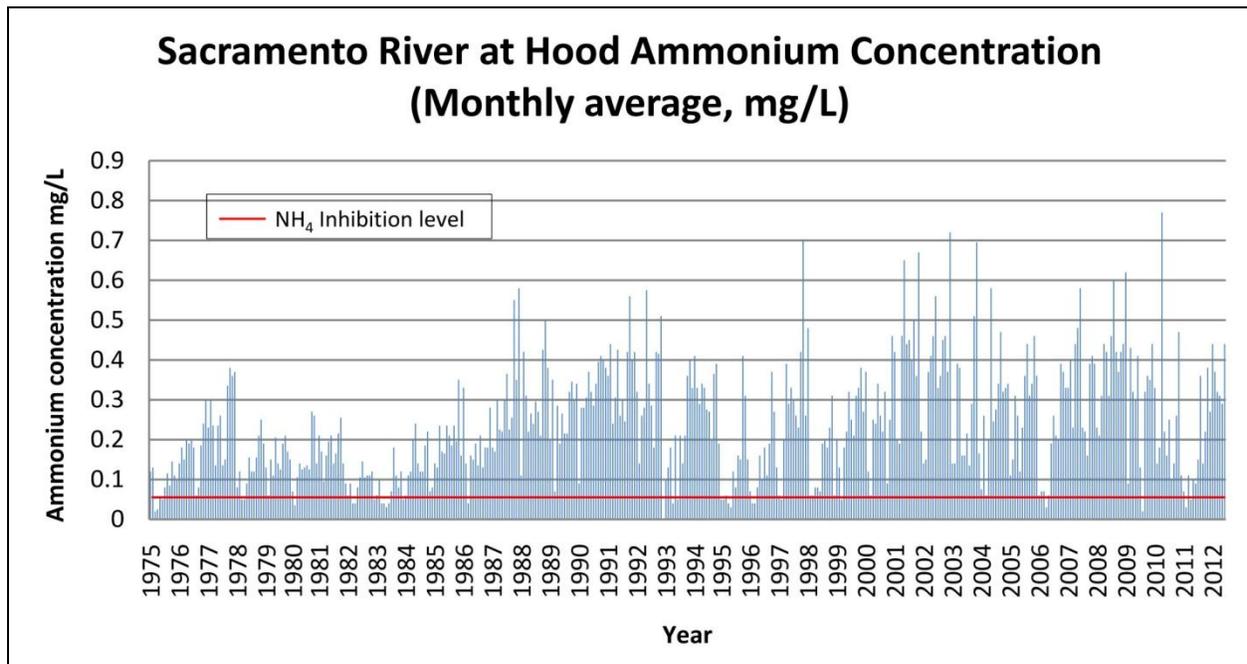


Figure 1. Average monthly ammonium concentration in the lower Sacramento River at Hood. Source: Environmental Monitoring Program data.

The fundamental mechanism contributing to HNLG systems is the inhibition of NO_3^- uptake by NH_4 . More correctly termed “repression” rather than “inhibition”, this phenomenon is well documented throughout the physiological literature. Both NH_4^+ and NO_3^- are transported across the cell membrane by both passive and active pathways. Active pathways involve transporters, specific proteins that transport the targeted molecules. For most algal functional groups, once transported into the cell, NO_3^- is first reduced to NO_2^- through the activity of Nitrate Reductase (*NR*) and subsequently further reduced to NH_4 , the form that can be assimilated into amino acids and proteins for cell growth. Both the process of NO_3^- transport into the cell and its subsequent reduction to NH_4 can be inhibited (or repressed) by NH_4 . Because of differences in the physiology of transporters and cell metabolism between different classes of

algae, different types of algae have different susceptibilities to NH_4^+ . Such differences contribute to the confusion about, different observations of, and different interpretations as to the importance of, NH_4 in regulating productivity of algae.

The inhibition or repression of NO_3^- uptake by NH_4^+ needs to be differentiated from the metabolic effect of toxicity of NH_4^+ or NH_3 . *NH_4^+ is, in fact, best characterized as a paradoxical nutrient – preferentially used at one end of the concentration spectrum when N is limiting and toxic to the cell when supplied at super-saturating levels* (Britto and Kronzucker 2002). Whereas NH_4^+ is transported across the cell membrane via active transport, the unionized form, NH_3 , can diffuse through membranes (Kleiner 1981). The presence of NH_3 increases with increased pH, a condition that is not the norm in aquatic systems, except under dense blooms. More about toxicity effects of NH_4 and NH_3 can be found in the Total Ammonia Toxicity section below.

NH_4 suppression of NO_3 uptake when both nutrients are in ample supply must also be differentiated from the preferential use of NH_4 by phytoplankton when N is limiting. Under the latter conditions, phytoplankton will use NH_4 preferentially because it requires less energy than NO_3 . Under the former conditions, the cells must cope with an excess; and in doing so, their metabolism is less capable of assimilating NO_3 .

The effects of changes in the proportion of $\text{NH}_4:\text{NO}_3$ have been shown for the Bay-Delta in both field observations and laboratory experiments. Parker et al. (2012a) observed a 60% decline in primary production in the Sacramento River below the Sacramento Regional Wastewater Treatment Plant, where NH_4 is discharged, compared to production above the Treatment Plant's outfall. Also supporting this finding, Parker et al. (2012b) found that “[b]y tracing both carbon (C) and N uptake we provide clear evidence that high rates of C uptake are linked to phytoplankton NO_3 , and not NH_4 , use.” They conclude that the increased proportion of NH_4 “may help explain some of the reduced primary production and phytoplankton biomass observed [in the San Francisco Estuary] since the 1970s.”

In enclosure experiments with samples from Central Bay, Suisun Bay, and the Sacramento River at Rio Vista, representing a gradient of both nutrient concentrations and proportions of different forms of N, Wilkerson et al. (in preparation) observed “a gradient of decreasing phytoplankton physiological rates in the upstream direction as far as Rio Vista.” Phytoplankton productivity rates (both carbon and nitrogen uptake) decreased with increasing concentrations of NH_4 .

In a series of recent experiments conducted with natural samples from the Sacramento River, the effects of altered proportions of $\text{NH}_4:\text{NO}_3$ were also apparent (Glibert et al. 2012b). In these experiments, the proportions of these nutrients were manipulated, and both short-term N uptake rates and longer-term N production rates were assessed. Two findings are of note. First, when 20 μmol NH_4 was added to the sample, a concentration commonly observed in the Sacramento River, and the rate of NO_3 uptake was measured across a concentration gradient, the rate of uptake of NO_3 decreased significantly compared to unamended rates measured over a period of < 1 hour (Figure 2). Second, when samples were enriched with NH_4 , NO_3 , or urea (at the molar equivalent dose) for a period of 24-48 hours, and then rates of uptake of all N forms measured, the summed rate of N uptake in the NH_4 -added treatment was significantly lower than that in the NO_3 -added or urea-added treatments (Figure 3).

