Biogeochemical processes in a small California estuary. 1. Benthic fluxes and pore water constituents reflect high nutrient freshwater inputs

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ABSTRACT: Elkhorn Slough, a small estuary in central California, receives nutrient inputs from agricultural and other non-point source runoff. To evaluate the effect of nutrient loading on ecosystem processes, rates of sediment oxygen consumption, benthic nutrient fluxes, pore water ammonium (NH_4^+), dissolved inorganic phosphate (DIP) and sulfide (S^{2-}) concentrations were measured at 5 sites several times each year between 1998 and 1999. Two sites near the head of the slough receive direct runoff from agricultural fields, while the sites in the middle reaches and mouth receive direct runoff from grasslands and woodlands. Sites receiving agricultural runoff usually had high water column concentrations of dissolved inorganic nitrogen and chlorophyll *a* (50 µM and 30 µg l^{-1}, respectively). High precipitation in the winter of 1998 caused low salinities in the slough at that time. In general, salinity and nutrient concentrations were inversely correlated. Sites receiving agricultural runoff had high pore water NH_4^+, DIP and dissolved S^{2-} concentrations, particularly in the summer. Oxygen penetration into sediments was the greatest at the sites adjacent to grasslands and woodlands, and least at the sites receiving high nutrient runoff. Sediment oxygen consumption was 4 times greater at sites near agricultural fields than the site near the mouth, while NH_4^+ flux out of sediments was between 4 and 20 times greater at agricultural sites than at non-agricultural sites. This study suggests that high nutrient runoff has primarily a local rather than estuary-wide effect on sediment biogeochemical processes in Elkhorn Slough, perhaps due to longer water residence time at the head of the slough compared to the mouth.

KEY WORDS: Estuary · Eutrophication · Nutrients · Oxygen · Elkhorn Slough

INTRODUCTION

Nitrogen (N), phosphorus (P) and silica (Si) are essential elements controlling primary production in estuarine environments (Ryther & Dunstan 1971, Howarth 1988, Conley & Malone 1992). These cycles are interrelated by organic matter production and decomposition. In shallow estuarine environments, most organic matter produced in the water column is decomposed and regenerated in sediments (Graf 1992, Kemp et al. 1992, Jørgensen 1996a). Within sediments, cycling of oxygen (O), carbon (C), N, P and sulfur (S) are intertwined as decomposition proceeds. Two electron acceptors that dominate organic C decomposition in sediments are O_2 and SO_4^{2-} (Mackin & Swider 1989). At the sediment-water interface, O_2 diffuses into sediments from the water column and aerobic processes such as aerobic respiration (estimated as sediment O_2 consumption, SOC) dominate. Once O_2 is

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depleted, other electron acceptors, NO_3^-, Mn^{3+}, Fe^{3+}, SO_4^{2-} and CO_2, are used sequentially by heterotrophic bacteria. Decomposition by SO_4^{2-} reducing bacteria produces S^{2-} and NH_4^+. NH_4^+ produced in sediments can diffuse out into the overlying water or be oxidized to NO_2^- and NO_3^- by nitrifying bacteria, which are obligate aerobes. Any NO_2^- or NO_3^- present in sediments can also diffuse out, or be reduced back to NH_4^+ by dissimilatory NO_3^- reduction or reduced to N_2 by the process of denitrification. Although P does not undergo oxidation-reduction reactions like N, it binds to FEOOH under aerobic conditions. When Fe^{3+} is reduced, P becomes more available and its concentration increases in pore waters. In contrast to the other elements described, remineralization of Si is a chemical process, not mediated by bacteria or oxidation-reduction reactions.

In addition to the availability of electron acceptors, rates of decomposition are dependent on the quantity and quality of organic matter in sediments. As organic matter production in the water column increases or as water depth decreases, decomposition in sediments increases (Kemp et al. 1992, Jørgensen 1996a). Decomposition rates are also dependent on the lability of organic matter, with refractory organic matter decomposing more slowly than labile organic matter (Westrich & Berner 1984).

Eutrophication is an increasing problem within estuaries and the coastal zone because of increasing nutrient loads to these systems (Nixon 1995, Bricker et al. 1999, NRC 2000). Runoff from agricultural fields has long been recognized as a significant source of nutrients to many estuaries including Chesapeake Bay (Staver et al. 1996, Jordan et al. 1997), Neuse River (Paerl et al. 1995), North San Francisco Bay (Hager & Schemel 1992), Roskilde Fjord (Kamp-Nielsen 1992), Scheldt estuary (Billen et al. 1985) and several Scottish estuaries (Balls 1994). The effect of agricultural runoff or any high nutrient runoff on estuaries can be modulated by the water residence time, stratification and other physical transport processes. Estuaries with low flushing rates may be more susceptible to eutrophication than well-flushed systems (NRC 2000, Cloern 2001). A possible mechanism is that as residence time increases, nitrogen export decreases (Nixon et al. 1996), leading to increased phytoplankton or macroalgal production which is then deposited to sediments. The effect of increased organic loading to sediments can be significant particularly as it affects the oxicanoxic interface and thus the balance between aerobic processes (e.g. SOC and nitrification) and anaerobic processes (e.g. sulfate reduction and denitrification) (Kemp et al. 1990, Caffrey et al. 1993, Jorgensen 1996a, Herbert 1999).

Interactions among nutrient loading, residence time and sediment/water fluxes were investigated in the present study in Elkhorn Slough, a small estuary, extending inland for 11.4 km from Monterey Bay in central California. The slough includes a variety of habitats that experience a range of freshwater/seawater exchanges and nutrient loading. Local environments in the slough, such as pocket marshes or tidal creeks, are important to wildlife and human usage of the site, and the characteristics of the main stem of the estuary are probably not representative of the great spatial and temporal variability experienced by other slough habitats. The slough is shallow (2.5 m water depth), with intertidal mudflats and pickleweed marsh *Salicornia virginica* bordering the main channel and tidal creeks. Extensive mats of macroalgae are found on the intertidal mudflats, while eelgrass *Zostera marina* colonizes a shallow subtidal zone near the mouth. Several pocket marshes and shallow ponds occur in Elkhorn Slough. These areas are isolated from the main channel by the Southern Pacific Railroad tracks. The tide range along the main channel in Elkhorn Slough is 1.7 m.

Agricultural land-use occurs on about 26% of the watershed and is dominated by row crops such as strawberries, flowers, artichokes and raspberries (Fig. 1). Most of the land is zoned rural residential, with a majority of the homes using septic systems. Urban and commercial land uses represent about 16% of the land cover and are concentrated in the towns of Moss Landing and Castroville. About 10% of the watershed is protected and managed in its natural state by a variety of state and federal agencies, or private non-profit organizations.

In this study, we examined seasonal changes in overlying water nutrient and chlorophyll *a* (chl *a*), and pore
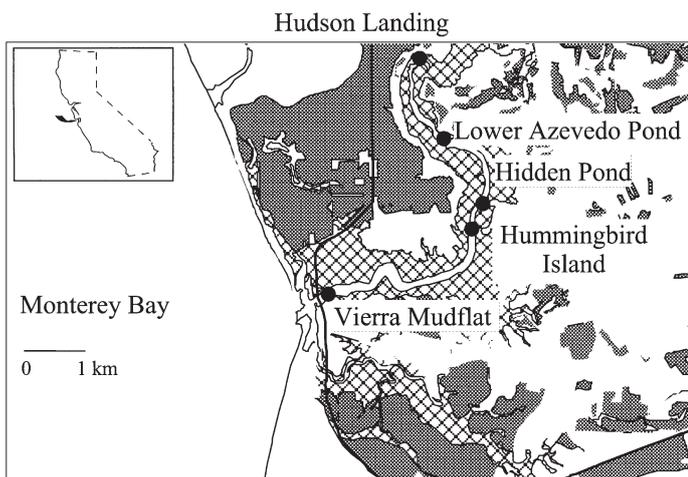

Fig. 1. Map of Elkhorn Slough, central California, showing station locations, marsh (crosshatch), and agricultural lands (stippled)

water NH_4^+, dissolved inorganic phosphate (DIP) and S^{2-} in 5 sites throughout Elkhorn Slough. Sites were chosen on the basis of their proximity to various land uses in the region. These sites received runoff either from agriculture and other non-point sources, or from grasslands and woodlands. These sites also spanned a gradient from poorly to well flushed water exchange. Benthic fluxes of O, N, P and Si were compared at 3 of the 5 sites.

