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A Regional Survey of the Extent and Magnitude of Eutrophication in Mediterranean Estuaries of Southern California, USA

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Abstract The magnitude and extent of eutrophication was assessed at 27 segments in 23 estuaries in the Southern California Bight (SCB) between October 2008 and 2009. We applied thresholds from the existing assessment frameworks from both the European Union and the U.S. National Eutrophication Assessment to measurements of three indicators [macroalgae biomass and cover, phytoplankton biomass, and dissolved oxygen (DO) concentration] to categorize eutrophic condition in each estuary. Based on these frameworks, a large fraction of segments had moderate or worse eutrophic condition—78 % based on macroalgae, 39 % for phytoplankton, and 63 % for DO. Macroalgal biomass exceeding 70 g dw m^{-2} and 25 % cover was found at 52 % of sites during any sampling event and in 33 % of segments for 8 weeks or longer, a duration found to negatively impact benthic infauna. Duration of hypoxic events (DO<4 mg L^{-1}) was typically short, with most events less than one day; although 53 % of segments had at least one event longer than 24 h. Assessment frameworks of eutrophic condition are likely to evolve over time as the body of literature on eutrophication grows, including aspects such as the applicability of indicators in specific habitat types, indicator thresholds, and how event frequency and duration are incorporated. This paper informs this debate by discussing how eutrophic conditions in SCB estuaries are categorized using different indicators and thresholds. To this end, categorization of estuarine eutrophic condition was found to be very sensitive to the type of threshold, how data are integrated to represent duration or spatial extent, and how indicators are used as multiple lines of evidence.

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Introduction

Eutrophication of estuaries is a global environmental issue, with demonstrated links between anthropogenic nutrient loading to coastal waters, harmful algal blooms, hypoxia, and impacts on aquatic food webs (Valiela et al. 1992; Smith et al. 1999; Kamer and Stein 2003). Management of eutrophication and development of appropriate nutrient water quality goals are often hampered by a lack of regional monitoring data characterizing the symptoms, extent, and magnitude of the problem. A 2007 study of US estuaries conducted through the National Oceanic and Atmospheric Administration's (NOAA) National Estuarine Eutrophication Assessment (NEAA) found the majority of estuaries assessed had overall conditions rated as moderate to highly eutrophic (Bricker et al. 2008), yet highlighted significant data gaps. In the Southern California Bight (SCB), one of the most populated regions in the US, only eight of the region's 76 enclosed bay, lagoonal, and river mouth estuaries were on the list of NEAA study sites, and there were adequate data in only two of the eight SCB estuaries to assess eutrophic status. Among SCB estuaries, smaller "bar-built" lagoons and river mouth estuaries, which represent 22 % of the areal extent but 82 % by number in the region, are particularly data poor (Fong and Zedler 2000). These Mediterranean-type "bar-built" estuaries often experience restriction or complete closure to surface water tidal exchange due to the formation of sandbars at their inlets (Webb et al. 1991; Largier et al. 1996). Consequently, they have increased susceptibility to eutrophication due to restricted flushing (Painting et al. 2007; Zaldivar et al. 2008). Additional data on the status of eutrophication in SCB estuaries are needed to determine the extent and magnitude of eutrophication in the region.

Over the past decade, much work has been done to establish standardized methodologies to assess eutrophication (Bricker et al. 2003, Zaldivar et al. 2008, Andersen et al. 2011, Devlin et al. 2011) and conduct surveys to evaluate the magnitude and extent of eutrophication (Bricker et al. 1999; Borja et al. 2009a; Andersen et al. 2011; Devlin et al. 2011; Garmendia et al. 2012). These assessment methodologies are the foundation worldwide for routine monitoring and establishment of water quality and biological objectives that are used to protect pristine habitat, identify impaired waterbodies, and provide targets for restoration or mitigation of systems where adverse effects of eutrophication have already occurred. Studies comparing assessment results generated for the same estuary have indicated that results vary slightly depending on which framework is applied (Devlin et al. 2011, Garmendia et al. 2012). Many of the frameworks apply similar indicators, but differences in time scales of data analysis (seasonal versus annual), characteristics included in the indicator metrics (concentration, spatial coverage, frequency of occurrence), and how to combine indicators into multiple lines of evidence had an effect on the overall outcome of the assessment (Devlin et al. 2011). Most of these assessment frameworks combine indicators of pressure (nutrient loads, estuarine surface water nutrients) with response indicators, often resulting in a numeric integrative index developed by expert best professional judgment (Ferreira et al. 2011). Evaluating the applicability of various indicators and respective thresholds from these frameworks and quantifying effect of data integration decisions is an important exercise to consider whether or how to adapt these assessment frameworks to use outside of the region in which they were originally developed.

The Mediterranean-type estuaries of the SCB are an excellent test case in which to conduct this evaluation. The SCB regional monitoring program (RMP) is an integrated multidisciplinary and multi-institutional program that provides a unique platform for collecting data for bight-wide perspectives on a number of management questions. The objectives of the Bight 2008 Eutrophication Assessment of the SCB RMP were to: (1) evaluate which indicators are relevant in Mediterranean estuaries such as those in the SCB, (2) explore how spatial and temporal integration of monitoring data affects the assessment of eutrophication status, and (3) estimate the extent and magnitude of eutrophication in SCB estuaries using appropriate indicators and thresholds. Results from this study can be used to inform the ongoing refinement of assessment approach and data integration decisions that affect the results of assessment of eutrophication in Mediterranean estuaries.

Methods

Study Area The SCB (Fig. 1) is an open embayment on the U.S. west coast between Point Conception, California, and Cabo Colnett (south of Ensenada, Mexico) and contains 76 estuaries ranging in size from 1 ha to over 50,000 ha. The SCB landscape overall is a highly developed urban environment; however, the watersheds within the SCB are highly variable in terms development. Watersheds ranged from a percent impervious surface of 1 % (San Mateo Lagoon and Topanga Canyon Lagoon) to greater than 60 % impervious (Anaheim and San Diego Bays). This conversion of open land into impervious surfaces has included dredging and filling over 75 % of bays and estuaries and extensive alterations of coastal streams and rivers (Brownlie and Taylor 1981; Horn and L. G. Allen 1985; NRC 1990; Zedler 1996). Agriculture was also prevalent in some watersheds ranging from 2 % agricultural land use (Anaheim Bay) to 30 % (Santa Margarita River Estuary). The SCB has a Mediterranean climate, with an average annual rainfall of 10–100 cm (Nezlin and Stein 2005), falling primarily during winter months (December through March), in approximately 20 annual storm events (Ackerman and Weisberg 2003). Winter runoff to the SCB contributes 95 % of the total annual runoff volume and 67 % of the total annual nitrogen and phosphorus loads (Ackerman and Schiff 2003; Sengupta et al. 2013).

Study Design The Bight 08 Eutrophication Assessment was a synoptic study of eutrophication indicators, estuarine water column nutrient concentrations, and riverine nutrient loads monitored for one water year between October 2008 and October 2009. Estuaries were randomly selected from a comprehensive list of estuaries, proportional to the number of estuaries in each geoform: enclosed bay, lagoon or river mouth, and tidal inlet status (open to tidal exchange, anthropogenically muted through tide gates, perennially or seasonally closed with no tidal exchange) in the northern portion of