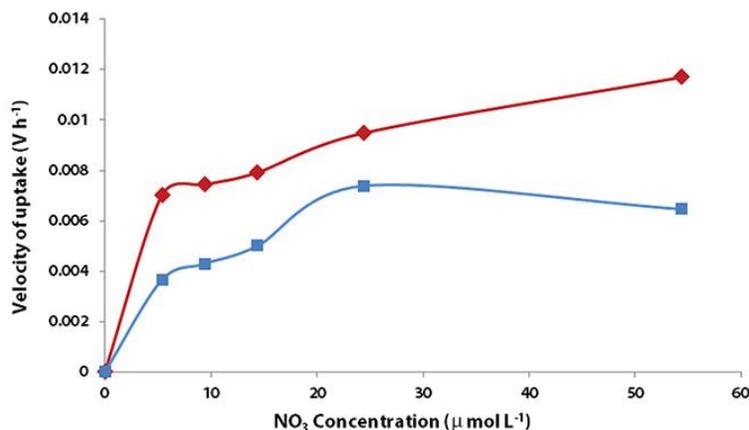


Figure 2. Velocity of uptake of NO₃ as a function of added NO₃ enrichment (red curve), and the same relationship but with a constant addition of 20 μmol L⁻¹ NH₄ (blue curve). Nitrate uptake is reduced when NH₄ is added. Experiment was conducted with water collected from the Sacramento River. Data from Glibert et al. (2012b).

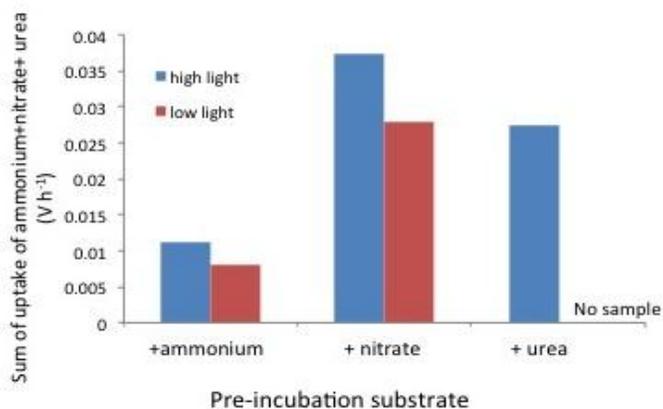


Figure 3. Summed rate of uptake of nitrate+ ammonium+ urea for samples collected from the Sacramento River and pre-incubated with the substrate indicated, after which short-term uptake rates were measured using ¹⁵N tracer techniques. The experiment was conducted under both high (blue bars) and low (red bars) irradiance levels. The summed rate of N uptake in the NH₄-added treatment was significantly lower than that in the NO₃-added or urea-added treatments. From Glibert et al. (2012b).

Longer term responses were also found in paired experiments in which NO₃ and NH₄ concentrations were manipulated for San Francisco Bay-Delta samples collected over 2 years, leading to different types of phytoplankton developing over a period of several days of exposure. A greater response by fucoxanthin-containing organisms (diatoms) was observed in those samples enriched with NO₃⁻, and greater responses by zeaxanthin and Chl *b*-containing organisms (predominantly cyanobacteria and chlorophytes) were observed in samples enriched with NH₄⁺ (Glibert et al., in review). A series of 3-week nutrient-rich mesocosm experiments conducted in Wascana Lake, Saskatchewan, Canada, yielded largely similar findings: total cyanobacterial biomass was associated with NH₄⁺ treatments and diatom biomass was associated with NO₃⁻ treatments (Donald et al., 2013). In recent experiments conducted in the tidal freshwater estuarine zone of the Guadiana Estuary (Spain and Portugal), it was also found that NO₃ consumption decreased with increasing NH₄ uptake, and these findings were most pronounced during the

most productive period. Total primary productivity was suppressed as a result (Domingues et al. 2011). As stated by Domingues et al. (2011), "...increased inputs of N as NH_4 due to urban waste effluents may result in a shift in phytoplankton community composition, towards a dominance of cyanobacteria and green algae."

This finding is further supported by the 37 years of data collected by the Environmental Monitoring Program in the Bay-Delta. When monthly data of chl-a and diatom cell count are plotted against NH_4 levels for the period 1975 to 2012 there is a marked decreasing trend in both as ammonium levels rise, with an increasing effect around the 0.056 mg L^{-1} ($4 \mu\text{mol L}^{-1}$) level (Figures 4 and 5). This level of NH_4 has been identified as a threshold value above which inhibition or repression of NO_3 uptake begins (Dugdale et al. 2007).

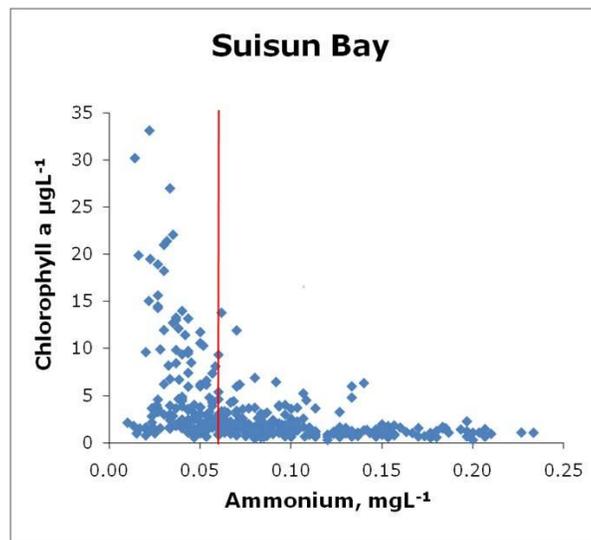


Figure 4. Chlorophyll-*a* concentration plotted with ammonium concentration in Suisun Bay. As NH_4 concentration increases above the level where inhibition is observed (red line at 0.056 mg L^{-1} NH_4), chl-*a* levels decline. (Source: Environmental Monitoring Program data)

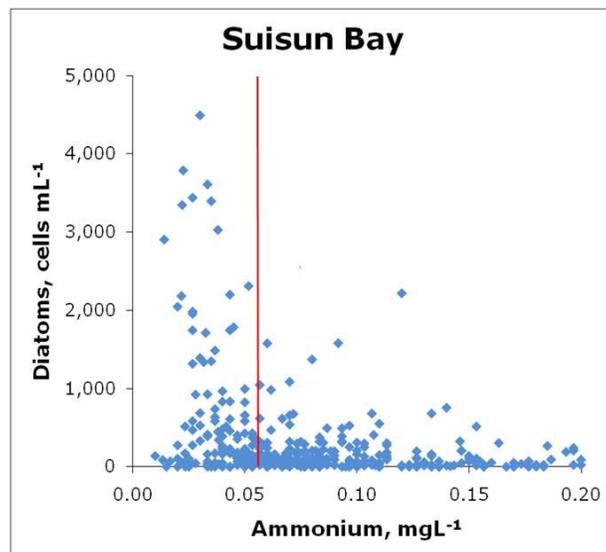


Figure 5. Diatom cell count plotted with ammonium concentration in Suisun Bay. As NH_4 concentration increases above the level where inhibition is observed (red line at 0.056 mg L^{-1} NH_4), diatom abundance declines. (Source: Environmental Monitoring Program data)

Similar threshold values have been reported by others. Lomas and Glibert (1999) described the threshold for inhibition of NO_3^- uptake at NH_4^+ levels of approximately $1 \mu\text{mol L}^{-1}$ (0.014 mg L^{-1}). Yoshiyama and Sharp (2006) saw a “striking decline in production at NH_4^+ levels above a low threshold (around $10 \mu\text{mol L}^{-1}$)” (0.14 mg L^{-1}).