MATERIALS AND METHODS

Site description. Five sites were chosen to represent a range of environments in the slough from the main stem to isolated pocket marsh ponds. Water and sediment samples were collected between February 1998 and November 1999 (Fig. 1). Sampling was usually conducted at low tide. During the first year of this study, 3 sites were sampled monthly between February and November. Two of the sites (Hudson's Landing and Lower Azevedo Pond) are adjacent to agricultural fields and directly receive non-point source runoff, while the third site (Hidden Pond), is surrounded by grasslands. Two additional sites were added in 1999 in order to include some well-flushed sites representative of the central slough (Hummingbird Island) and the mouth of the estuary (Vierra mudflat). Hudson's Landing, Vierra mudflat and Hummingbird Island were sampled in February, March, July, and November 1999. Hidden and Lower Azevedo Ponds were also sampled in July 1999. All sites were characterized by fine-grained, muddy sediments except for Vierra mudflat, which was muddy sand.

Runoff and tidal flushing were quite different at each site. Hudson's Landing is adjacent to a pickleweed marsh at the head of Elkhorn Slough. It is exposed at low tides (+1.0 m mean low low water [MLLW]). This site receives runoff from agricultural fields (strawberries, flowers and truck crops), a cattle pasture, golf course and the unincorporated town of Los Lomas. Lower Azevedo Pond (referred to in the text as Azevedo Pond) is a tidal pond (4300 m^2), bounded on one side by railroad tracks and on the other side by strawberry fields extending to the upper edge of the pickleweed marsh. This site is poorly flushed with the only water exchange between the pond and the main channel of Elkhorn Slough occurring via seepage through a blocked culvert. Hidden Pond is located on the Elkhorn Slough National Estuarine Research Reserve (ESNERR). It is a small tidal pond (2600 m^2) bounded by railroad tracks and pickleweed marsh and surrounded by grasslands. A culvert allows for moderate flushing with the main channel of Elkhorn Slough. Both ponds generally had 5 to 30 cm of overlying water

except during the lowest tides. The Hummingbird Island site is also located on the ESNERR near Hidden Pond and receives runoff from nearby grasslands. This site is a mudflat adjacent to the main channel and is exposed during low tides (+0.45 m MLLW). Vierra mudflat is located near the mouth of the slough and is extremely well flushed. Sediments are exposed at low tides (+0.2 m MLLW).

Sampling. Water samples were collected for chl *a* and nutrient analysis in plastic bottles, and returned to the lab on ice, where the samples were filtered. The chlorophyll filters and nutrient samples were frozen until analysis. Sediment cores (several types, see below for descriptions) were collected at each site for pore water nutrient analysis, benthic fluxes, percent water and percent organic matter. In July 1999, cores were collected for measurement of depth of O_2 penetration using O_2 microelectrodes. Temperature and salinity were measured at all sites at the time of sampling.

Benthic flux cores: Triplicate cores (6 cm diameter) were collected at each site along with approximately 10 l of water. Only 2 sites were sampled on each trip, due to the labor and time commitment required for flux measurements. Water height in the cores was 25 to 35 cm, while sediment thickness was 15 to 20 cm. The cores were brought back to the lab in the dark in coolers filled with ambient water from the sites to maintain temperature. Cores were gently aerated by bubbling to maintain ambient conditions, near saturation, and held in a controlled temperature room for 20 h before conducting experiments. The bubbling was stopped, overlying water was replaced with the collected water and cores were sealed with tops equipped with a magnetic stirring system and sampling ports. Cores were incubated in the dark at ambient temperatures. The first water sample (30 ml) was taken after a 1 h incubation period and 3 additional samples were removed at subsequent 1 h intervals. Water removed from the cores was replaced with water from the sampling site. Water samples were filtered through GF/F filters for nutrient — NH_4^+, $NO_3^- + NO_2^-$, NO_2^-, DSi (dissolved silicate) and DIP — analysis, or placed unfiltered in glass bottles (7 ml) with Winkler reagents for O_2 analysis.

Pore water constituents: Triplicate sediment cores (4.6 cm diameter) were collected at each site for analysis of pore water constituents. Within 4 h of collection, sediments were sliced into 0 to 0.5, 0.5 to 1, 1 to 2 and 2 to 4 cm sections in a N_2 filled glove bag, placed in 50 ml centrifuge tubes and spun to remove pore waters. An aliquot (approximately 0.5 ml) was used for sulfide analysis, the remainder was filtered through GF/F filters and frozen for analysis of NH_4^+ and DIP. Pore water S^{2-} concentrations were measured monthly between February and July 1998, in October 1998,

November 1998, July 1999 and November 1999. The water content and organic matter content of surficial (0 to 0.5 cm) sediments was determined by drying 10 to 15 g wet weight of sediment at 60°C for 48 h. Sediment samples were reweighed and combusted at 450°C for 12 h to determine loss on ignition (L01).

O$_2$ profiles: Clark style oxygen microelectrodes (100 µm tip diameter, Diamond General Development Corp., Ann Arbor, MI, USA) were used to measure O$_2$ concentrations in pore waters at Hudson's Landing, Azevedo Pond, Hidden Pond and Hummingbird Island sites in July 1999. A single core (9.5 cm diameter) was collected at each site and returned to the lab. Overlying water was gently aerated by bubbling before and during measurements. A micromanipulator was used to position the microelectrodes at 100 µm depth increments to measure O$_2$ concentrations. Triplicate profiles were measured in each core.

Analytical techniques: Water samples from the benthic flux experiments (above) were fixed with Winkler reagents for determination of dissolved O$_2$ and stored in 7 ml glass bottles. O$_2$ concentrations were measured within 4 h of collection using the starch endpoint titra-

tion method (APHA 1985). All nutrient samples were analyzed within 3 mo of collection on a Lachet autoanalyzer using standard seawater methods provided by Lachet (Lachet Instruments). Total dissolved sulfides (H$_2$S, HS$^-$, S^{2-}), referred to in the text as S^{2-}, were analyzed using the methylene blue method (Parsons et al. 1984). Frozen chlorophyll samples were extracted overnight in 90% acetone. The acetone extracts were centrifuged and read on a Turner Model 10 fluorometer (Turner Designs) (Parsons et al. 1984).

Pore water flux calculations: Pore water nutrient concentrations were used to calculate fluxes across the sediment-water interface according to Fick's first law of diffusion:

$$J = ø \, Ds \, \delta C/\delta x \qquad (1)$$

where *J* is the flux, ø is porosity, *Ds* is the effective diffusion coefficient, and $\delta C/\delta x$ is the concentration gradient. The linear portions of profiles were used to calculate the concentration gradient. The effective diffusion coefficient equals *D*ø, where *D* is the molecular diffusion coefficient: 2.3×10^{-5} cm s^{-1} for O$_2$ (Lerman 1979), 1.96×10^{-5} cm s^{-1} for NH$_4^+$ (Berner 1980), $1.63 \times$
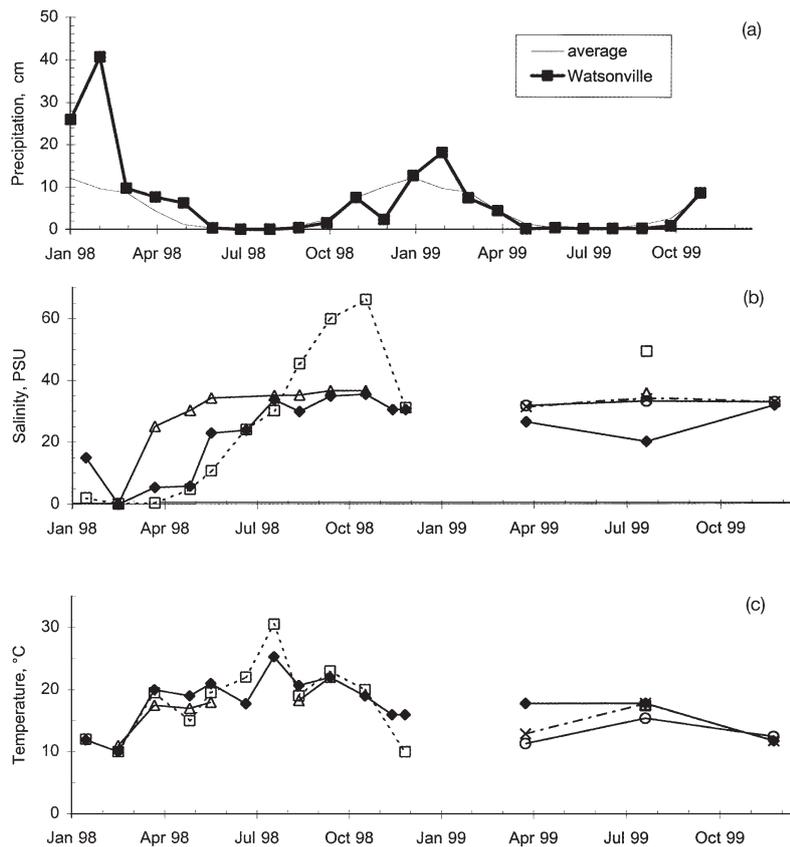

Fig. 2. (a) Precipitation during study period and 50 yr average precipitation (cm) at Watsonville, CA (approximately 10 km from Elkhorn Slough), (b) salinity and (c) temperature at Elkhorn Slough sampling sites. Azevedo Pond (□- - -□), Hudson's Landing (◆◆), Hidden Pond (△—△), Hummingbird Island (×- – - –×), Vierra mudflat (○—○)

10^{-5} cm s^{-1} for DIP (Berner 1980) and 2.53×10^{-5} cm s^{-1} for S^{2-} (Roden & Tuttle 1993). In the text, these fluxes are referred to as diffusive fluxes to distinguish them from the fluxes measured by incubating sediment cores.

Statistical analysis: We performed ANOVA to examine site and seasonal differences in pore water constituents and benthic fluxes. Correlation analysis was also performed examine relationships among constituents and benthic fluxes. Systat (Version 9; SPSS Science) was used for all analyses.

RESULTS

Water column constituents

Central California has a Mediterranean climate with most rainfall normally occurring between October and May, and maximum rainfall occurring between December and March. In January and February 1998, rainfall was 2 to 3 times higher than the 50 yr average

and continued to be above average into May (Fig. 2a). In 1999, rainfall was generally similar to the 50 yr average, except for February (above average) and November (below average). Salinity declined during the rainy season, reaching a minimum in February 1998 when all stations were fresh (Fig. 2b). Salinities gradually increased throughout the summer and fall, reaching peaks of 35.4, 36.5 and 66 PSU in October at Hudson's Landing, Hidden Pond and Azevedo Pond, respectively. Azevedo Pond also had the most extreme temperature variations between 7°C in December and 30°C in July (Fig. 2c). Temperatures at the other sites were similar and ranged between 10 and 25°C (Fig. 2c).

Water column $NO_3^- + NO_2^-$ concentrations were often high following heavy rain events and were inversely correlated with salinity (p = 0.01, r = −0.45), although there was considerable variation between sites (Fig. 3). $NO_3^- + NO_2^-$ concentrations were always high at the head of the slough (Hudson's Landing), usually in excess of 50 µM, while concentrations at the mouth (Vierra mudflat) were below 15 µM. On average, NO_2^- concentrations represented about 20%
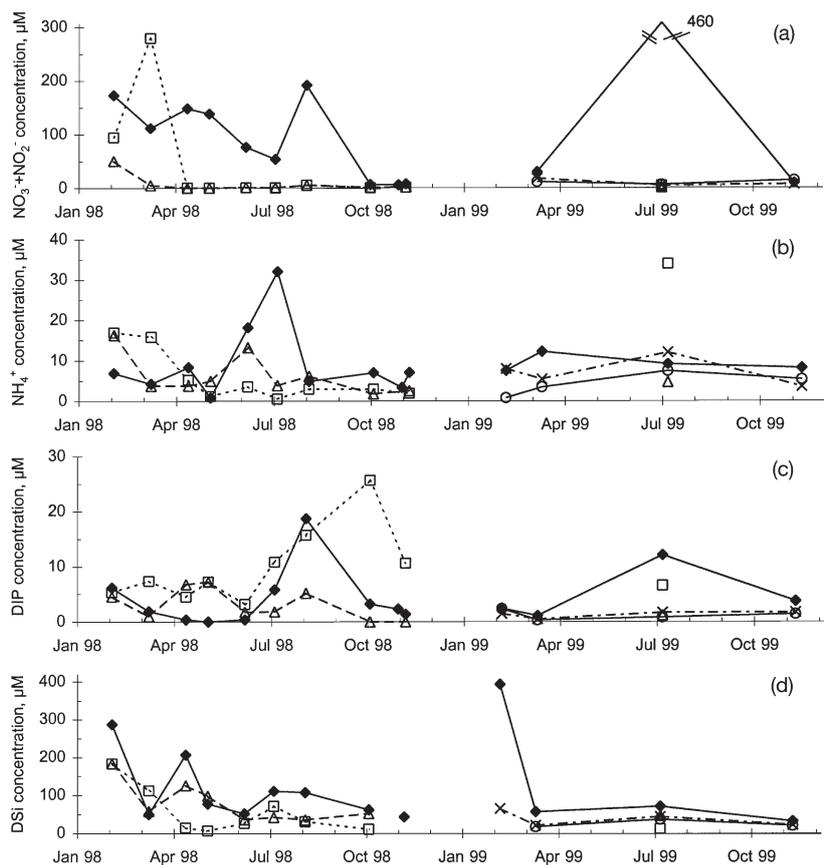

Fig. 3. Nutrient concentrations (µM) at sampling sites in Elkhorn Slough: (a) $NO_3^- + NO_2^-$, (b) NH_4^+, (c) DIP, (d) DSi. Azevedo Pond (□- - -□), Hudson's Landing (◆—◆), Hidden Pond (△—△), Hummingbird Island (×- – - –×), Vierra mudflat (○—○)
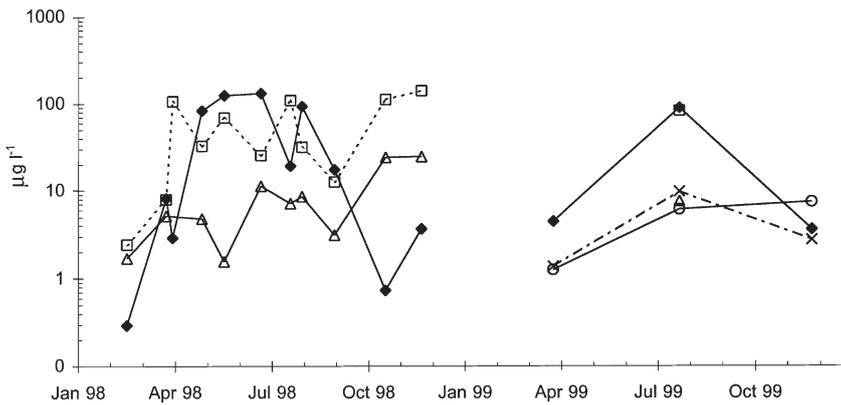
Fig. 4. Chl *a* concentrations (µg l^{-1}) at sampling sites. Azevedo Pond (□- - -□), Hudson's Landing (◆—◆), Hidden Pond (△—△), Hummingbird Island (×- – - –×) Vierra mudflat (○—○)