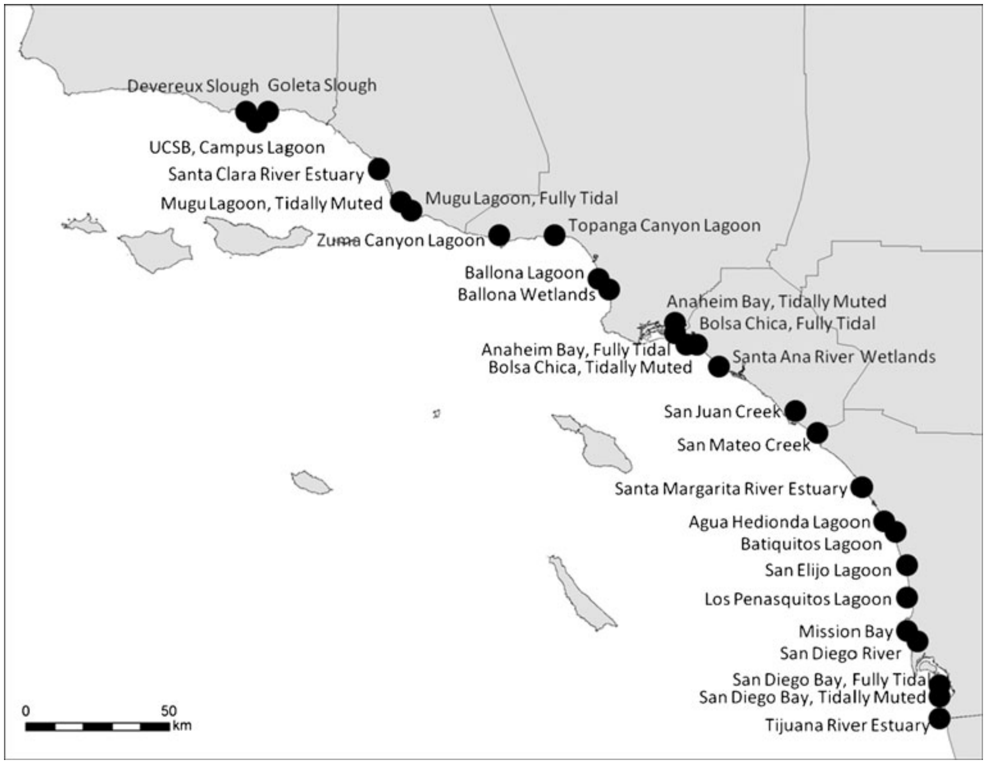

Fig. 1 Map of Bight 08 Eutrophication Assessment segment sites

the SCB; all estuaries in San Diego County were included. Because primary producer expression is highly spatially variable within an estuary, we endeavored to make the results comparable across estuaries by selecting an index area, or "segment," which varies in size depending on the estuary (Table 1). The segments were proximal to the source of freshwater nutrient loading rather than the ocean inlet, representing a location within the estuary that is more likely exhibit symptoms of eutrophication. Segment sites were revisited each time period to determine seasonal variability in eutrophication indicators. Consequently, this survey represents a conservative assessment of eutrophication for each estuary and the region overall. A total of 27 segments were selected in 23 estuaries (Tables 1 and 2, Fig. 1).

Estuaries are highly variable in how they respond to nutrient loading due to differences in site-specific controls on hydrology and other factors (Dettmann 2001; Pinckney et al. 2001; Zaldivar et al. 2008; Duarte 2009; Duarte et al. 2009). Several studies have demonstrated the shortcomings of using estuarine nutrient concentrations or loads alone to predict eutrophication (Cloern 2001; Kennison et al. 2003; Devlin et al. 2007). Consequently, there has been a shift toward the use of eutrophic response indicators to estimate extent and magnitude of eutrophication (Bricker et al. 2003; Devlin et al. 2007; Zaldivar et al. 2008). Therefore, reporting of eutrophic condition in SCB estuaries was focused on the ecological response to nutrient over-enrichment rather than estuarine nutrient concentrations or watershed nutrient loads. For this assessment, data on a number of indicators were collected, though we opted to focus on dissolved oxygen and primary producer abundance because they have the most supporting data in the literature. Nutrient loads for each watershed are also reported for context.

Field and Laboratory Methods Field methods consisted of three types of sampling: (1) continuous monitoring of dissolved oxygen, chlorophyll *a* fluorescence, temperature, salinity, pH, and turbidity using a moored data sonde; (2) sampling of estuarine surface water and sediment nutrients and primary producer biomass every other month; and (3) monitoring of storm and dry weather freshwater nutrient loads. Riverine nutrient loads were estimated using a combination of field measurements and modeling.

Riverine Nutrient Loads Total riverine nutrient fluxes to the SCB were estimated using empirical wet and dry weather data for monitored watersheds in combination with modeled wet weather fluxes for unmonitored watersheds. Continuous discharge was measured with flow gauges at some sites and from ratings curves based on continuous water level (recorded with a HOBO data logger, Onset Corp.) and quarterly measurements of flow and channel topography. Discrete total nitrogen (TN) and phosphorus (TP) samples were collected for wet and dry weather (October 2008–2009) and analyzed via persulfate digest (Patton and J. R. Kryskalla 2003) and analyzed colorimetrically using an autoanalyzer. The spreadsheet

Table 1 Bight 2008 eutrophication assessment estuary segments

Name (code)[a]	Latitude	Longitude	Watershed area (km^2)	Estuary area (m^2)	% of Estuary in segment	Habitat type				Max salinity (ppt)	Min salinity (ppt)	Annual TP load (kg)[b]	Annual TN load (kg)[b]
						Subtidal unveg	Subtidal eelgrass	Mudflat	Marsh				
Goleta Slough (GS)	34.4203	−119.8437	119	804,812	50 %	20 %	0 %	12 %	68 %	57.6	35.4	8,518	20,369
Devereux Lagoon (DL)	34.4137	−119.8777	10	231,885	100 %	41 %	0 %	27 %	32 %	59.6	18.1	3,198	7,287
UCSB Campus Lagoon (UCL)	34.4103	−119.8485	10	124,229	75 %	96 %	0 %	0 %	4 %	55.9	46.9	222	1,257
Santa Clara River (SCR)	34.2317	−119.2626	4,210	1,412,587	75 %	50 %	0 %	48 %	2 %	3.2	1.6	22,483	239,378
Mugu Lagoon, muted (MLM)	34.1098	−119.1407	803	14,098,340	25 %	8 %	0 %	13 %	79 %	61.7	40.1	65,281	209,210
Mugu Lagoon, full (MLF)	34.1033	−119.0927	803	14,098,340	15 %	8 %	0 %	13 %	79 %	42.8	32.7	65,281	209,210
Topanga Canyon Lagoon (TC)	34.0386	−118.5831	51	3,678	100 %	100 %	0 %	0 %	0 %	41.3	12.4	25	187
Zuma Canyon Lagoon (ZC)	34.0143	−118.8210	23	17,544	100 %	56 %	0 %	13 %	31 %	7.3	0.6	154	578
Ballona Lagoon, muted (BL)	33.9709	−118.4583	354	22,950	100 %	71 %	0 %	29 %	0 %	54.4	45.6	13,939	95,098
Ballona Wetlands-Muted (BW)	33.9648	−118.4482	354	627,857	100 %	0 %	0 %	19 %	81 %	48.7	46.0	13,939	95,098
Anaheim Bay/SealBeach-Muted (SBM)	33.7463	−118.0703	130	4,194,998	10 %	28 %	11 %	5 %	57 %	55.3	51.3	550	13,832
Anaheim Bay/SealBeach-Full (SBF)	33.7354	−118.0756	130	4,194,998	50 %	28 %	11 %	5 %	57 %	53.4	50.7	550	13,832
Bolsa Chica-Full (BCF)	33.7002	−118.0427	25	1,708,457	50 %	62 %	8 %	21 %	10 %	53.4	51.4	25	2,500
Bolsa Chica-Muted (BCM)	33.6960	−118.0457	59	651,724	50 %	72 %	0 %	7 %	20 %	55.2	49.8	1,081	5,840
Santa Ana R. Wetlands-Muted (SAR)	33.6378	−117.9547	4,336	331,670	75 %	37 %	0 %	14 %	50 %	52.7	51.1	16,528	89,981
San Juan Creek (SJC)	33.4628	−117.6832	458	64,839	100 %	35 %	0 %	65 %	0 %	29.7	9.3	7,695	30,062
San Mateo Lagoon (SMC)	33.3865	−117.5937	346	126,262	100 %	20 %	0 %	0 %	80 %	1.4	0.7	656	8,895
Santa Margarita Estuary (SME)	33.2346	−117.4090	1,918	1,020,806	75 %	33 %	0 %	24 %	43 %	51.7	13.7	19,177	137,799
Agua Hedionda Lagoon (AHL)	33.1421	−117.3284	546	1,410,480	50 %	60 %	17 %	11 %	12 %	52.4	49.8	3,283	57,185
Batiquitos Lagoon (BQL)	33.0910	−117.3008	131	2,022,063	50 %	18 %	29 %	20 %	33 %	51.6	49.3	46,966	60,037
San Elijo Lagoon (SEL)	33.0127	−117.2722	210	1,261,824	75 %	9 %	0 %	54 %	38 %	47.3	17.2	2,080	63,262
Los Penasquitos Lagoon (LPL)	32.9350	−117.2604	244	1,306,657	50 %	7 %	0 %	7 %	85 %	51.3	48.1	5,669	28,325
Mission Bay-Full (MB)	32.7720	−117.2138	118	8,795,281	15 %	18 %	75 %	5 %	2 %	55.9	52.5	7,902	41,283
San Diego River (SDR)	32.7597	−117.2153	1,120	1,142,328	50 %	45 %	0 %	23 %	32 %	49.4	29.9	4,629	54,066
San Diego Bay-Muted (SDM)	32.6211	−117.1175	362	17,898,631	5 %	28 %	0 %	71 %	1 %	71.1	63.2	8,045	147,537
San Diego Bay-Full (SDF)	32.5992	−117.1190	1,037	50,094,111	10 %	63 %	22 %	13 %	2 %	55.7	48.1	8,045	147,537
Tijuana River Estuary (TJE)	32.5683	−117.1313	4,452	2,755,819	25 %	7 %	0 %	17 %	76 %	52.7	46.2	82,988	348,018