The Delta’s algal species composition has shifted over the past decades from diatoms to smaller and lower quality food species such as flagellates, cryptophytes and cyanobacteria (Lehman, 2000; Lehman *et al.*, 2005; Lehman *et al.*, 2010; Jassby *et al.*, 2002; Sommer *et al.*, 2007; Glibert, 2010; Glibert *et al.*, 2011; Winder and Jassby, 2010). Dugdale *et al.* (2012) developed a conceptual model that correctly predicted the development of two rare, spring phytoplankton blooms in Suisun Bay based on only three criteria: the rate of NH_4^+ loading (based on present day sewage effluent loads), the water column concentration of NH_4^+ , and river flow (analogous in steady-state chemostat growth to “washout”). This conceptual model was further advanced in a one-dimensional, N-based model (Dugdale *et al.*, 2013), having the unique features of the inclusion of terms for the time-varying rates of maximum NO_3^- uptake as a function of NO_3^- concentration and for inhibition of NO_3^- uptake by NH_4^+ . The modeled high-biomass, NO_3^- -based, high-productivity state is analogous to the pre-1982, diatom era of the Bay-Delta. The importance of NH_4^+ inhibition of NO_3^- uptake was considered to be a necessary interaction to include in a recent model of the emergent phytoplankton community in the California Current System (Goebel *et al.* 2010).

The evidence is continuing to mount that the increasing loads of NH_4^+ are an important contributor to the observed changes in phytoplankton community. The emerging experimental data – assembled from samples collected from the Bay-Delta and elsewhere- provide direct experimental evidence that dichotomous phytoplankton communities develop when enriched with the same absolute concentration of NO_3^- and NH_4^+ , even when sufficient N nutrient was available to the community prior to the N inoculations. Although there is much yet to be learned about the role of nutrient control in many aspects of the Bay-Delta ecosystem, the patterns of response by algal communities to NH_4^+ vs. NO_3^- are consistent with observations in other systems and are consistent with the differential physiology of different algal groups.

Total Ammonia Toxicity

In addition to altering phytoplankton community structure, growth rates and abundance, ammonia is also toxic to some higher trophic level organisms. Scientists at University of California, Davis have investigated the effects of total ammonia nitrogen ($\text{NH}_3 + \text{NH}_4^+$) on the growth, reproduction and survival of the calanoid copepod *Pseudodiaptomus forbesi* using a full life-cycle bioassay approach (Teh *et al.* 2011). *P. forbesi* is an important prey item for the young of many fish species in the Bay-Delta including delta smelt and longfin smelt (Nobriga 2002; Hobbs *et al.* 2006; Feyrer *et al.* 2003). Teh *et al.* (2011) found that total $\text{NH}_3 + \text{NH}_4^+$ at 0.36 mg L^{-1} ($25.7 \mu\text{mol L}^{-1}$) significantly affects the recruitment of new adult copepods, and that total $\text{NH}_3 + \text{NH}_4^+$ at 0.38 mg L^{-1} ($27.1 \mu\text{mol L}^{-1}$) significantly affects the number of newborn nauplii surviving to 3 days (Teh *et al.* 2011). For comparison, monthly water samples collected between 2009-2010 from the Sacramento River between Hood and Isleton, approximately 30 miles downstream from the discharge point for the Sacramento Regional Wastewater Treatment Plant, exceeded the level of $0.36 \text{ mg/L NH}_4\text{-N}$ 44% of the time (Figure 6).

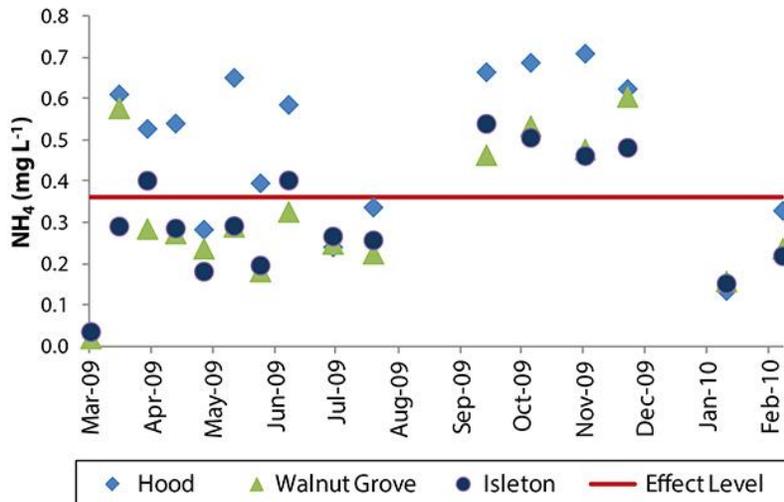


Figure 6. Ammonium concentrations in the Sacramento River measured at Hood, Walnut Grove, and Isleton between 2009 and 2010. Hood and Isleton are approximately 8 and 30 miles downstream of the discharge, respectively. The horizontal line at 0.36 mg L^{-1} is the level at which significant toxicity to copepods is observed. Data from Foe et al. (2010).

The Teh et al. (2011) study was recently cited in the U.S. Fish and Wildlife Service 12-month finding on the petition to list the Bay-Delta longfin smelt as threatened or endangered under the ESA. (77 Fed. Reg. 19755, 19776 [April 12, 2012].) The proposed rule states that “[a]quatic insects on which the longfin smelt relies upon for food have been shown to be sensitive to ammonia.” (77 Fed. Reg. 19776.) The proposed rule states that “[a]mmonia also can be toxic to several species of copepods important to larval and juvenile fishes.” (*Id.*) The toxic effect of total ammonia is a major stressor on aquatic life that has a pervasive impact across the Bay-Delta estuary.

Further, the U.S. Environmental Protection Agency’s (USEPA) Bay-Delta Action Plan highlights concerns with aquatic life toxicity caused by total ammonia nitrogen and identifies total ammonia levels as one of the suspected contributors to the pelagic organism decline in the Bay-Delta (USEPA 2012). In the Bay-Delta Action Plan, USEPA states that they will finalize the new national ammonia aquatic life criteria and they encourage the Regional Boards to consider adopting the criteria and using the criteria to develop effluent limitations for ammonia. USEPA adopted the new aquatic life water quality criteria for ammonia in 2013, which are more stringent than previous criteria adopted in 1999 and consider ammonia toxicity to freshwater mussels (USEPA 2013).

Nitrogen:Phosphorus

Extensive research has found that the N:P ratio also has profound effects on community structure. The principles of ecological stoichiometry suggests that different organisms will dominate under different relative proportions of critical elements (C, N, or P) due to differences in allocation of C, N and P in the various structures that form the biomass of different types of organisms (Sturner and Elser, 2002). As noted by Hall (2009, p. 504), “Ecological stoichiometry formalizes what should be obvious: Organisms interacting in food webs are composed of different elements, such as C, N, or P. As a result, energy and nutrient flow through consumer-resource interactions obey fundamental constraints.” Elemental differences in biomass are found at all levels of

organismal structure across trophic levels, from the subcellular to the macrocellular structural components (Sterner and Elser, 2002). The Redfield ratio (Redfield, 1934, 1958), in which organismal C:N:P ratios are assumed to be in the proportion of 106:16:1 by atoms, is likely the most well-known stoichiometric relationship.