of the NO_3^- + NO_2^- concentration, although the percentage ranged between 2 and 100% depending on location and site. Water column NH_4^+ concentrations were usually above 3 µM at all sites and were particularly high at Azevedo Pond in the early spring and Hudson's Landing and Hidden Pond in the summer, when concentrations exceeded 13 µM (Fig. 3). Water column DIP concentrations were highest in late summer, particularly at the sites receiving agricultural runoff (Hudson's Landing and Azevedo Pond) where concentrations exceeded 15 µM (Fig. 3). DSi concentrations were usually highest following runoff events and inversely correlated with salinity (p = 0.001,
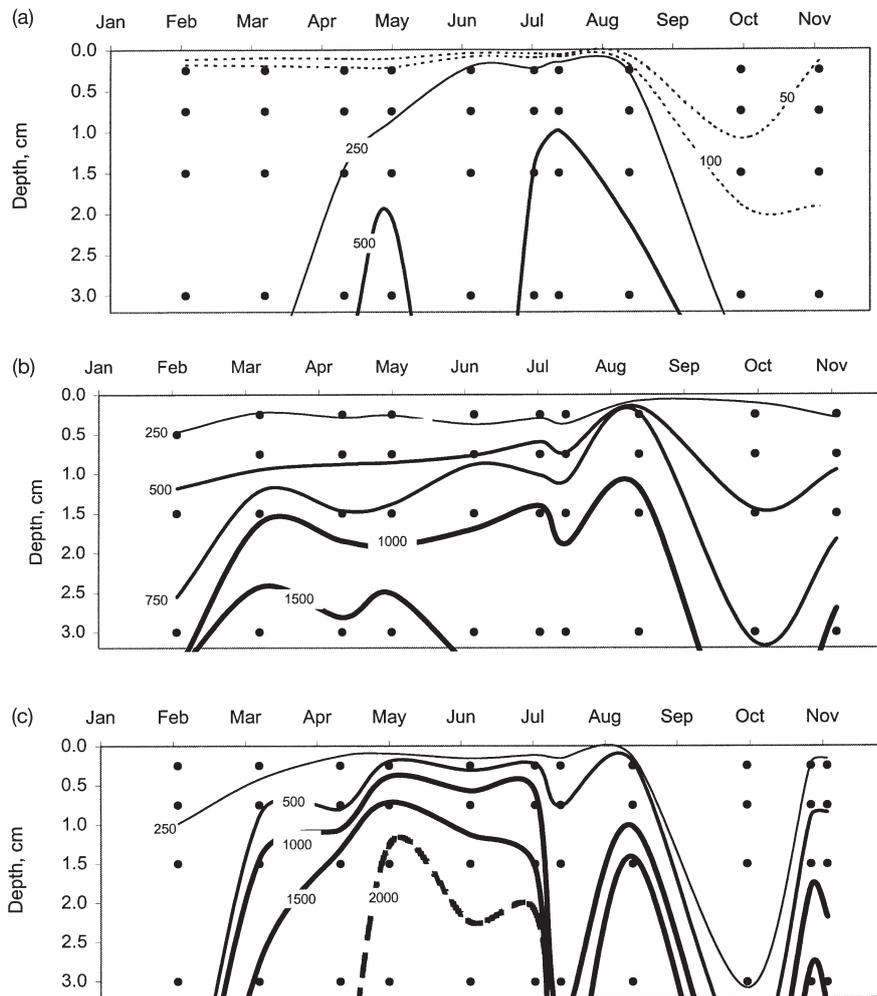

Fig. 5. Contour plot of pore water NH_4^+ concentrations (µM) with depth and sampling period at (a) Hidden Pond, (b) Azevedo Pond and (c) Hudson's Landing in 1998

r = −0.59), particularly at the head of the slough (Hudson's Landing). $NO_3^- + NO_2^-$, NO_2^-, and DSi were all significantly positively correlated with one another at all sites and times (p < 0.04).

Chl *a* concentrations varied by station and season in Elkhorn Slough (Fig. 4). At the head of the slough (Hudson's Landing), chl *a* concentrations usually exceeded 80 µg l^{-1} throughout the summer. The other site receiving agricultural runoff, Azevedo Pond, exhibited no distinct seasonal pattern and concentrations ranged between 2.4 and 140 µg l^{-1}. In Hidden Pond, values ranged between 1.5 and 24.6 µg l^{-1} with peak concentrations occurring in October and November. Chl *a* concentrations in the central slough (Hummingbird Island) and at the mouth of the slough (Vierra mudflat) were always below 10 µg l^{-1}.

Pore water constituents

Pore water concentrations of NH_4^+, DIP and S^{2-} exhibited consistent seasonal and spatial patterns at the different sampling sites. Concentrations always increased with depth in sediment and were generally the highest in the summer, particularly at the agricultural site at the head of the slough (Hudson's Landing). Pore water NH_4^+ concentrations were lowest at the non-agricultural site (Hidden Pond) with concentrations rarely exceeding 500 µM even at 2 to 4 cm depth interval (Fig. 5). In contrast, summer pore water NH_4^+ concentrations were above 1000 and 2000 µM at the 2 to 4 cm depth interval at the 2 agricultural sites, Azevedo Pond and Hudson's Landing, respectively (Fig. 5). Seasonal patterns and site differences were similar for pore water DIP and S^{2-} concentrations
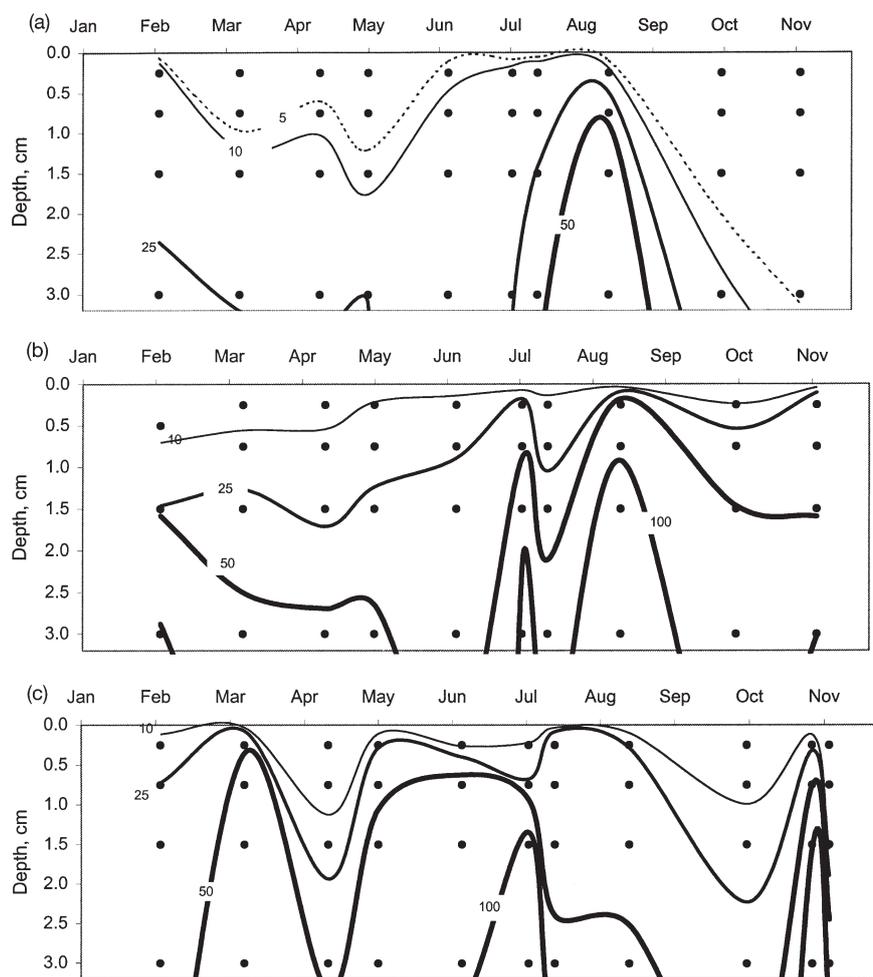