[a] Sites indicated as “muted” have an anthropogenically muted tidal regime through presence of dikes, tide gates, or weirs

[b] Loads calculated from continuous flow data and discrete nutrient concentration measurements collected from the freshwater source

Table 2 Summary of data integration options that were evaluated through sensitivity analyses

Issue	Indicator		
	Macroalgae	Phytoplankton	Dissolved oxygen
Data format	• Wet weight ✓ Dry weight	✓ Continuous • Discrete	✓ Continuous
Data integration period	• Period with highest biomass/cover	• Maximum instantaneous value	• Minimum instantaneous value
	✓ Average of two consecutive periods of highest biomass/cover	• Various percentiles (90th percentile)	• 5th percentile
	• Bight-wide index period	✓ Annual average	✓ 10th percentile
	• Annual average		15th percentile
Smoothing applied to the data set	Transect values generated from	• Instantaneous	• Instantaneous
	✓ Quadrat averages	• Quadrat maximum	✓ Hourly running average
	• Quadrat percentile		
	• Quadrat maximum		
Spatial extent over which data were integrated	Segment values generated from	✓ Single location	✓ Single location
	• Average of transects		
	• Maximum transect		
	• Percentile of transect data		

Check mark designates options applied for the B08 Eutrophication Assessment

model originally developed by Ackerman and Schiff (2003), based on the Rational Method, was updated and modified to predict TN and TP loads using updated land use-specific runoff concentrations for nutrients (Howard et al. 2012; Sengupta et al. 2013).

Dissolved Oxygen and Water Column Physio-chemistry Water column physiochemical parameters and water level were measured continuously using a YSI 6600 data sonde from January through October 2009. Each sonde was outfitted with a conductivity/temperature sensor, ROX optical dissolved oxygen probe, extended deployment pH probe, chlorophyll optical sensor, and a turbidity optical sensor. All sensors were treated with anti-fouling tape and calibrated at a minimum of once monthly. Sondes were deployed at one location in each segment, in bottom water (30 cm from the sediment surface). Measurements were collected every 15 min, and an hourly running average was applied to the data set.

Macroalgae Biomass and Cover Macroalgal biomass and cover was measured at three 30–50 m transects in the lower intertidal zone (Kennison et al. 2003). Percent cover was measured at ten randomly allocated points along each transect using the point intercept method with 0.5 m^2 quadrats. Biomass was comprehensively collected at five of the quadrat locations from a prescribed surface area. Biomass samples were stored at 4 °C and processed within 24 h of collection. In the laboratory, algal samples were cleaned of macroscopic debris, mud, and animals. Excess water was shed from each sample, weighed wet, dried at 60 °C to a constant weight, and then weighed dry.

Phytoplankton Biomass Phytoplankton biomass was estimated from fluorescence measurements collected via in situ optical probe (YSI 6600 sonde, chlorophyll fluorescence probe) and discrete chlorophyll *a* water grab samples taken every other month. Water column chlorophyll *a* samples were filtered on a Whatman GF/F and frozen for subsequent analysis using EPA 445 protocols on a Turner Designs fluorometer within 28 days of collection. In situ chlorophyll fluorescence probes were maintained according to factory specifications and were routinely calibrated. Fluorescence measurements were calibrated to chlorophyll *a* concentrations using least-squares regression of daily averaged data probe measurements and discrete concentration data collected on the same day. The least squares fit had R^2 values ranging from 0.413 to 0.995 with an average of 0.747 and 89 % of sites had an R^2 greater than 0.5. The poor fit for some sites is likely related to the disparity between where the measurements were collected (surface waters for discrete samples, bottom water for chlorophyll fluorescence probe). Most sites were shallow and the depth difference was insignificant, but for the few deeper sites, it seemed to impact fit.

Eutrophication Indicators Several assessment frameworks have been developed to assess eutrophic condition of estuaries utilizing a range of indicators (Bricker et al. 2003; Devlin et al. 2007; Zaldivar et al. 2008). The most representative assessment frameworks incorporate annual data with sampling throughout the year, to capture frequency of occurrence and spatial extent in indicator metrics, and use of a combination of indicators into an overall condition rating (Devlin et al. 2011). For this study, we selected individual indicators from established assessment frameworks to evaluate how well they worked in SCB estuaries. Indicators were evaluated from the European Union-Water Framework Directive (EU-WFD) and the U.S. Assessment of Estuarine Trophic Status (ASSETS). The EU-WFD was developed to regulate and monitor water bodies in EU member countries, organizing management of waterbodies by catchment, and standardizing protocols across Europe (Borja et al. 2006; Hering et al. 2010). Several assessment frameworks are associated with the EU-WFD (Ferreira et al. 2011; Birk et al. 2012); we selected two of these that utilized indicators prevalent in SCB estuaries: French Research Institute for Exploration of the Sea (IFREMER) and the United Kingdom WFD protocols (UK-WFD). ASSETS (Bricker et al. 2003) was developed to assess the status of eutrophication in U.S. estuaries through NOAA's National Estuarine Eutrophication Assessment (Bricker et al. 1999). These frameworks characterize the state of the estuary by assessing response indicators, rather than ambient physical or chemical variables alone, although pressure variables such as nutrient loads and concentrations are included in the overall assessment (Borja and Dauer 2008; Borja et al. 2011b).

Because the estuaries in this study were not comprehensively characterized for pressure and susceptibility factors, which are required for an overall assessment of eutrophic condition in these frameworks, we conducted an analysis of eutrophication based on measurements of response indicators prevalent in SCB estuaries to develop a regional estimate of extent and magnitude of eutrophication in the SCB. Furthermore, we examine the sensitivity of the results to changes in threshold selection, data format, and data integration. A eutrophic condition category was assigned to each segment for each indicator, generating a set of category assignments for each estuary segment. Details of how monitoring data were used to calculate final segment categorization are given by indicator below.

Macroalgal Abundance We applied a modification of the UK-WFD element for macroalgae based on a combination of biomass and cover (Scanlan et al. 2007). Thresholds for wet weight biomass were converted to dry weight utilizing the median dry: wet weight ratio of all ulvoid biomass samples. These thresholds were used to determine condition using the average biomass and cover scores from the two consecutive periods of highest biomass and cover (Peak Season) (Fig. 2a), although other data integration options were explored (Table 2).

Phytoplankton Biomass Phytoplankton assessment as an indicator were available from ASSETS (Bricker et al. 2003) and from the IFREMER (Souchu et al. 2000; Zaldivar et al. 2008) (Fig. 2b). We chose to apply the IFREMER framework because French Mediterranean lagoons were expected to be similar to SCB estuaries. Thresholds were applied to the annual average of chlorophyll *a* data, although other data integration options were explored (Table 2).

Dissolved Oxygen Concentration Dissolved oxygen assessment frameworks from ASSETS and the UK-WFD are based on adverse effects of hypoxia on benthic and demersal fauna (Fig. 2c). However, the proposed framework for the UK-WFD (Best et al. 2007) incorporates the effect of salinity on oxygen solubility by shifting the threshold based on the measured salinity. We opted to use a hybrid of the two approaches by applying the UK-WFD thresholds to the tenth percentile of annual data as prescribed by ASSETS (90 % of data exceeds DO thresholds; Fig. 2c), although other data integration options were explored (Table 2). The UK-WFD thresholds are comparable to those generated independently for susceptible California fish and invertebrate species (Sutula et al. 2012), but use of the fifth percentile was designed for use in well-ventilated waters, and we measured DO in bottom waters that were often stratified. Because the ASSETS thresholds are applied to the tenth percentile of bottom water DO, a hybrid approach was considered appropriate in this case.