The N:P ratio of nutrients has doubled in the Bay-Delta estuary over the last 35 years as is apparent from the data on Figure 7. These increases are a result not only of the increasing total N load (due to increasing effluent as well as other sources), but also as a function of declining P loads (Van Nieuwenhuysse 2007; Glibert 2010, Glibert et al. 2011).

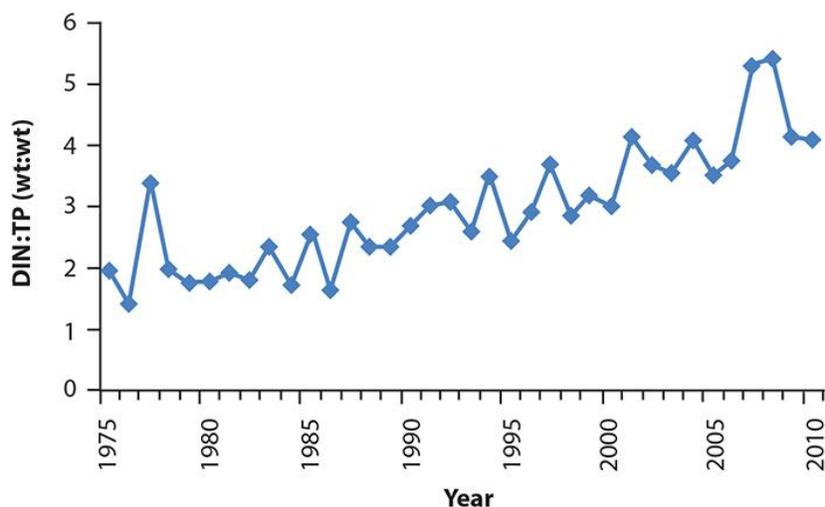


Figure 7. Average annual (March–November) ratio of dissolved inorganic nitrogen to total phosphorus has doubled in the Bay-Delta. (Source: IEP monitoring data from stations in Suisun Bay and the confluence (D4, D6, D7, D8)).

There are a number of strategies available to different types of phytoplankton for coping with an environment where nutrient ratios are not in proportion to their internal requirements (Glibert and Burkholder 2011). For example, cell size is an important determinant of elemental composition (Harris 1986; Finkel et al. 2010). Small cells have a lower requirement for P due to the smaller need for structural components in the cell (Finkel et al. 2010). In comparison to diatoms, very small cyanobacteria such as *Synechococcus* have a much larger cellular ratio of carbon to phosphorus (C:P), on average (Finkel et al. 2010). This explains why small cells, such as *Synechococcus*, have been found to thrive in waters that are comparatively P poor, as is the case in Florida Bay (Glibert et al. 2004).

These basic principles of algal response to changing N:P have relevance to San Francisco Bay-Delta phytoplankton communities. Jassby (2008) stated:

A decrease in percentage of diatom biovolume occurred during 1975–1989, caused by both a decrease in diatoms and an increase in green algae, cyanobacteria, and flagellate species biovolume (Kimmerer 2005; Lehman 1996), i.e., probably in the direction of declining nutritional value per unit biomass. In principle, the total nutritional value of a community could decrease even as its biomass increases. Moreover, changes in size, shape, and motility of species comprising the phytoplankton community could also affect their availability as food particles for crustacean zooplankton and other consumers.

There is strong support in the scientific literature for the proposition that the N:P ratio influences phytoplankton community composition. For example, in the Seto Inland Sea of Japan, removal of phosphorus also led to a shift in phytoplankton community structure from “nonharmful diatoms to harmful raphidophytes...and then finally to harmful/toxic dinoflagellates” (Yamamoto 2002). In this case the reduction in phosphorus which increased N:P, led to a change in phytoplankton community composition and was suggested to be “the major cause of the reduction in fishery production” (Yamamoto 2002).

In a retrospective analysis of 30 years of data from the Bay-Delta estuary, Glibert et al. (2011) found that the variation in these nutrient concentrations and ratios is highly correlated to variations in the total amount and composition of phytoplankton. This analysis revealed relationships between biological parameters and nutrients and/or nutrient ratios using both the original data and data that were adjusted for autocorrelation. At the phytoplankton level, as described earlier, there has been a decline in total chl-a and a decline in total diatoms over the past several decades in proportion to the increase in total inorganic N to total P (Figure 8). The change in chl-a with N:P is apparent in different regions of the Bay-Delta; as N:P increases, chl-a declines (Figure 9).

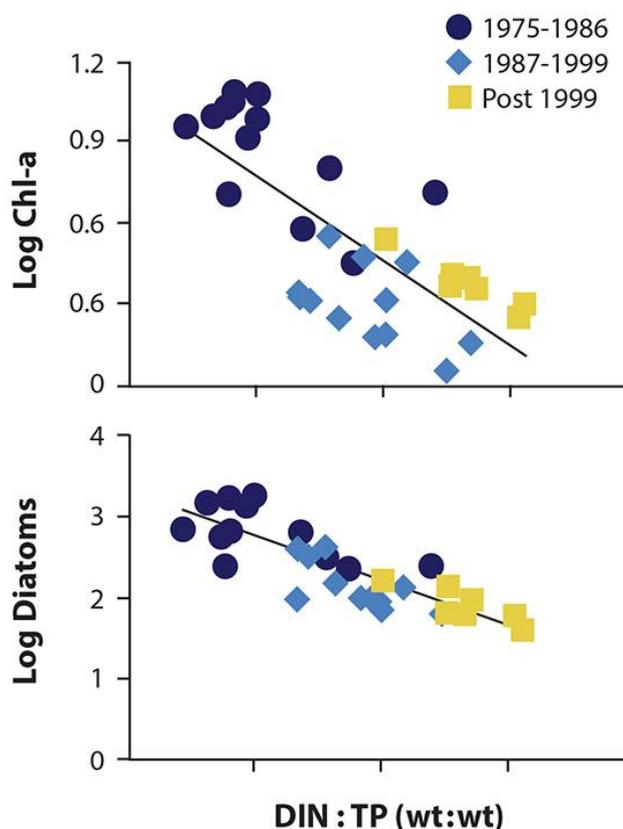


Figure 8. Change in the concentration of chl-*a* ($\mu\text{g L}^{-1}$) and abundance of diatoms (Bacillariophyceae, cells mL^{-1}) as a function of dissolved inorganic N to total phosphorus. A loss of total chl-*a* and a loss of total diatoms in the phytoplankton community have occurred over the past several decades in proportion to the change in total inorganic N to total P. The relationship is significant at $p < 0.05$. Different periods of time are represented by different symbols: 1975-1986, filled circles; 1987-1999, diamonds; post-1999- filled squares. Data shown are for the years 1975-2005 and cover the region from the confluence to Suisun Bay. All data log-transformed. Data from Glibert et al. (2011).