Fig. 6. Contour plot of pore water DIP concentrations (µM) with depth and sampling period at (a) Hidden Pond, (b) Azevedo Pond and (c) Hudson's Landing in 1998
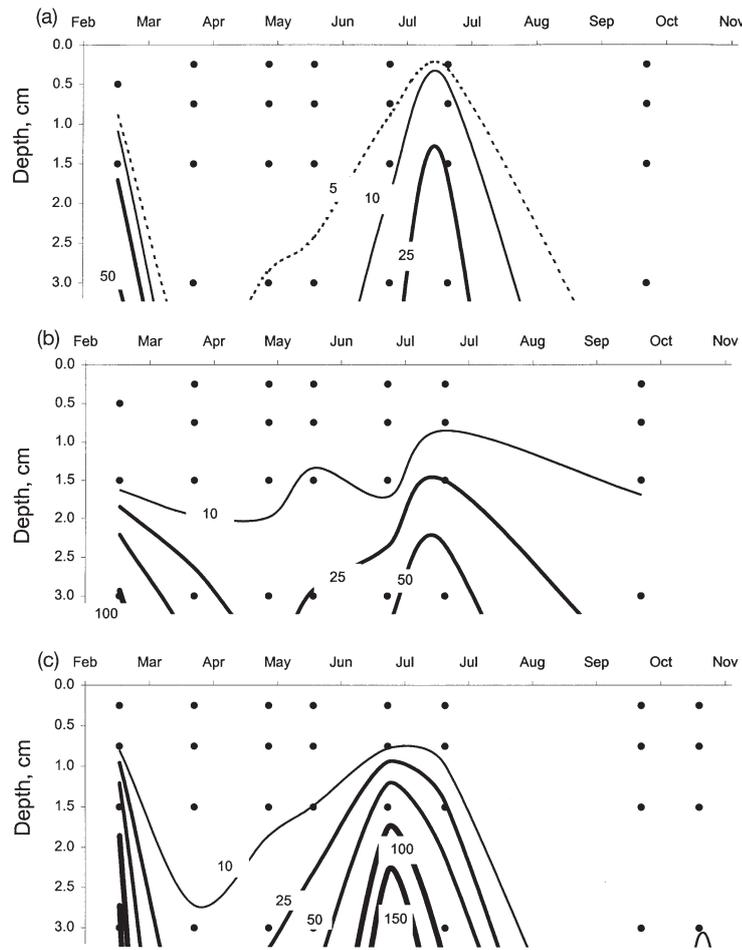
Fig. 7. Contour plot of pore water S^{2-} concentrations (µM) with depth and sampling period at (a) Hidden Pond, (b) Azevedo Pond and (c) Hudson's Landing in 1998

(Figs. 6 & 7), which were lowest at Hidden Pond and highest at Hudson's Landing, particularly in the summer.

Concentrations of pore water NH$_4^+$, DIP and S^{2-} in 1999 were similar to those of 1998, with the highest

Table 1. Sediment water content and organic matter content at Elkhorn Slough sampling sites, mean and standard error (SE)

Station	Water content (%)		Organic content (%)	
	Mean	SE	Mean	SE
Hudson's Landing	76	2	17	2
Lower Azevedo Pond	70	1	13	1
Hidden Pond	68	2	14	1
Hummingbird Island	36	5	7[a]	1
Vierra mudflat	20	1	0.48	0.02

[a]From Caffrey (1996)

concentrations occurring in July at Hudson's Landing (Fig. 8; NH$_4^+$, other data not shown). Pore water NH$_4^+$ concentrations at the mouth (Vierra mudflat) and central slough (Hummingbird Island) were usually very low, often less than 150 µM even at the 2 to 4 cm depth, except during February (Fig. 8). Similar seasonal patterns were observed at the 3 sites for pore water DIP and S^{2-} (data not shown). Pore water NH$_4^+$, DIP and S^{2-} were all significantly positively correlated with one another ($p < 0.001$). ANOVA for each of the pore water constituents indicated that both station and depth were significant variables ($p < 0.001$), while date was only significant for NH$_4^+$ concentrations ($p = 0.003$).

In July 1999, dissolved O$_2$ concentrations measured using O$_2$ microelectrodes decreased rapidly with increasing depth in sediments. The depth of dissolved O$_2$ penetration was greatest at the Hummingbird Island site (6.0 mm), intermediate at Hidden Pond (2.1 mm) and least at Hudson's Landing (0.9 mm) and Azevedo Pond (1.1 mm) (Fig. 9).

The water content of sediments showed little seasonal variation. Annual averages are shown in Table 1 for each site. Similarly, organic matter content was fairly constant over the year except in Azevedo Pond. Organic matter content at this site declined from 27 to 9.4 % between November and March, and then gradually increased back to November levels by the fall (data not shown). Average organic matter content was highest in Azevedo Pond (17 % LOI) and lowest at Vierra mudflat (0.5 % LOI) (Table 1).

Benthic fluxes

SOC was very similar at the 2 agricultural sites, Hudson's Landing and Azevedo Pond during March and April 1998 (Table 2), although in May 1998 SOC was significantly higher in Azevedo Pond than at Hudson's Landing. In 1999, SOC at a non-agricultural site at the mouth (Vierra mudflat) was about $^1/_5$ of the rates at an agricultural site (Hudson's Landing) during the same period. ANOVA revealed that there were significant differences among stations and dates (p = 0.002).
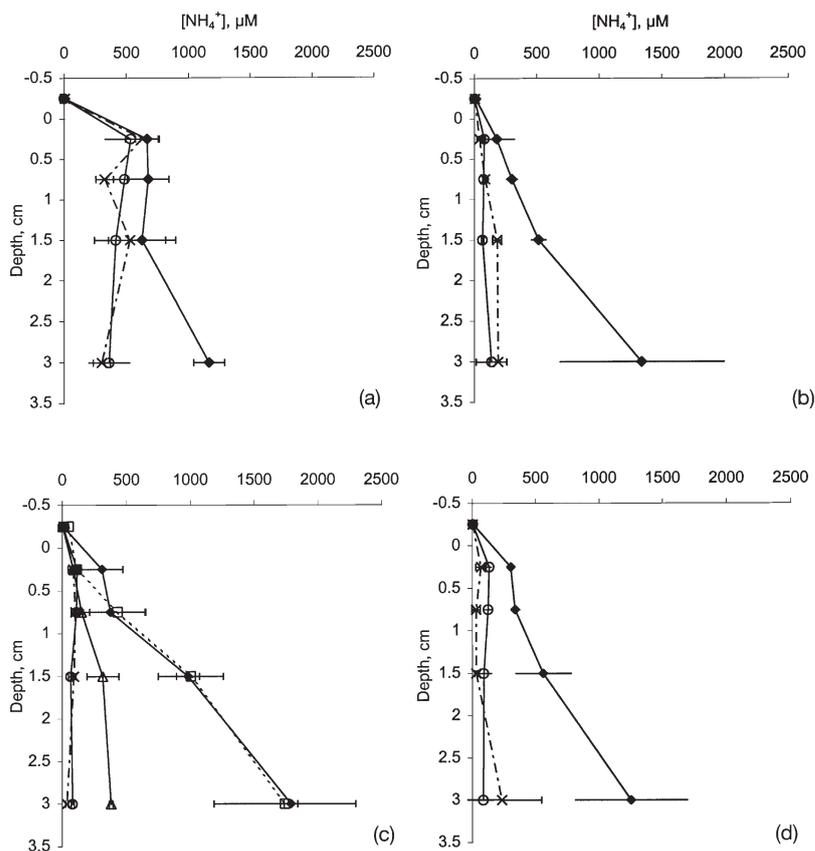

Fig. 8. Pore water NH_4^+ concentrations (µM) at Vierra mudflat (○—○), Hummingbird Island (×- – – –×), Hudson's Landing (◆—◆), Hidden Pond (△—△), and Azevedo Pond (□- - -□) in (a) February, (b) March, (c) July and (d) November 1999. Mean ± SE (n = 3)

Table 2. Benthic fluxes from intact core incubations. Sediment oxygen consumption (SOC), NH_4^+ flux, NO_{3+2}^- flux, NO_2^- flux, NO_3^-, dissolved inorganic phosphate (DIP) flux and dissolved silica (DSi) fluxes in Elkhorn Slough, mean and standard error (SE). Rates in mmol m^{-2} d^{-1}