Results

Extent and Magnitude of Eutrophication in SCB Estuaries

Macroalgal Abundance Seventy-eight percent of segments had moderate or worse eutrophic condition with respect to macroalgae based on application of the UK-WFD macroalgal indicator applied to peak season biomass and cover (Fig. 3a). Estuaries in the "moderate" category (37 %) had cover between 25 % and 50 % and biomass less than 70 g dw m^{-2} (Fig. 3b). Peak season average biomass and cover ranged from 0 g dw m^{-2} and 0 % cover to 295 g dw m^{-2} and 65 % cover (site with highest biomass, DL) and 91 g dw m^{-2} and 93 % cover (site with highest cover, GS). Eutrophic condition at each segment was also categorized for each sampling period, and the number of consecutive periods any segment spent in each eutrophic condition category was variable from site to site (Fig. 3c). Some segments had chronically high biomass and cover and were classified as moderate or worse for more than 80 % of the year (e.g., DL, GS, UCL, MLM, BL), whereas other sites had episodic blooms measured during a single period (e.g., ZC, SJC, SBF). However, for most segments, the overall score was driven by more than one period of moderate or worse eutrophic condition.

Fig. 2 Assessment frameworks used for evaluating eutrophic condition in SCB segments

a UK-WFD numeric thresholds for assessing eutrophic condition using macroalgae biomass and cover (adapted from Scanlan et al. 2007).

Biomass (g ww m^{-2})	Biomass (g dw m^{-2}) (converted)	Percent Cover				
		≤5%	> 5%	> 15%	> 25%	> 75%
> 3000	> 415	Moderate	Poor	Bad	Bad	Bad
1000 - 3000	140 - 415	Moderate	Moderate	Poor	Bad	Bad
500- 1000	70 - 140	Good	Moderate	Moderate	Poor	Poor
100 - 500	15 - 70	High	Good	Good	Moderate	Poor
< 100	≤ 15	High	Good	Good	Moderate	Moderate

b IFREMER (Souchu et al. 2000) and ASSETS (Bricker et al. 2003) numeric thresholds for assessing eutrophic condition using phytoplankton biomass (as suspended chlorophyll a).

EU-WFD: IFREMER		ASSETS	
Mean Annual Chl a Conc.	Eutrophic Condition Category	90th percentile of Annual Chl a Conc.	Range Definition
0 - 5 μg Chla L^{-1}	High	0 - 5μg Chla l^{-1}	Low Levels
5 - 7 μg Chla L^{-1}	Good	-	-
7 - 10 μg Chla L^{-1}	Moderate	5 - 20 μg Chla l^{-1}	Medium Levels
10 - 30 μg Chla L^{-1}	Poor	20 - 60 μg Chla l^{-1}	High Levels
> 30 μg Chla L^{-1}	Bad	>60 μg Chla l^{-1}	Hypereutrophic

c UK-WFD (Best et al. 2007) and ASSETS (Bricker et al. 2003) numeric thresholds for assessing eutrophic condition using dissolved oxygen concentration.

Framework	Proposed Eutrophic Status	Freshwater	Marine	Rational
EU-WFD	High	≥ 7 mg L^{-1}	≥ 5.7 mg L^{-1}	Supports all life stages of salmonids and transitional fish
	Good	5 - 7 mg L^{-1}	4.0 - 5.7 mg L^{-1}	Supports presence of salmonids and transitional fish
	Moderate	3 - 5 mg L^{-1}	2.4 - 4.0 mg L^{-1}	Supports most life stages of non-salmonid adults
	Poor	2 - 3 mg L^{-1}	1.6 - 2.4 mg L^{-1}	Supports non-salmonid presence, poor survival salmonids
	Bad	< 2 mg L^{-1}	< 1.6 mg L^{-1}	No salmonids, marginal survival of resident species
ASSETS	High	> 5 mg L^{-1}		Protective of fish and invertebrate species
	Biologically Stressful	2 - 5 mg L^{-1}		Field and laboratory observations show stress response in invertebrates and fish
	Hypoxia	0 - 2 mg L^{-1}		Significantly reduced benthic macroinfauna, epifauna and demersal species

EU-WFD scores are "High" = nearly undisturbed conditions; "Good" = slight change in composition and/or biomass; "Moderate"= moderate change in composition and/or biomass; "Poor" = major change in biological communities; "Bad" = severe change in biological communities.
Moderate or worse status is considered actionable for the EU-WFD; estuaries must have a good or high condition.
We imposed the EU-WFD color scheme on ASSETS threshold ranges to compare results between the two frameworks.

Fifteen segment sites (55 %) had macroalgae biomass and cover indicative of moderate or worse eutrophic condition for two or more consecutive periods (>8 weeks). Thirty percent of segments had two or more periods of poor/bad eutrophic condition, and 11 % of the segments had two or more consecutive periods of bad eutrophic condition. Thirty-seven percent of segments had moderate or worse biomass for 12 or more weeks (three or more consecutive periods), and 26 % had moderate or worse biomass for longer than 20 weeks (five or more consecutive periods). Three sites (11 %) had continuous coverage of moderate or worse eutrophic condition throughout the sample year, and one of these sites was continuously in a poor or bad condition (UCL).

Phytoplankton Biomass Annual average chlorophyll *a* concentrations ranged from 0.5 to 42 μg L^{-1}. Of the three biological response indicators, phytoplankton biomass had the fewest number of segments categorized in moderate or worse eutrophic condition (39 %; Fig. 4a). Eleven percent of segments scored in the "moderate" category. SCR and SJC had little macroalgae (categorized as "very high eutrophic condition" based on macroalgal abundance) but high chlorophyll *a* (categorized as "bad" for SCR and "poor" for SJC).

Daily eutrophic condition was also categorized for each segment to determine duration of phytoplankton bloom events. One third of segments spent less than 10 % of the time in a eutrophic condition category of moderate or worse (90 % of the time in eutrophic condition of good or very high according to the IFREMER thresholds). Some segments had chronically high chlorophyll *a* defined by having moderate or worse condition for over half of the year (e.g., SCR, SDM, SJC, SDR, MLM), whereas other sites have more episodic blooms lasting only a few weeks (e.g., BW, SBM, SBF, BCM, BCF, SEL, MB). All segments categorized with moderate or worse eutrophic condition overall had daily average chlorophyll *a* concentrations greater than 7 μg L^{-1} for more than 25 % of the year. With respect to bloom duration, 40 % of segments had continuous phytoplankton blooms greater than 7 and 10 μg L^{-1} for longer than 1 month (Fig. 4b). Twenty-five percent of segments had chlorophyll *a* greater than 7 μg L^{-1}, and 22 % greater than 10 μg L^{-1}, for longer than 2 months. Fifteen percent of segments had chlorophyll *a* greater than 30 μg L^{-1} for 1 month and 8 % for 2 months. Some systems with annual average chlorophyll *a* indicative of "good" or "high" eutrophic condition, had concentrations above 7 μg L^{-1} for periods less than 1 month in duration.

Five segments had data sets that were less than 80 % complete due to logistical issues and probe failures (MB, SDR, SAR, SJC, SBM). However, data gaps for all but two of the systems were spread evenly throughout the year and do not likely impact the condition category. For two of the sites (MB and SDR), sondes were not deployed until April, so the data gap is largely at the beginning of the data set. Phytoplankton blooms in SCB estuaries were typically in late spring/early summer; thus, the data gap for these two sites may have resulted in a lower condition category than if the sondes had been deployed for the full period.