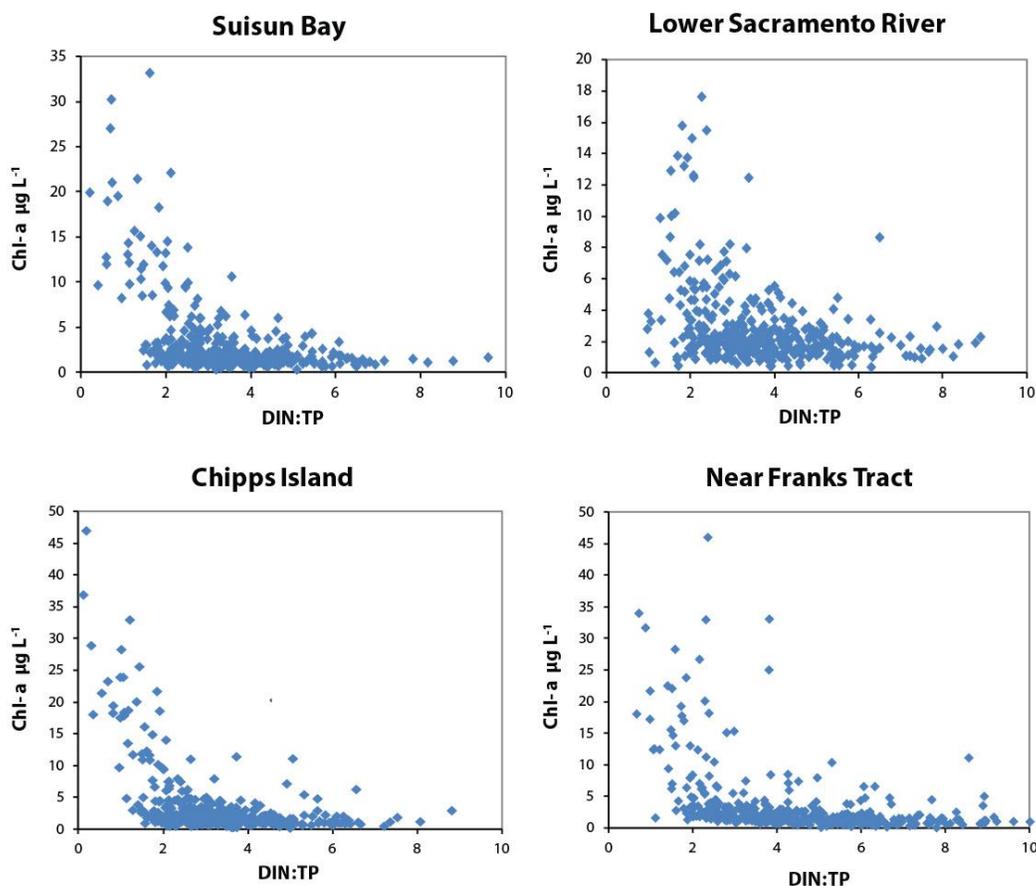


Figure 9. Chl-*a* concentration plotted against DIN:TP for subregions of the Delta, 1975-2011. As DIN:TP increases, there is a loss of chl-*a*. (Source: Environmental Monitoring Program data).

Fast-growing phytoplankton require proportionately more P to satisfy metabolic demands. Diatoms are typically fast-growing, and thus require proportionately more P to meet this metabolic demand. In ecological terms, they are considered a *r*-selected group, would be expected to have a low N:P biomass ratio (due to the high P cellular demand), and thus would be expected to be outcompeted if N:P in the environment increases. So-called *r*-selected species are out-competed when the environment changes (e.g., Heckey and Kilham 1988). In contrast, many cyanobacteria are considered to be *k*-selected, implying a slower growth rate and a higher metabolic N:P. In fact, “Reynolds (1984) singled out *Microcystis* as an example of a *k*-selected phytoplankter because it grows slowly in nature” (Heckey and Kilham 1988).

The balance of N:P can also affect other metabolic aspects of phytoplankton besides growth, including toxin production, cell membrane thickness, and other chemical constituents that have been considered to turn good food “bad” (Mitra and Flynn 2005). At the primary producer level, for example, emergent properties that can change in response to changes in elemental proportions include the relative proportions of ribosomes, enzyme activities, gene regulation, toxin production, cellular pigmentation complement, and ultimately the cell elemental composition, specifically, chl:carbon (Chl:C), C:N, and N:P (Glibert et al. 2013). Toxin production by numerous harmful algae has been shown to increase when the cells are grown under nutrient-imbalanced conditions and when there is a change in N or P availability (Flynn et al. 1994; Johansson and Granéli 1999; Granéli and Flynn 2006). In Daechung Reservoir, Korea, researchers found that toxicity of cyanobacteria was related not only to an increase in N in the water, but

to the cellular N content as well (Oh et al. 2000). A recent report by Van de Waal et al. (2009) demonstrated in chemostat experiments that under high carbon dioxide and high N conditions, microcystin (an algal toxin) production was enhanced in *Microcystis*. Similar relationships were reported for a field survey of the Hiro-sawa-no-ike fish pond in Kyoto, Japan, where the strongest correlations with microcystin were high concentrations of NO_3 and NH_4 and the seasonal peaks in *Microcystis* blooms were associated with extremely high N:P ratios (Ha et al. 2009). Thus, not only is *Microcystis* abundance enhanced under high N:P, but its toxicity appears to be as well (Oh et al. 2000).

It is well accepted that the nutritional value of phytoplankton differs from one species to another. Toxin production can inhibit grazing. Some phytoplankton species are rejected by grazers due to their size. Others vary in their nutritional quality. For example, some diatom species produce certain highly unsaturated fatty acids that are essential for zooplankton reproduction (reviewed by Kilham et al. 1997) while flagellates generally produce different fatty acids than diatoms (Olsen 1999). Many trophic interactions, such as rates of growth or fecundity, are dependent on the acquisition of particular fatty acids, as a measure of the food quality of algae (e.g., Ahlgren et al. 1990; Coutteau and Sorgeloos 1997; Weers and Gulati 1997; Brett and Müller-Navarra 1997). In feeding experiments, Ger et al. (2010) observed reduced survival of the copepods, *Pseudodiaptomus* and *Eurytemora*, even when *Microcystis* was only a small portion of their available diet. Brett and Müller-Navarra (1997) developed a food quality rank for 10 species from 5 major phytoplankton groups based on the average of the observed change in the abundance of individual zooplankters that preyed upon these phytoplankton in growth bioassays. They and others (see Park et al. 2003) have applied a 0-1 scale of phytoplankton food quality in which cyanobacteria ranks at 0.2; green algae, 0.525; diatoms, 0.7; and cryptomonads, 0.95. Thus, a trend of decreasing diatoms and increasing cyanobacteria in the Bay-Delta would suggest, based on these rankings, a decrease in food quality for higher trophic levels.

Cloern and Dufford (2005) state, “[t]he efficiency of energy transfer from phytoplankton to consumers and ultimate production at upper trophic levels vary with algal species composition: diatom-dominated marine upwelling systems sustain 50 times more fish biomass per unit of phytoplankton biomass than cyanobacteria-dominated lakes.”

For species that prey on phytoplankton (e.g., zooplankton), stoichiometry affects all aspects of behavior, such as growth rate, fecundity, and ultimately the success of different populations (Jeyasingh and Weider 2005, 2007), but may affect various life stages differently (Moe et al. 2005, p.31): “[a]n organism’s requirements for different elements may vary throughout its life cycle, and thus certain life stages may be more sensitive than others to variation in the stoichiometry of its resource.” For example, copepod juveniles have a relatively high demand for C, N, and P, but at a later stage, while C is still needed for metabolism, more P must be allocated to eggs. Therefore, P-poor food sources can disproportionately affect egg production while not affecting survival (Faerovig and Hessen 2003; Laspoumaderes et al. 2010). In a laboratory study where *Acartia tonsa* was fed diatoms grown on different N concentrations, Kiørboe (1989), confirmed that this zooplankter changes its feeding rate in response to phytoplankton of different chemical composition – thus, in response to food quality. Moreover, egg production followed the variation in algal N content and increased with increasing algal N. In the Bay-Delta, Slaughter and Kimmerer (2010) observed lower reproductive rates and lower growth rates of the copepod, *Acartia* sp. in the low salinity zone compared to taxa in other areas of the estuary. Their observation may be due to differences in nutrient stoichiometry between these areas.