Station Date	SOC Mean	SE	NH_4^+ Mean	SE	NO_{3+2}^- Mean	SE	NO_2^- Mean	SE	NO_3^- Mean	SE	DIP Mean	SE	DSi Mean	SE
Lower Azevedo Pond														
Mar 98	33.7	1.9	8.6	2.9	−3.29	2.11	1.30	0.45	−4.43	2.67	0.28	1.34	−13.38	2.39
Apr 98	48.6	29.3	8.1	2.6	−0.31	0.12	0.44	0.58	−0.75	0.61	1.08	0.52	−0.25	1.91
May 98	54.8	7.4	−0.4	0.5	0.07	0.07	0.27	0.38	−0.21	0.17	0.97	1.04	16.30	1.96
Hudson's Landing														
Mar 98	37.1	2.0	3.1	0.7	−3.93	2.66	1.51	1.67	−5.44	4.32	−0.45	0.10	2.80	7.54
Apr 98	37.3	21.8	13.8	5.6	−0.71	0.32	6.62	4.09	−7.33	4.39	0.32	0.19	34.87	4.92
May 98	24.2	2.0	5.9	4.6	1.92	0.58	2.27	0.18	−0.35	0.76	−0.10	0.40	0	0
Nov 98	57.8	9.6	3.8	0.6	−1.11	0.68	−0.13	0.05	−0.98	0.63	−0.28	0.19	0	0
Mar 99	66.4	15.9	9.6	3.3	−7.20	1.64	−0.24	0.26	−6.96	1.89	1.22	0.10	12.9	8.5
Jul 99	49.5	29.6	0.1	4.1	4.30	16.01	3.35	2.54	0.95	17.68	−5.55	2.75	−0.8	1.1
Nov 99	107.5	35.9	3.2	3.2	−0.21	0.33	−0.07	0.09	−0.14	0.27	−0.69	0.56	2.8	2.1
Vierra Mudflat														
Mar 99	12.9	7.3	−1.0	1.2	1.86	1.28	0.04	0.05	1.82	1.25	0.06	0.08	1.7	1.1
Jul 99	7.3		1.4	1.3	−0.55	0.98	0.02	0.02	−0.57	0.99	−0.11	0.10	0	0
Nov 99	19.0	2.0	0.4	0.3	−0.39	0.25	−0.04	0.04	−0.35	0.28	0.05	0.13	−1.37	1.11

Jul 99
Oxygen, µM
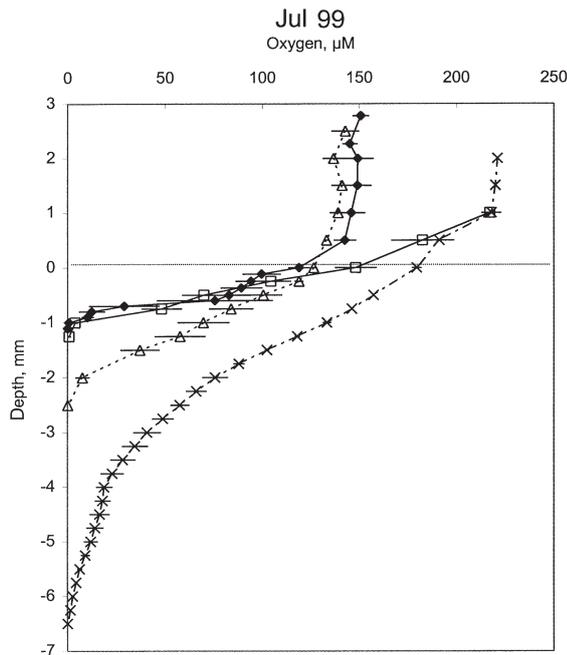
Fig. 9. Dissolved oxygen concentrations (µM) in sediments from Hummingbird Island (×- – - –×), Hudson's Landing (◆—◆), Hidden Pond (△—△) and Azevedo Pond (□- - -□) in July 1999. Mean ± SE (n = 3)

The temporal and spatial patterns for SOC and NH_4^+ fluxes were similar at the 3 sites. NH_4^+ release from sediments was usually comparable at the agricultural sites (Hudson's Landing and Azevedo Pond), except in May at Azevedo Pond when fluxes were near zero (Table 2). In contrast, the non-agricultural site at the mouth (Vierra mudflat) had very low fluxes that varied in direction between NH_4^+ release and uptake by sediments (Table 2). Rates were significantly different between different stations and dates (p = 0.003).

In general, NO_3^- + NO_2^- uptake by sediments occurred at the agricultural sites (Hudson's Landing and Azevedo Pond) and the non-agricultural site at the mouth (Vierra mudflat), although rates were much lower at the non-agricultural site (Table 2). On several occasions, NO_3^- + NO_2^- release occurred, at the agricultural sites in May 1998 and July 1999 (Hudson's Landing site) and at the mouth (Vierra mudflat) in March 1999 (Table 2). There were no significant differences in rates between different stations and dates. NO_2^- release from sediments was observed at the agricultural sites on most sampling dates, with particularly high release (over 1.5 mmol m^{-2} d^{-1}) at Hudson's Landing between March and May 1998, and in July 1999 (Table 2). Rates at the estuary mouth (Vierra mudflat) were near 0 for all sampling dates. Rates were significantly different between stations and dates (p = 0.01).

NO_3^- uptake (calculated by subtracting NO_2^- flux from NO_3^- + NO_2^- flux) was observed at all sites except on 2 occasions, Vierra mudflat in March 1999 and Hudson's Landing in July 1999.

DIP fluxes were near zero at Vierra mudflat while DIP release consistently occurred at Azevedo Pond on all sampling dates (Table 2). Rates were variable at Hudson's Landing showing either small uptake or release in 1998, but higher values in 1999 (Table 2). ANOVA indicated that rates were significantly different between stations and dates (p < 0.001). DSi fluxes were also quite variable and significantly different between stations and dates (p < 0.001). There was a significant release of DSi from Hudson's Landing sediments on about half of the sampling dates; on the remaining dates, rates were near 0. In Azevedo Pond in 1998, DSi was taken up by sediments in March, there was no net flux in April and DSi was released by sediments in May. Rates at Vierra mudflat were low and varied between uptake and release by sediments.

Diffusive fluxes

Diffusive NH_4^+, DIP and S^{2-} fluxes were estimated based on pore water concentrations (Table 3). Diffusive NH_4^+ fluxes were generally highest during the summer when pore water concentrations were high. Seasonal patterns were not as distinct in DIP and S^{2-} fluxes where rates were at a minimum during the spring. Diffusive NH_4^+, DIP and S^{2-} fluxes were highest at Hudson's Landing and Azevedo Pond, intermediate at Hidden Pond and Hummingbird Island, and lowest at Vierra mudflat. Diffusive O_2 flux, measured in July 1999, showed a similar pattern with rates of 23 mmol m^{-2} d^{-1} at Hudson's Landing and Azevedo Pond, 9.4 mmol m^{-2} d^{-1} at Hidden Pond, and 8.1 mmol m^{-2} d^{-1} at Hummingbird Island.