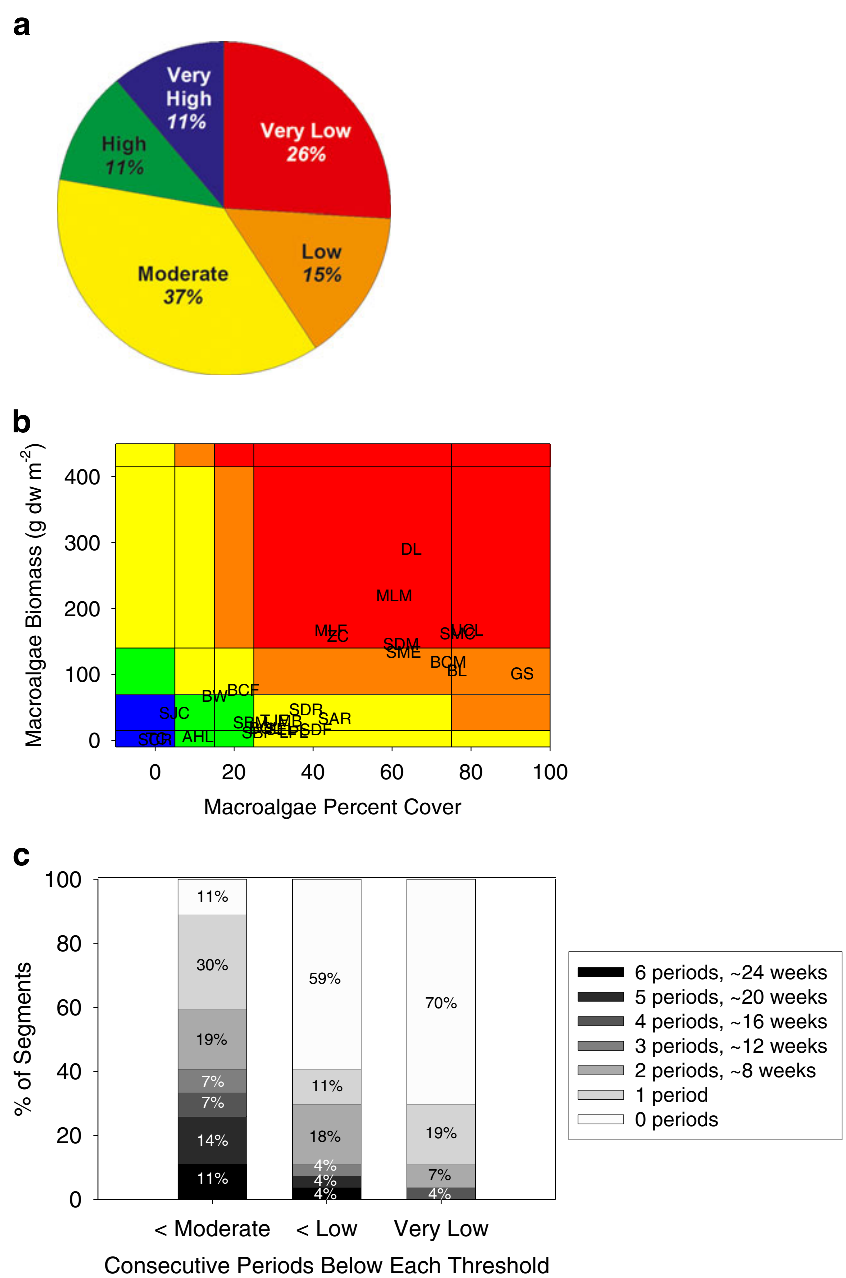

Fig. 3 **a** Percent of segments falling into each eutrophic condition category based on macroalgae biomass and cover, **b** macroalgae biomass and cover values for each SCB segment, **c** duration of macroalgae bloom for each segment

Dissolved Oxygen For dissolved oxygen (DO), tenth percentile concentrations over the 9-month period of January–October 2009 ranged from 0 to 7 mg L^{-1}. Sixty-one percent of segments fell into a eutrophic condition category of moderate or worse using the UK-WFD thresholds (Fig. 5a), and of these systems, half were categorized with bad eutrophic condition (36 % of all segments). Thirty-nine percent of segments had good or high eutrophic condition protective of adult salmonid survival, and 11 % of segments fell in the high condition category, which protects all life stages of salmonids.

Eutrophic condition at each segment was also categorized continuously for each segment using the hourly running average of 15-min DO concentrations to determine the duration of hypoxic events. The percentage of time any segment spent in each eutrophic condition category was variable (Fig. 3c). Most segments fall into the moderate or worse condition for a portion of the diel cycle (night), and the overall percentage of time in a moderate or worse eutrophic condition as well as the tenth percentile value is reflective of many consecutive nights of low dissolved oxygen concentration rather than a single continuous period of low DO. However, for some segments, continuous low DO events exceeded diel cycles (e.g., SCR, DL, GS, UCL). All segments had some period less than the moderate threshold of 4 mg L^{-1}, and 82 % of sites spend some time below the poor and bad thresholds (2.4 and 1.6 mg L^{-1}, respectively), though for 29 % of segments, the longest continuous period less than the moderate and poor thresholds was 12 h or less (Fig. 5b). For longer duration events, 35 % of segments had DO less than 4 and 2.4 mg L^{-1} for longer than 5 days. Twenty-eight percent of SCB segments had DO continuously less than 1.6 mg L^{-1} for longer than 5 days and 14 % for longer than 10 days (GS, SCR, SDR, UCL).

As with phytoplankton, data gaps in the continuous DO dataset were a concern for some of the segments, particularly MB and SDR, due to the delay in deployment described above. For most segments, hypoxia occurs in late spring/early summer; thus, the data gap for these two sites could result in a lower eutrophic condition category than would have been achieved had the sondes been deployed for the full time.

Multiple Lines of Evidence The three indicators of biological response to eutrophication did not necessarily agree in many SCB segments (Table 3). All but one segment (96 % of segments) were assigned a eutrophic condition class of moderate or worse based on at least one indicator. This percentage drops to 63 % if any two indicators are considered and to 53 % if the two indicators must include one of the primary producers and dissolved oxygen concentration [a primary and secondary indicator as prescribed by the ASSETS framework (Bricker et al. 2003)]. Fifteen percent of segments (MLM, SDR, SMR, UCL) fell in a category of moderate or worse in all three indicators.

Segments in a Regional Context We ranked segments from highest eutrophic condition to lowest (Table 3). The five highest ranked estuary segments in the SCB were: BQL, SBF, LPL, BCF, and MLF. The five lowest ranked segments were: MLM, SDR, UCL, SCR, and DL. However, in a number of segments, indicators often gave conflicting results. For example, SCR and SJC ranked among the segments with highest eutrophic condition for macroalgae but among the lowest for phytoplankton and low overall. Similarly, some sites ranked among the lowest condition based on macroalgae and highest for phytoplankton (BCM, BL). However, rankings based on phytoplankton are somewhat arbitrary because estuaries ranked 1 through 15 are all in a high eutrophic condition category. There are also two sites that rank among the lowest eutrophic condition for both primary producers but among the highest for DO (MLF, SMC), though most sites that ranked as low eutrophic condition for DO were also ranked as low eutrophic condition for one or both primary producers.

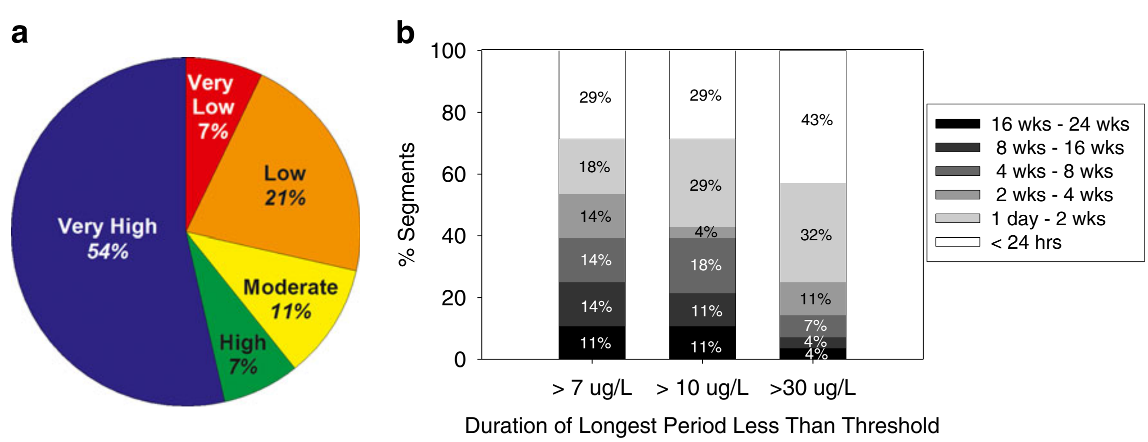

Fig. 4 **a** Percent of segments falling into each eutrophic condition category based on chlorophyll *a*; **b** duration of phytoplankton bloom events in SCB segments