In a review of field and laboratory-based research on stoichiometry in food webs, Hessen (1997) showed that a shift from copepods to *Daphnia* tracked N:P; copepods retain proportionately more N, while *Daphnia* are proportionately more P rich. Often, those organisms that are most able to retain the nutrient in limited supply, in this case P, have the competitive advantage in an unbalanced system. Glibert et al. (2011) illustrated a finding similar to Hessen’s, that the decline in calanoid copepods in the Bay-Delta,

and the invasion of cyclopoids tracked N:P over time. Variation in proportional densities of the calanoid copepod *Eurytemora* with the cyclopoid copepod *Limnoithona* over time has followed changes in the DIN:TP (dissolved inorganic nitrogen to total phosphorus) ratio (Figure 10), a pattern consistent with these grazers being responsive to changes in elemental stoichiometry and maintenance of altered dynamic equilibria on a long-term scale. In fact, Glibert et al. (2011) found relationships between many zooplankton species and nutrient composition. Results from whole-lake experimentation suggest that the N:P ratio is linked to alterations in zooplankton size, composition, and growth rate, as those animals with increased RNA allocation (more P available for growth) will grow at higher rates due to increased protein synthesis rates (Sterner and Elser 2002, Schindler 1974). Similar findings were reported from annual studies in the Baltic Sea (Walve and Larsson 1999).

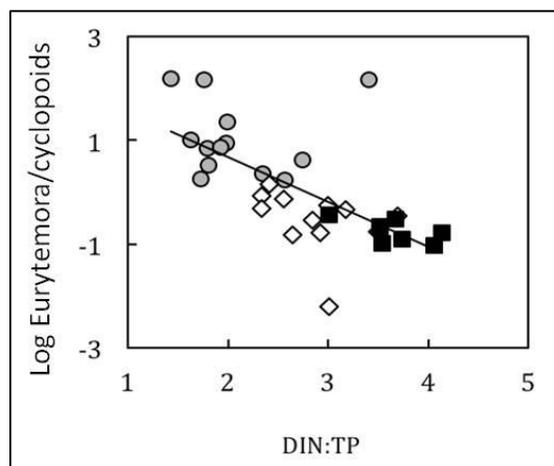


Figure 10. Change in the ratio of *Eurytemora* to cyclopoid copepods (all data log transformed) as a function of DIN:TP for annually averaged data from 1975-2005 for samples collected between the confluence and Suisun Bay. As DIN:TP increases, the proportion of *Eurytemora* to Cyclopoids decreases. Different periods of time are represented by different symbols: 1975-1986, filled circles; 1987-1999, diamonds; post-1999 filled squares. The correlation for these and for data that were detrended (not shown) are significant ($p < 0.05$). From Glibert et al. 2011.

Superimposed on these empirical observations is consideration of whether substrate quality or food quality is altered on an episodic basis, or whether changes are long-term and sustained. As conceptualized by Hood and Sterner (2010), a change in predator growth rate depends on the extent to which a diet is sustained or switches between low-quality food and high-quality food as defined by the relative P content.

Higher Trophic Level Effects

Disproportionate N and P loads are now recognized to have effects at all scales, from genomic to ecosystems that need further empirical resolution (Peñuelas et al. 2012). When N:P availability changes, food webs change, biogeochemical cycling can change, and these changes can be positively reinforcing. Sterner and Elser (2002) state: "[s]toichiometry can either constrain trophic cascades by diminishing the chances of success of key species, or **be a critical aspect of spectacular trophic cascades with large shifts in primary producer species and major shifts in ecosystem nutrient cycling**" [emphasis added]. Just as different elemental ratios may affect the composition of the primary producers, different nutrient requirements of organisms occupying higher trophic levels will have an impact on their ability to thrive as community composition changes at the base of the food web. At the ecosystem scale, the total load and balance of nutrient elements have effects that propagate through the food web, with the potential of transforming ecosystems to new stable states. Although the shift in algal community composition in terms of diatoms and cyanobacteria has been emphasized above, this shift in the Bay-Delta estuary has been far

more complicated. With the decline in water column chl-*a* and an increase in light availability, other primary producers have increased in abundance, including invasive macrophytes such as *Egeria densa* (Sommer et al. 2007; Nobriga et al. 2005; Glibert et al. 2011). *E. densa* may be particularly well suited to the low DIP:DIN environment of the Bay-Delta since it is able to access sediment bound phosphorus through its roots. In fact, similar increases in macrophytes were observed in many other systems in which N:P increased following N enrichment and P reduction, including the Potomac River, Chesapeake Bay, Ebro River in Spain, and the Rhine River in Germany (Glibert et al. 2011; Glibert 2012). Such macrophyte invasions can have profound impacts on ecosystems, not only because they alter the flow of C and the overall productivity of the system, but they also serve as “ecological engineers,” decreasing nutrients through uptake, reducing turbidity by trapping sediments, and providing refuge for zooplankton and habitat for other species, including fish (Yarrow et al. 2009; Glibert 2012).

The interplay between nutrient stoichiometry and biogeochemistry is well illustrated when a system is driven to higher macrophyte productivity. Macrophytes can be highly productive, which can result in elevation of pH due to carbon drawdown in the process of photosynthesis. As noted by Glibert (2012), once pH is elevated, the fundamental physical–chemical relationships related to P adsorption–desorption in sediments change, as does N biogeochemistry (Jordan et al. 2008; Gao et al. 2012). Moreover, under increased pH conditions, the biogeochemistry of calcification is altered, increasing the potential for calcification and the growth of calcifying organisms. Thus, the change in the abundance of the clam *Potamocorbula amurensis* from the time of its introduction in the mid-1980s to 2005 has been shown to be highly and positively correlated to the increase in total N:total P ($r^2 = 0.46$; $n = 20$; $p < 0.01$; all data log transformed), and the average annual abundance of this species has also been found to be highly and positively correlated with mean annual average pH in the estuary ($r^2 = 0.64$; $n = 19$; $p < 0.01$; species abundance data log transformed) (Glibert et al. 2011). Interestingly, the Potomac River, Rhine River and the Ebro River have had similar invasions of macrophytes and *Corbicula* clams that relate to increases in N:P loading (Ibanez et al. 2008; Glibert et al. 2011; Glibert 2012).

In the Bay-Delta estuary, data show top-down grazing of phytoplankton by the clam *P. amurensis* exerts a strong control on phytoplankton biomass, as is also the case for other systems when invaded by bivalve mollusks. Prior interpretations, emphasizing stochastic invasions largely via ballast water exchange imply that the invasive event was the ultimate cause of the change in top-down control of phytoplankton. The ecological stoichiometric interpretation does not preclude strong top-down control of selected component organisms, nor ballast water exchange as the mechanism of introduction. The distinction is that, at the overall ecosystem level, the structuring of species is affected by alterations in nutrients and ecosystem biogeochemistry.