DISCUSSION

The distinct seasonal pattern of central California's rainfall leads to extreme variation in runoff and nutrient inputs to estuaries such as Elkhorn Slough. Rain and runoff were important sources of nutrient input to the slough. Nutrient input and concentration patterns were also clearly related to land use, as seen in the high variability and high concentrations of nutrients in sites nearest to agricultural sources, whereas sites situated in the mainstream of the slough and near the mouth of the estuary were less variable and never experienced the highest concentrations detected upstream. Nutrient inputs from agricultural runoff were particularly evident at Lower Azevedo Pond which was

Table 3. Diffusive fluxes calculated from pore water NH_4^+, DIP and S^{2-} concentrations using Fick's first law. Rates in mmol m^{-2} d^{-1}. nd = no data

Station Date	NH_4^+	DIP	S^{2-}
Lower Azevedo Pond			
Feb 98	3.5	0.57	1.61
Mar 98	7.4	0.28	0.40
Apr 98	6.6	0.23	0.15
May 98	7.3	0.22	0.18
Jun 98	5.8	0.18	0.32
Jul 98	5.7	0.47	0.45
Aug 98	3.8	0.27	nd
Sep 98	4.0	0.50	nd
Oct 98	2.9	0.28	0.12
Nov 98	4.3	0.31	nd
Jul 99	7.8	0.51	nd
Hudson's Landing			
Feb 98	3.0	0.25	1.07
Mar 98	6.8	2.15	0.14
Apr 98	13.0	0.18	0.14
May 98	12.4	0.33	0.28
Jun 98	9.4	0.34	1.04
Jul 98	10.7	0.66	0.53
Aug 98	0.3	−0.13	nd
Sep 98	5.6	0.16	nd
Oct 98	0.9	0.14	0.003
Nov 98	6.4	0.66	nd
Nov 98	5.1	0.29	nd
Feb 99	3.7	0.16	nd
Mar 99	5.3	0.06	nd
Jul 99	7.3	0.25	0.06
Nov 99	4.8	0.07	0.45
Hidden Pond			
Feb 98	0.5	0.36	0.33
Mar 98	0.7	0.10	nd
Apr 98	1.5	0.07	0.03
May 98	2.6	0.11	0.04
Jun 98	1.2	−0.01	0.09
Jul 98	2.2	0.07	0.19
Aug 98	0.9	0.02	nd
Sep 98	1.2	−0.12	nd
Oct 98	0.7	−0.005	0.01
Nov 98	0.5	0.01	nd
Jul 99	1.7	0.09	nd
Hummingbird Island			
Feb 99	0.5	−0.04	nd
Mar 99	1.4	0.002	nd
Jul 99	1.2	−0.005	0.0006
Nov 99	0.8	0.01	nd
Vierra mudflat			
Feb 99	1.31	−0.003	nd
Mar 99	0.09	0.0001	nd
Jul 99	0.02	−0.005	0.000
Nov 99	0.03	−0.008	nd

surrounded by strawberry fields and isolated from tidal mixing with the slough. High nutrient inputs led to high production of organic matter as shown by the chl *a* concentrations in the water column (Fig. 4) and

extensive mats of macroalgae on intertidal mudflats (Allen 1992). Deposition of phytoplankton to sediments as well as burial of macroalgal biomass resulted in high organic matter concentrations at several of the sampling sites.

The pattern of sediment organic matter content indicated large differences among sampling sites, consistent with nutrient inputs and chl *a* concentrations. Organic matter content usually exceeded 10% (Table 1). These organic matter values were much higher than values observed in other estuarine sediments such as Long Island Sound (Mackin & Swider 1989), Buzzards Bay (Banta et al. 1995), Chesapeake Bay (Roden & Tuttle 1993, Reay et al. 1995) and San Francisco Bay (Hammond et al. 1985, Caffrey 1995), where values range between 2 and 8%. Decomposition of this organic matter led to high pore water NH_4^+, DIP and S^{2-} concentrations, particularly in the summer. Except for Hidden Pond, sites with high organic matter content had high pore water nutrient concentrations. The difference between Hidden Pond and other sites with high organic matter content may have been because the major source of the organic matter in Hidden Pond sediments was pickleweed from the marsh surrounding the pond and that plant material was more refractory than phytoplankton or macroalgae. Pore water concentrations were lowest at the mouth at the sandy Vierra mudflat site where the organic matter content was only 0.5%.

The depth of oxygen penetration also reflected the varying intensity of decomposition and available organic matter at the different sites. The least penetration occurred at the agricultural sites (Hudson's Landing and Azevedo Pond) and the greatest at a non-agricultural site in the central Slough (Hummingbird Island) (Fig. 9). Despite having an organic matter content as high as the agricultural sites at Hidden Pond, the depth of O_2 penetration was twice that of the agricultural sites. Perhaps organic matter at Hidden Pond was less labile than at the agricultural sites. Oxygen penetration at Vierra mudflat was not measured. Differences in organic content of the sediments were consistent with the observed variation in benthic fluxes among sites. Low organic matter concentrations at the non-agricultural site at the mouth (Vierra mudflat) resulted in lower rates of decomposition and thus lower SOC and NH_4^+ release from sediments compared to the agricultural sites (Hudson's Landing or Azevedo Pond). These results are consistent with previous research from Elkhorn Slough showing high SOC and NH_4^+ fluxes at sites receiving agricultural runoff compared to sites receiving runoff from the National Estuarine Research Reserve (Caffrey 1996).

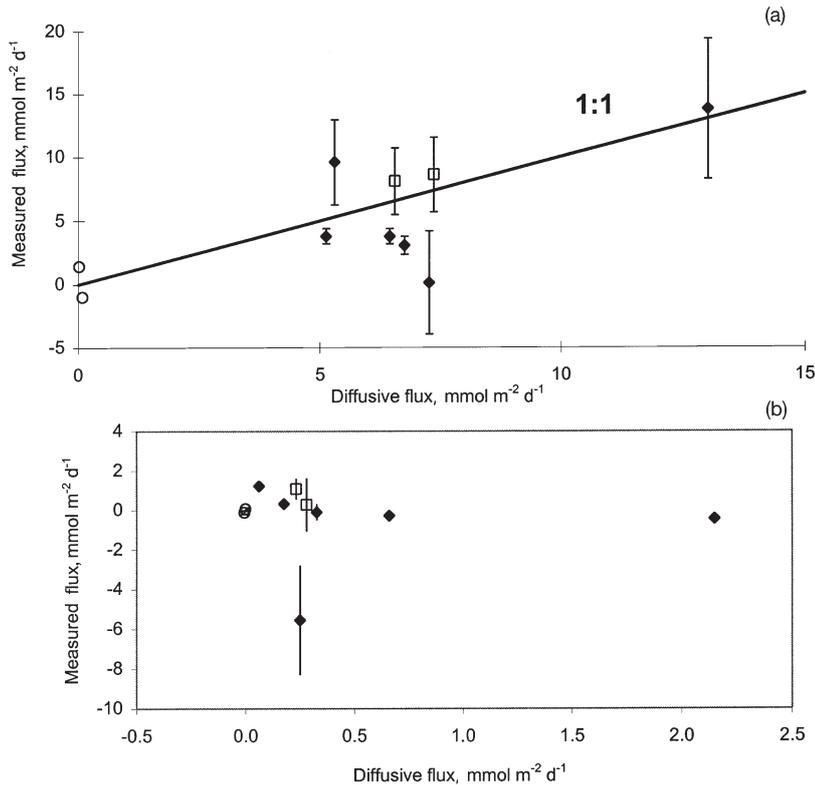In addition to sediment organic matter, other factors influencing benthic fluxes include overlying water

Fig. 10. Measured fluxes versus calculated diffusive fluxes of (a) NH$_4^+$ and (b) DIP in mmol m^{-2} d^{-1}. Azevedo Pond (□), Hudson's Landing (◆), Vierra mudflat (○)

nutrient concentrations, sediment type, benthic macrofauna and benthic micro- and macroalgae. Nutrient concentrations in the overlying water were important, with lower concentrations in the overlying water than in pore water resulting in release of nutrients, while higher concentrations in the overlying water than pore water resulted in uptake by sediments. Water column concentrations and fluxes of DIP were significantly correlated (p = 0.03), as were concentrations and fluxes of NO$_2^-$ (p = 0.02). DIP concentration and flux were negatively correlated (r = –0.61), suggesting that water column DIP concentrations may control DIP flux across the sediment-water interface. In contrast, NO$_2^-$ concentration and flux were positively correlated (r = 0.64). Because NO$_2^-$ concentrations in sediments were usually very low, we infer that production of NO$_2^-$ in sediments by nitrification or the reduction of NO$_3^-$ to NO$_2^-$ and subsequent flux out into the overlying water was responsible for the high water column concentrations. Water column concentrations and fluxes were not significantly correlated for the other nutrients.