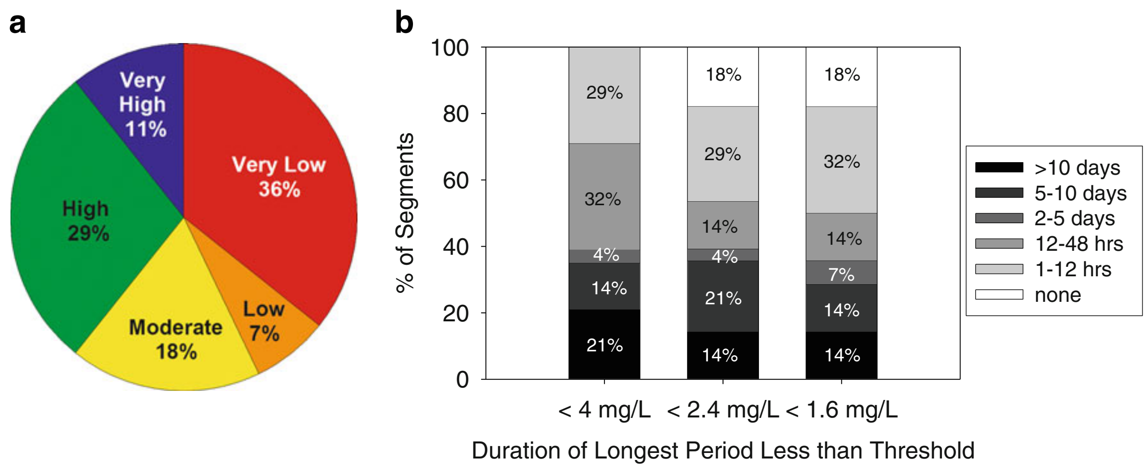

Fig. 5 **a** Percent of segments in each condition category based on DO, **b** duration of hypoxic events in SCB segments

Sensitivity of Results to Threshold, Data Format, and Spatial and Temporal Integration

A number of segments were on the borderline of thresholds that would place them in a different category, making them prone to reclassification given a different data management regime or slight change in threshold. This would be particularly important for segments in the "moderate" category, since the threshold between good and moderate drives management action according to the EU-WFD. To investigate the sensitivity of eutrophic condition category to changes in data format, data integration, and threshold selection, we compared the results from the assessment as described above to results generated using a different data management options and thresholds from the ASSETS framework (Table 4).

For macroalgae, assessment outcome was sensitive to data format, and spatial and temporal integration of the data (Table 4). Use of wet weight versus dry weight had a significant effect on condition category, with 19 % segments "improving" a eutrophic condition category and 11 % "declining." Categorizing a segment based on the transect with the highest biomass and cover (rather than the average of the three transects) during the period of maximum biomass decreased the eutrophic condition category of 44 % of segments, and for 11 %, the condition category changed by two or more. Similarly, if an annual average of all segment data is used, 41 % segments increase in eutrophic condition category and one decreases. If the average of all transect data during the period of highest biomass and cover (maximum period) is used, 7 % segments increase in condition category and 22 % decrease. Use of an annual average of segment values generated from an average of the three transects results in the maximum number of segments being in the highest possible eutrophic condition category, although 56 % segments will not change. Use of the single period of highest biomass and cover (maximum period) generated from the transect with highest biomass and cover (maximum transect) results in the lowest possible condition category for each segment, although 37 % segments will not change.

With respect to phytoplankton biomass, results were sensitive to temporal integration of the data as well as thresholds (Table 4). The difference between using daily averages versus the instantaneous data set did not have a large effect on the outcome. However, there was an effect of using discrete data versus continuous data: 22 % of systems increased in eutrophic condition category and 19 % decreased when discrete data were used compared with continuous daily averages. Changing the data integration period also has a significant effect on the outcome. Using a 75th or 90th percentile instead of the annual average resulted in 22 % and 44 % of segments changing eutrophic condition class, respectively. Using ASSETS as described (thresholds applied to 90th percentile of annual data), 7 % of segments increased in condition category and 30 % decreased in category relative to IFREMER thresholds applied to annually averaged data.

Dissolved oxygen assessments were sensitive to changes in temporal integration and assessment framework (Table 4). Use of the fifth percentile resulted in eight segments (30 %) scoring lower, whereas use of the 15th percentile resulted in 22 % of segments scoring higher in eutrophic condition than the tenth percentile. Applying ASSETS thresholds to the tenth percentile of continuous data resulted in category change in 78 % of segments, with roughly equal numbers of segments increasing and decreasing in condition category. Changing the data format from an hourly running average to instantaneous generally had no effect, with the exception of the ASSETS framework, in which 37 % of segments changed class when the hourly running average was used.

Table 3 Ranks of segments

overall rank		Segment	Annual TN Load Rank	Segment Rank by Indicator		
				Peak Season Macroalgal Abundance	Annual Mean Phytoplankton Biomass	10th Percentile Dissolved Oxygen
Highest Condition ------------> Lowest Condition	1	Batiquitos Lagoon (BQL)	16	7	10	3
	2	Seal Beach (SBF)	9	5	12	4
	3	Los Penasquitos Lagoon (LPL)	11	6	7	10
	4	Bolsa Chica (BCF)	4	15	11	5
	5	Mugu Lagoon (MLF)	24	18	19	1
	6	San Diego Bay (SDF)	22	10	15	7
	7	Santa Ana R. Wetlands-Diked (SAR)	18	14	2	9
	8	Seal Beach- Diked (SBM)	8	9	8	14
	9	San Elijo Lagoon (SEL)	17	8	5	17
	10	San Mateo Lagoon (SMC)	7	24	18	2
	11	Topanga Lagoon (TC)	1	2	13	16
	12	Tijuana River Estuary (TJE)	27	11	1	19
	13	San Diego Bay- Diked (SDM)	23	22	24	6
	14	Agua Hedionda Lagoon (AHL)	15	3	20	11
	15	Ballona Lagoon- Diked (BL)	19	19	3	13
	16	Santa Margarita Estuary (SME)	21	20	21	12
	17	San Juan Creek (SJC)	12	4	25	8
	18	Zuma Canyon Lagoon (ZC)	2	17	14	18
	19	Bolsa Chica- Diked (BCM)	5	21	4	15
	20	Ballona Wetlands (BW)	20	13	6	26
	21	Mission Bay (MB)	13	12	9	27
	22	Goleta Slough (GS)	10	23	17	23
	23	Mugu Lagoon – Diked (MLM)	25	26	26	20
	24	San Diego River (SDR)	14	16	23	24
	25	UCSB Campus Lagoon (UCL)	3	25	22	22
	26	Santa Clara River (SCR)	26	1	27	21
	27	Devereux Lagoon (DL)	6	27	16	25

Annual total nitrogen load is ranked from 1 (lowest load) to 27 (highest load). Indicators are ranked from 1 (best) to 27 (lowest eutrophic condition). Overall rank is determined from the average of dissolved oxygen and the lowest ranked algal group. The colors represent the EU-WFD condition class: high (blue), good (green), moderate (yellow), poor (orange), and bad (red)

Discussion

Extent and Magnitude of Eutrophication in SCB Estuaries

Regional Condition Based on the EU-WFD framework, eutrophication was found to be pervasive in SCB estuarine segments during the 2008–2009 water year (having an EU-WFD eutrophic condition category of moderate or worse) regardless of whether indicators were applied individually (78 % based on macroalgae, 39 % for phytoplankton, and 63 % for DO), or as a part of a multi-metric approach (53 % based on one primary producer and DO). In terms of multi-metric strategies, the EU-WFD applies a “one out, all out” approach in determining eutrophic status wherein the lowest score for any single element becomes the overall score for the state of the waterbody (Borja et al. 2004; Zaldivar et al. 2008). Applying this, all but one of 27 segments assessed would require management action to improve eutrophic condition. However, several studies have demonstrated the shortcomings of using a single indicator to establish eutrophic condition, showing that multiple metrics provide a more robust accounting of condition (Borja et al. 2009a, b, 2011b; Borja and

Table 4 Changes in eutrophic condition class due to data format, framework, or data integration