The arguments presented here make the case that bottom-up control contributed to the conditions that allowed *P. amurensis* to become a dominant regulator of phytoplankton production. In other words, invasive species effects and nutrient effects are interrelated. This interpretation is consistent with Ware and Thompson’s (2005) insights from a broad survey of the relative contributions of “bottom-up” vs. “top-down” factors that potentially control fish catch in the coastal waters of the western U.S.; they, too, reported that bottom-up factors were more important.

Several recent reviews have investigated the stoichiometry of fish (Sterner and George 2000; Hendrixson et al. 2007; McIntyre and Flecker 2010). Not only does a strong shift in body N:P occur with growth stage (Pilati and Vanni 2007), but strong differences between taxonomic families also occur. In fact, Hendrixson et al. (2007) demonstrated, for 20 families of fish, that a phylogenetic tree could be developed based on the body nutrient composition.

In the Bay-Delta estuary, numerous changes in fish community composition occurred in relation to phytoplankton and zooplankton changes, and to N:P (Glibert 2010; Glibert et al. 2011) (Figure 11).

Glibert et al. (2011) also found that total P “explained at least as much of the variability in delta smelt as did the [Feyrer et al. 2011] habitat index, and dinoflagellate abundance explained even more.” Unlike correlations with Bay-Delta outflow or with the location of the 2 practical salinity unit isohaline, where the underlying mechanisms driving the correlations are largely unknown, the nutrient relationships have a strong mechanistic explanation in ecological stoichiometry and stable state principles. For this reason, there is relatively low uncertainty that changes in nutrient stoichiometry in the Bay-Delta estuary, achieved through both external forces (altered land-based nutrient loads) and internal, organism-driven, assimilative and dissimilative processes, are related to community compositional changes (Glibert et al. 2011; Glibert 2012).

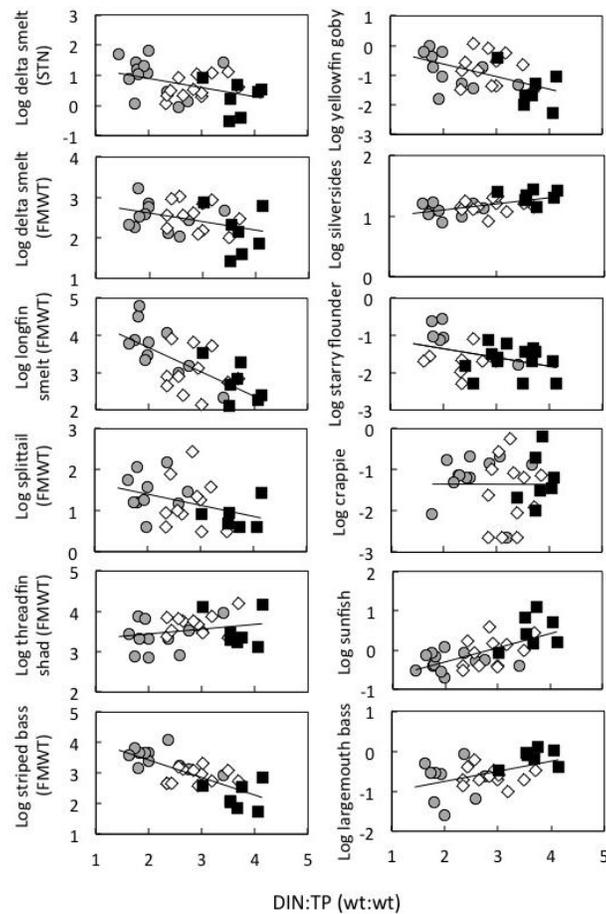


Figure 11. Changes in the abundance of major fishes in relation to ratio of dissolved inorganic nitrogen to total phosphorus from 1975-2005. Different periods of time are represented by different symbols: 1975-1986, filled circles; 1987-1999, diamonds; post-1999 filled squares. All data were log-transformed. The correlations for all fish except crappie were significant ($p < 0.05$) in these data as well as in data that were detrended. Source: Glibert et al. (2011).

Total Nutrient Loads

Total nutrient load sets the upper limit on total primary production, and increases in nutrient loading are commonly associated with eutrophication. Nutrient levels in water diverted from the Delta are at concentrations that can produce nuisance algal and aquatic weed growth and adversely affect drinking water beneficial uses in downstream conveyance facilities and reservoirs. Algal and aquatic plant growth in the SWP conveyance facilities and downstream reservoirs is neither light limited nor inhibited by high

ammonium concentrations since most of the ammonium from the Delta has been nitrified to nitrate. Elevated levels of nutrients (phosphorus and nitrogen compounds) stimulate nuisance algal and aquatic weed growth that includes production, by specific cyanobacteria, of noxious taste and odor compounds and algal toxins. In addition to algal produced taste and odor and algal toxin concerns, increases in algal and aquatic weed biomass can impede flow in conveyances, shorten filter run times and increase solids production at drinking water treatment plants, and add to organic carbon loading.

Frequently annual phosphorus concentrations at Clifton Court Forebay have averaged 0.11 mg/L and total nitrogen has averaged 0.87 mg L⁻¹. Phosphorus is significantly higher than the 0.020 to 0.042 mg L⁻¹ that has been associated with a high risk of nuisance growth and eutrophication (USEPA 1980, 2001a). Levels of both nutrients exceed USEPA Ecoregion I phosphorus and total nitrogen reference conditions of 0.047 mg L⁻¹ and 0.31 mg L⁻¹, respectively (USEPA 2001b)². Ecoregion I includes the Central Valley.

Literature values and USEPA's ecoregion reference conditions³ provide a starting point for determining whether nutrient concentrations in Delta waters are at levels that could cause water quality impairments, such as algal production of compounds that produce noxious tastes and odors. More importantly, there is already significant evidence of nutrient-related adverse impacts from Delta water. Through 2006, the Department of Water Resources (DWR) has applied algaecide treatments to Clifton Court Forebay for aquatic weeds and algae multiple times each summer. This practice was halted in 2007, however, over concerns of potential impacts to listed fish species. DWR has also treated the South Bay Aqueduct (SBA) to control algae that are stimulated by nutrient-rich Delta water. Including preventative treatments, DWR has treated the SBA for algal control between 10 and 16 times per year in recent years. Periodic treatment of the California Aqueduct and State Water Project (SWP) terminal reservoirs is also necessary for the same reason. Given the increasing environmental concerns about the use of copper-based algaecides, it is likely that effective control will become increasingly more difficult and reduce the ability of downstream users to manage algae-related problems in the future.

Eutrophication, ecological stoichiometry and alternate stable state theories combine to serve as a unifying framework for understanding the complexity of responses not only in the Bay-Delta estuary but also, more generally, in many comparative systems. This interpretation does not negate the importance of ecological invasions, habitat changes, multiple stressors and food-web complexities, but adds an explanatory mechanism to those interpretations through biogeochemistry and organismal stoichiometry. Ecological stoichiometry affects systems by setting elemental constraints on the growth of organisms. This, in turn, affects food quality and the relationships between predators and prey.