Comparing diffusive and measured fluxes can also provide insights into the factors controlling exchange across the sediment-water interface. In general, measured NH$_4^+$ fluxes were less than the diffusive flux, suggesting that diffusion of pore water NH$_4^+$ was not the sole determinant of the benthic NH$_4^+$ flux, and that some pore water NH$_4^+$ was removed either by benthic micro- or macroalgae or by nitrification (Fig. 10a). There was no correlation between diffusive and measured DIP fluxes (Fig. 10b), probably because both measured and calculated fluxes were very close to 0. Pore water DIP way have been adsorbed to FEOOH at the aerobic sediment-water interface and thus was unable to diffuse across the sediment-water interface.

Benthic macrofauna can greatly enhance fluxes across the sediment-water interface by bioturbation as well as excretion of NH$_4^+$ (Hammond et al. 1985, Mackin & Swider 1989, Peligri & Blackburn 1994, Banta et al. 1995, Caffrey et al. 1996). Often the diffusion coefficient used in the diffusive flux calculation is increased to account for macrofauna bioturbation (Hammond et al. 1985, Mackin & Swider 1989). In general, benthic macrofauna were not very abundant at these sites (J. Caffrey pers. obs.), so the diffusion coefficient was not adjusted in this study. However, on a majority of the occasions when macrofauna were present, measured NH$_4^+$ fluxes were higher than diffusive fluxes. This was particularly apparent in March 1999 at Hudson's Landing when numerous amphipods were observed in the benthic flux cores.

Benthic micro- and macroalgae at the sediment-water interface can also remove nutrients from the water column. Although the highest uptake by micro- and macroalgae occurs in the light, some uptake can occur in the dark (Sundbäck et al. 1991, Thybo-Christesen et al. 1993). Benthic algal concentrations were not measured in this study. However, previous work found that benthic chl *a* levels in the top centimeter ranged from 21 mg m^{-2} at sites in the ESNERR to 574 mg m^{-2} at sites receiving agricultural runoff (Caffrey 1996). Nutrient uptake by this active community may explain the negative fluxes of DIP, DSi, NO$_3^-$+NO$_2^-$ and NH$_4^+$. Benthic diatoms, which are commonly observed in surficial sediments (K. Parker pers. comm.), may be responsible for the high DSi uptake occasionally observed. Nutrient uptake by benthic algae was probably underestimated, because all of the incubations were performed in the dark. Given the high nutrient inputs to Elkhorn Slough, further studies examining the role of benthic
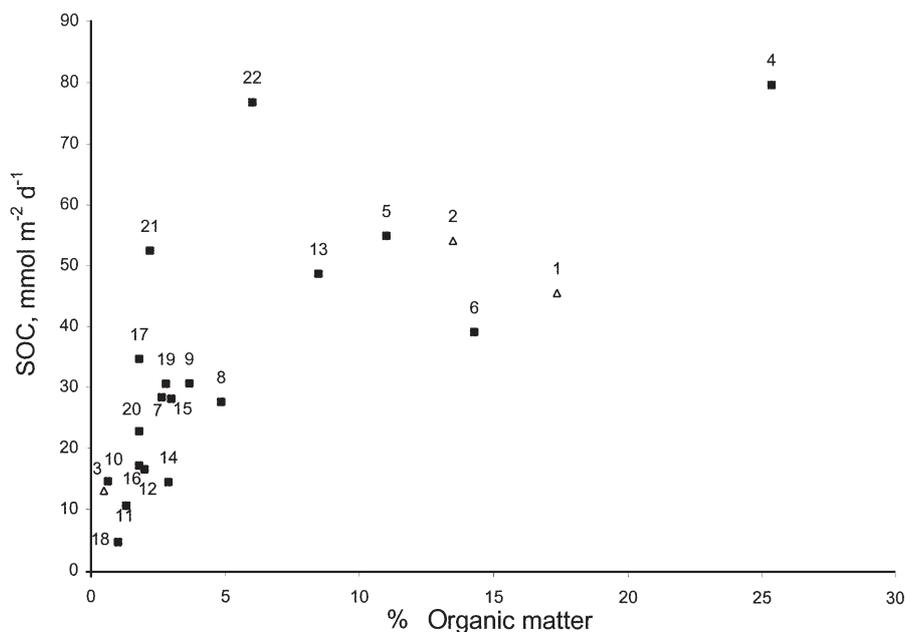
Fig. 11. Sediment oxygen consumption (SOC, mmol m^{-2} d^{-1}) versus % organic matter in Elkhorn Slough and other estuaries. Elkhorn Slough (this study △): 1 Azevedo Pond, 2 Hudson's Landing, 3 Vierra mudflat. Other studies: ■. Elkhorn Slough: 4 Agricultural sites, 5 Grazing sites, 6 Reserve sites (Caffrey 1996), 7 San Francisco Bay (Hammond et al. 1985, Caffrey et al. 1996), 8 Chesapeake Bay (Kemp & Boynton 1992, Reay et al. 1995), 9 Narragansett Bay (McCaffrey et al. 1980, Seitzinger et al. 1984), 10 Neuse River (Rizzo et al. 1992), 11 North Sea (Upton et al. 1993), 12 Bering Sea (Henriksen et al. 1993), 13 Parker River (Hopkinson et al. 1999), 14 Baltic Sea (Enoksson et al. 1990, Jørgensen 1996b), 15 Ochlockonee Bay (Twilley et al. 1999), 16 Mobile Bay (Twilley et al. 1999), 17 Fourleague Bay (Twilley et al. 1999), 18 Trinity-San Jacinto estuary (Twilley et al. 1999), 19 Guadeloupe estuary (Twilley et al. 1999), 20 Nueces estuary (Twilley et al. 1999), 21 Laguna Madre (Twilley et al. 1999), 22 Flax Pond (Mackin & Swider 1989)

micro- and macroalgae in nutrient cycling are needed.

Benthic flux and pore water NH$_4^+$ measurements from this study were similar to previous measurements in Elkhorn Slough (Caffrey 1996). SOC and NH$_4^+$ flux from sites in Elkhorn Slough receiving agricultural and other non-point source runoff are among the highest recorded values from the literature (Nixon 1981, Hammond et al. 1985, Mackin & Swider 1989, Enoksson et al. 1990, Dollar et al. 1991, Kamp-Nielsen 1992, Kemp & Boynton 1992, Rizzo et al. 1992, Upton et al. 1993, Banta et al. 1995, Lohse et al. 1995, Reay et al. 1995, Caffrey et al. 1996, Hopkinson et al. 1999, Twilley et al. 1999, Mortazavi et al. 2000). In contrast, benthic fluxes from Vierra mudflat at the well-flushed mouth of Elkhorn Slough were similar to other estuarine and shallow coastal systems. Across many estuaries, sediment oxygen consumption is generally highest in systems with high sediment organic matter content (Fig. 11), although other factors such as water depth, bottom water column dissolved oxygen concentrations, sediment grain size, quality of organic matter and rate of organic matter deposition to sediments may also be significant.

This study suggests that high nutrient runoff has a local rather than estuary-wide effect on sediment bio-geochemical processes in Elkhorn Slough. At the head of Elkhorn Slough, where residence time during summer months reaches 50 d (Largier et al. 1997), high nutrient concentrations lead to increased chl a concentrations, high organic content in the sediments and ultimately high rates of benthic mineralization, particularly SOC and NH$_4^+$ release. In contrast, nutrient inputs to the well-flushed, lower reaches of the slough are rapidly diluted by tidal exchange between Elkhorn Slough and Monterey Bay, resulting in low to moderate concentrations of chl a and sediment organic matter and low to moderate benthic remineralization. This is a sharp contrast to many stratified and partially stratified estuaries that have longer residence times and more estuarine-wide effects of high nutrient runoff (Kemp & Boynton 1992, Nixon et al. 1996, Herbert 1999, Twilley et al. 1999). This study suggests that 2 factors appear to control the processing of organic matter in sediments: (1) proximity to agricultural or other non-point source runoff; and (2) tidal exchange or residence time within the system. These factors lead to large differences in nutrient and oxygen status of local environments, which can differ significantly from nearby sites and from the overall average conditions of the estuary.

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