Indicator and comparator	Data format	Framework	Data management	# Segments that change condition category				
				No change	+1	+2 or more	−1	−2 or more
Macroalgae WFD framework Dry biomass peak season average/transect average	Dry biomass	EU-WFD	Annual avg/transect avg	15	9	2	1	0
		EU-WFD	PS avg/transect max	15	0	0	9	3
		EU-WFD	MP/transect avg	22	0	0	4	1
		EU-WFD	MP/max transect	10	0	0	13	4
	Wet biomass	EU-WFD	Annual avg/transect avg	14	8	4	1	0
		EU-WFD	PS avg/transect avg	19	5	0	3	0
		EU-WFD	PS avg/transect max	12	2	0	10	3
		EU-WFD	MP/transect avg	19	2	0	6	0
		EU-WFD	MP/Max transect	10	0	0	12	5
Phytoplankton IFREMER framework Annual average of daily averages	Daily averages	EU-WFD	75th percentile	22	1	0	5	0
		EU-WFD	90th percentile	16	0	0	9	3
		ASSETS	Annual avg	21	5	0	2	0
		ASSETS	75th percentile	20	5	0	2	1
		ASSETS	90th percentile	18	2	0	5	3
	Instantaneous	EU-WFD	Annual avg	27	0	0	1	0
		EU-WFD	75th percentile	23	3	0	2	0
		EU-WFD	90th percentile	15	0	0	10	3
		ASSETS	Annual avg	20	5	0	2	1
		ASSETS	75th percentile	22	2	0	4	0
		ASSETS	90th percentile	18	2	0	5	3
	Discrete	EU-WFD	Annual avg	16	2	4	3	2
		EU-WFD	75th percentile	17	2	3	0	5
		EU-WFD	90th percentile	11	1	2	5	8
		ASSETS	Annual avg	14	4	3	2	4
		ASSETS	75th percentile	18	1	3	2	3
		ASSETS	90th percentile	14	3	0	2	8
Dissolved oxygen WFD framework 10th percentile hourly running average of annual data	Hourly running average	EU-WFD	5th percentile	20	0	0	8	0
		EU-WFD	15th percentile	22	6	0	0	0
		ASSETS	5th percentile	5	9	14	0	0
		ASSETS	10th percentile	7	2	8	8	3
		ASSETS	15 percentile	18	6	0	4	0
	Instantaneous	EU-WFD	5th percentile	20	0	0	8	0
		EU-WFD	10th percentile	28	0	0	0	0
		EU-WFD	15th percentile	23	5	0	0	0
		ASSETS	5th percentile	17	0	0	9	2
		ASSETS	10th percentile	18	4	0	6	0
		ASSETS	15th percentile	18	6	0	4	0

PS peak season, *MP* maximum period, *avg* average

Rodriguez 2010). The ASSETS framework utilizes multiple lines of evidence where each indicator is scored based on intensity of expression with respect to threshold values, spatial extent, and frequency of occurrence (Bricker et al. 2003). Scores for primary symptoms (primary producer response) and secondary symptoms (DO) are combined to generate an overall score of eutrophic condition for the estuary (Bricker et al. 2003; Nobre et al. 2005). The applicable "primary symptom" would vary depending on the estuary. For example, two SCB estuaries had little macroalgae, but high suspended chlorophyll *a*, indicating that eutrophic condition in these systems is driven by phytoplankton. Furthermore, the dominant primary producer type may vary from year to year (Cloern and Nicols 1985; Nixon et al. 2001; Sousa-Dias and

Melo 2008). This underlines the importance of selecting the most critical primary producer response indicator for each system and from year to year, rather than a one-size-fits-all approach (Bricker et al. 2003; Zaldivar et al. 2008; Borja et al. 2009b). We applied the ASSETS primary and secondary symptoms strategy to this assessment, requiring that each segment score moderate or worse on either of the two primary producer indicators and dissolved oxygen. This resulted in 53 % of segments requiring management action to improve eutrophic condition to a "good" status.

Event Duration Macroalgal and phytoplankton blooms in the SCB were of sufficient duration to impact benthic and pelagic fauna in some segments. Several studies have shown negative impacts from moderate levels of macroalgae biomass on benthic communities after 8 to 20 weeks of exposure (Norkko and Bonsdorff 1996; Bolam et al. 2000; Cardoso et al. 2004; Cummins et al. 2004). Fifty-eight percent of SCB segments sampled had moderate or worse biomass for eight or more weeks, and 26 % had moderate or worse biomass for longer than 20 weeks. For phytoplankton, blooms of short duration are vital to sustain estuarine food-webs (Cloern 1996; Cloern and Jassby 2008); however, blooms lasting longer than 1 to 2 months will begin to have a negative impact on submerged aquatic vegetation, decreasing habitat diversity and impacting eutrophic condition (Moore and Wetzel 2000; Ruiz and Romero 2001). Within the SCB, 19 % of segments had continuous phytoplankton greater than 10 μg L^{-1} for longer than 2 months. For biomass greater than 30 μg L^{-1}, 15 % of segments exceeded this threshold continuously for 1 month and 7 % for 2 months. Thus, it is reasonable to assume a quarter or more SCB segments have sufficient macroalgae and/or phytoplankton bloom duration to significantly affect ecosystem health.

The length and frequency of hypoxia in SCB estuaries was also a concern. The response of aquatic organisms to low DO will depend on the intensity of hypoxia, duration of exposure, and the periodicity and frequency of exposure (Rabalais and Harper 1992). Furthermore, the length of exposure to hypoxia that is lethal or sublethal but stressful varies widely depending on the organism, from a few hours to many months (Vaquer-Sunyer and Duarte 2008). All SCB segments had a period less than the moderate threshold of 4 mg L^{-1} and 82 % of sites below 2.4 mg L^{-1}. However, for 29 % of segments, the longest continuous period less than the moderate and poor thresholds was 12 h or less, a length of time that can be endured by most organisms (Vaquer-Sunyer and Duarte 2008). Frequent hypoxic periods of short duration are typical of shallow subtidal or intertidally dominated habitats, where DO concentrations are driven by high sediment oxygen demand (Diaz et al. 1992; Rabalais et al. 1994; Sohma et al. 2008). While nightly hypoxia may not exceed the duration for lethal effects on many estuarine species, it can create chronic stress on animals, adversely affecting feeding, capacity to escape predation, reproduction, and growth. Assessment of these effects is species-specific, and these data are not available for most species native to SCB estuaries (Sutula et al. 2012). For longer-duration events, 35 % of segments had concentrations less than 4 mg L^{-1} for longer than 5 days, the median lethal time upon exposure to hypoxia (Vaquer-Sunyer and Duarte 2008). These estuaries were largely the intermittently tidal bar-built lagoons and river mouths, systems known for the occurrence of natural hypoxia in bottom waters due to trapping of salty, dense water when the sandbar inlet closes (Largier et al. 1991, 1996). Because our assessment only measured DO at one location in bottom waters, it is not clear the extent to which our assessment adequately reflects adverse effects of low DO in each estuary. Conversely, many of these intermittently tidal estuaries are home to salmonids. Levels less than 6.3 mg L^{-1} are not considered protective of the reproduction and survival over longer durations (Best et al. 2007; Sutula et al. 2012), so application of the threshold of 4 mg L^{-1} to designate management action may underestimate the need for species-specific management for these systems.

Comparison to Other Regions The observed predominance of moderate to hyper-eutrophic condition in SCB estuaries is similar to other regional or national studies of U.S. estuaries, as well as in Europe and Australia. Widespread coastal eutrophication has been reported for estuaries in the United States (NEEA), but prior to this study, the status of southern California estuaries was largely unknown (Bricker et al. 1999, 2008). The majority of NEEA estuaries showed signs of eutrophication, with 65 % displaying at least one symptom and 78 % of assessed estuarine area falling into moderate or worse eutrophic condition conditions (Bricker et al. 1999, 2008). Several studies of eutrophication throughout the European and Australian coastlines have found that symptoms of eutrophication were found to inhibit the eutrophic condition of estuaries in half or more of estuaries assessed (Hillman et al. 1990; Ærtebjerg et al. 2001; Borja et al. 2004, 2009a; Ferreira et al. 2007).