Examples of Responses to Nutrient Load Reductions

A growing body of literature documents improvements in ecosystem functions where nutrient loading is reduced and stoichiometric balance is restored. Reducing nutrient loading in the Chesapeake Bay, Tampa Bay, and coastal areas of Denmark has proven to be effective at reversing the harmful effects of previously undertreated discharges and restoring the native food webs. For example, within several years of increasing nutrient removal at the Blue Plains treatment plant in Washington D.C., N:P ratios in the Potomac River declined, the abundance of the invasive *Hydrilla verticillata* and *Corbicula fluminea* began to decline (Figure 12 showing *Corbicula Fluminea* and other relationships with N:P), and the abundance of native grasses increased (Ruhl and Rybicki 2010).

² The reference condition is the 25th percentile of the nutrient data for sites within the ecoregion and is meant to represent the nutrient concentrations in minimally impacted water bodies.

³ Significant questions have been raised about the use of reference conditions to establish regulatory criteria for nutrients. Nevertheless, they provide a starting point for evaluating water quality.

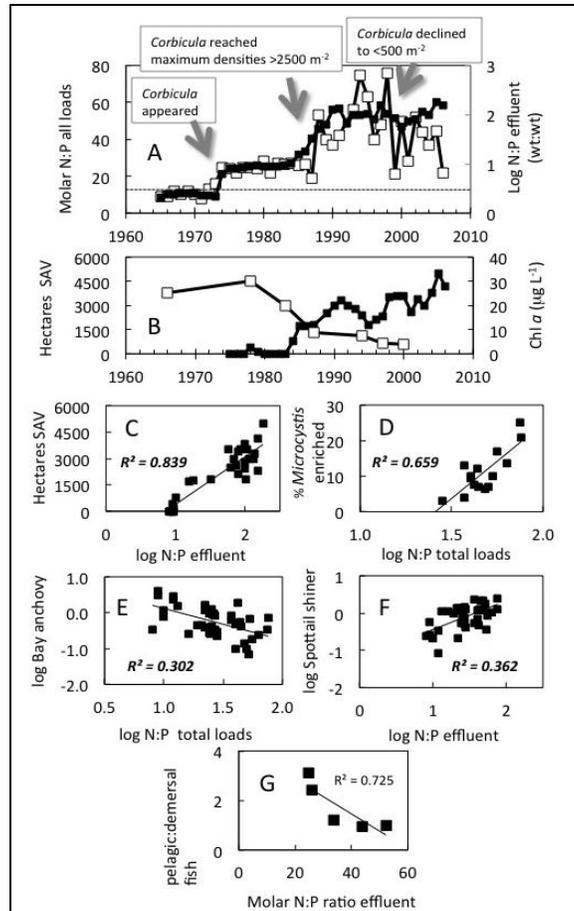


Figure 12. Comparative relationships for the Potomac River. Panel A shows the change in effluent N loading and the relative abundance of the invasive clam, *Corbicula fluminea*. *C. fluminea* appeared coincident with a sharp increase in N:P and increased in abundance as N:P increased. When N:P decreased sharply around 1999, *C. fluminea* abundance also declined sharply from >2500 m⁻² to <500 m⁻². Data derived from Dresler and Cory (1980), Jaworski et al. (2007), and Cummins et al. (2010). Figure reproduced from Glibert et al. (2011).

Tampa Bay provides another important example. Eutrophication problems in Tampa Bay were severe in the 1970s, with N loads approximating 24 tons per day, about half of which was due to point source effluent (Greening and Janicki 2006). Several years after nitrogen and phosphorus reductions were achieved, native seagrass began to increase. Lower nutrient discharges also had positive effects on the coastal waters around the island of Funen, Denmark (Rask et al. 1999). Since the mid 1980s, there has been a roughly 50% reduction in the loading of N and P in the region due to point source reductions. Again, native grasses returned and low oxygen problems were reversed.

Cloern (2001) provides additional examples of recovery following reductions in nutrient and waste inputs. Citing other researchers, Cloern (2001) shows improvements in dissolved oxygen levels in the Forth Estuary in Scotland following improvements in wastewater treatment. Citing a second study, Cloern (2001) shows increases in fish diversity in the Thames Estuary following improvements in wastewater treatment there (Figure 13).

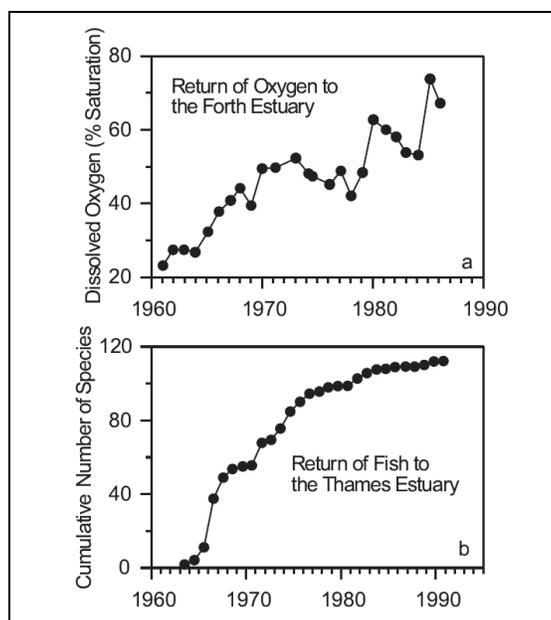


Figure 13. Two examples of recovery following actions to restore water quality in estuaries impacted by nutrient and waste inputs: (a) trend of increasing dissolved oxygen concentration (summer months) in the Forth Estuary, Scotland, following Improvements in wastewater treatment; (b) trend of increasing diversity of fishes in the Thames Estuary following implementation of advanced wastewater treatment and increases in oxygen concentrations (Source: Figure 20 from Cloern 2001).

Moreover, there is recent evidence that diatom blooms may be restored in the Bay-Delta estuary if NH_4 loading is reduced. In Suisun Bay, an unusual diatom bloom in Spring 2000 reached chl-*a* concentrations of $30 \mu\text{g L}^{-1}$ when NH_4 concentrations declined to $1.9 \mu\text{mol L}^{-1}$ (0.027 mg L^{-1}) (Wilkerson et al. 2006). Similarly, chl-*a* concentrations in Suisun Bay reached $35 \mu\text{g L}^{-1}$ during spring 2010 when NH_4 concentrations declined to $0.5 \mu\text{mol L}^{-1}$ (0.007 mg L^{-1}) (Dugdale et al. 2011, 2012). These blooms are comparable to spring chl-*a* levels from 1969 to 1977 (Ball and Arthur 1979) when NH_4 concentrations were $1.8 \mu\text{mol L}^{-1}$ (0.025 mg L^{-1}) during summer and $4.0 \mu\text{mol L}^{-1}$ (0.056 mg L^{-1}) during winter (Cloern and Cheng 1981).

In sum, ecological stoichiometry affects ecosystems by setting elemental constraints on the growth of organisms. However, as is the case with all conceptual frameworks, it must be viewed within the context of other factors in the environment, including the multiple stressors that now impact systems, as well as the scale (spatial, temporal and organismal) on which the system is being examined. Imbalances in stoichiometry may have impacts on ecosystems even at nutrient loads normally taken to be saturating or supersaturating. Ecological stoichiometry has several important implications for the health and sustainability of aquatic systems. Stoichiometric imbalances may accelerate transformations of nutrients or may alter the processes by which nutrients are cycled in the ecosystem, further altering nutrient availability or form for primary producers (Elser and Hamilton 2007, Nugraha et al. 2010). When food *quality* is linked to food web outcome, feedback effects and nutrient biogeochemical processes may play large roles in species success (Glibert 2012).

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