In the SCB, eutrophic condition was not exclusively related to nutrient loading. The NEEA study which showed that estuaries with high overall eutrophic conditions were generally those that received the greatest watershed nutrient loads (Bricker et al. 1999). Nutrient loads into SCB estuaries were highly variable (Table 1). To explore the relationship between nutrient loads and estuarine response, we ranked estuaries in terms of nutrient loads and compared these to the ranks derived from the expression of eutrophication indicators (Table 3). Well flushed sites with high nutrient loads were typically ranked with higher eutrophic condition than their muted counter parts (MLF versus MLM, SDF versus SDM, SBF versus SBM, BCF versus BCM), though all sites with relatively high nutrient loads scored moderate or worse in at least one indicator. Some sites with lower nutrient loading had worse ranks for eutrophication indicators than sites with higher nutrient loads if the segments had restricted circulation and high residence times (DL, UCL, ZC). Low tidal flushing and high water residence times have been identified as factors which increase susceptibility to eutrophication (Painting et al. 2007), which might explain the difference in the rankings of nutrient loading versus expression of eutrophication in these estuaries. This underscores the need to incorporate site-specific factors in assessment of nutrient pressure (as is the case in the ASSETS framework; Bricker et al. 2003).

Uncertainty in the Assessment and Applicability of Existing Frameworks

The confidence with which managers will pursue remediation of a problem is dependent on the level of uncertainty in the assessment. Our data can be used to discuss the applicability of existing assessment frameworks and provide insights into improvements and adaptations to suit a wide range of estuaries. Uncertainty in the assessment can arise from several factors: (1) appropriateness of indicators, (2) how well the assessment captured the temporal and spatial variability, (3) applicability of assessment framework thresholds, and (4) how the data were used to categorize estuaries. These uncertainties are explored below.

Adequacy of Indicators Many studies support the use of macroalgae, phytoplankton, and DO as indicators to assess eutrophication (Bricker et al. 2003; Zaldivar et al. 2008) and our experience supports this consensus. Macroalgae was the dominant aquatic primary producer in SCB estuaries (Fong and Zedler 1993; Fong et al. 1994; Kennison et al. 2003; McLaughlin et al. 2012) and, as an indicator, was applicable in most segments. However, in some systems, phytoplankton was the dominant primary producer. Thus, assessing both macroalgae and phytoplankton is important for shallow subtidally dominated estuaries because it is difficult to predict how estuary primary producers will respond to nutrient loading, and response may vary from year to year (Short et al. 1995; Flindt et al. 1999). Phytoplankton biomass and DO are most applicable in estuaries dominated by sub-tidal habitat and are less relevant in estuaries dominated by intertidal habitat. Twenty-six percent of SCB estuaries have more intertidal area than subtidal area (Table 1, seven segments: BW, LPL, MLF, MLM, SEL, SJC, TJE). Therefore, although DO and phytoplankton thresholds were applied to all segments, they may not be relevant in all. However, clear guidance for when phytoplankton or DO should no longer be applied is generally not available and research to support development of guidance is limited. Other indicators have been proposed for assessment of eutrophication, including sediment nutrient or organic carbon content, toxic blooms, and others (Bricker et al. 2003; Zaldivar et al. 2008). The relevance of these other indicators is worth exploring as is the relative priority and weight each should be given for making assessments.

Uncertainty from Temporal and Spatial Variability Time and space matter when monitoring indicators for assessment of extent and magnitude of eutrophication. As noted above, primary producer dominance can vary season to season and year to year (Cloern and Nicols 1985; Nixon et al. 2001, Nobre et al. 2005; Sousa-Dias and Melo 2008) and can be spatially heterogeneous within an estuary (Kamer et al. 2001; Kennison et al. 2003; Nobre et al. 2005). Furthermore, DO is also expected to be spatially and temporally variable (Diaz et al. 1992; Rabalais et al. 1994; Nezlin et al. 2009; Vaquer-Sunyer and Duarte 2011); for example, back channel habitats and mudflats, which have an abundance of organic carbon and high sediment oxygen demand, would be expected to have lower DO than other areas in the same estuary. Furthermore, nutrient loading into estuaries can differ dramatically from year to year (Nixon et al. 2001; Kemp et al. 2005; Gilbert 2010), and this inter-annual variability will greatly affect expression of eutrophication symptoms (Pinckney et al. 2001).

Our study adequately captured seasonal variability of changes in primary producer groups and DO. The timing of sampling has been shown to be critical to the assessment of overall eutrophic condition (Nobre et al. 2005), with one-time site visits unable to capture the variable nature of expression of eutrophication (Nelson et al. 2005). However, we did not capture inter-annual changes in the indicators of eutrophication. The proposed EU-WFD framework recommends monitoring be conducted in at least three out of the 5-year reporting cycle (Best et al. 2007; Scanlan et al. 2007; Zaldivar et al. 2008). The 2008–2009 water year was relatively dry, and thus, nitrogen and phosphorus loading to SCB estuaries was in the 16th percentile of a 13-year estimate of nutrient loads (Howard et al. 2012), so we expect that our data are representative of below average conditions, and more research is needed to adequately capture extent and magnitude of eutrophication inter-annually in the SCB.

Spatial heterogeneity is also characteristic of estuaries, so our use of a targeted index area introduces uncertainty in our ability to report on extent of eutrophication for each estuary individually. However, in roughly half of the estuaries, the segment represents 75 % or more of the total estuarine area because SCB estuaries are typically small (<40 ha). Spatial variability within the segment was better accounted with macroalgae, which relied on data from three transects, distributed throughout the segment. However, phytoplankton and dissolved oxygen were only monitored at a single location.

Another area of concern in terms of accounting for spatial variability in monitoring are floating mats of macroalgae. Our macroalgae monitoring protocols worked well in sites with large intertidal flats but were less effective for assessment of floating mats typically found in estuaries with "closed" tidal inlets (26 % of SCB estuaries). These estuaries have little to no intertidal area (due to closure of the ocean inlet), and thus, the floating algal mats do not become entrained on intertidal flats but are distributed, often unevenly, throughout open water area. Our protocol involved returning to a fixed transect, which may result in over- or under-estimation of biomass in these systems as wind can push the algae to one side of an estuary (e.g., ZC, SMC). Further work is required to develop effective protocols to capture areal extent and biomass of floating macroalgal mats in systems with a closed tidal inlet, typical of Mediterranean estuaries. In addition, the representativeness of intertidal protocols to assess the effects of macroalgae in seagrass-dominated estuaries is questionable (21 % of SCB estuaries). In these estuaries, the assessment of macroalgal effects is better made in the seagrass beds, though costs are higher and logistics of sampling more complicated (Huntington and Boyer 2008).

Adequacy of Assessment Frameworks Authors of the ASSETS (Bricker et al. 2003) and EU WFD frameworks (Scanlan and Wilson 1999; Souchu et al. 2000; Best et al. 2007; Zaldivar et al. 2008) have recognized that the lack of data on ecosystem response to nutrient over enrichment as well as on reference condition may mean that the applicability of indicators to specific habitat types, the thresholds, and how event duration and frequency are incorporated, are likely to change over time as the body of literature grows (Patricio et al. 2007; Scanlan et al. 2007; Domingues et al. 2008). One objective of this study was to inform this debate by discussing to what degree these frameworks were applicable to SCB estuaries and the associated uncertainties in their application.

For SCB estuaries, macroalgae is a key indicator for extent and magnitude of eutrophication. The Scanlan et al. (2007) macroalgal assessment framework accounts for both the abundance (biomass) and spatial patchiness (cover) inherent in this indicator, although the adequacy of thresholds between categories is worthy of further discussion. Results of a recent study of two California estuaries by Green et al. (2013) in Bodega Bay and Newport Bay show significant impacts on benthic invertebrates at 110–120 g dw m^{-2} and 100 % cover after 4 weeks of constant biomass. Similarly, Bona (2006) showed an effect threshold on benthic habitat quality at biomass levels greater than 700 g ww m^{-2} (~90 g dw m^{-2}) and >70 % cover. Therefore, an "effects" threshold in the range of 70–120 g dw m^{-2} [as proposed by Scanlan et al. (2007)] seems reasonable. Similarly, Green et al. (2013) quantified a "natural background" abundance of macroalgal biomass in the range of 2–16 g dw m^{-2}, similar to the range of very high (0–10 g dw m^{-2}) established by best professional judgment in the EU WFD (Scanlan et al. 2007). At what areal percent cover this threshold is applied is another question. Diversity and biomass of epifauna were shown to increase with biomass until macroalgae covered 50 % of the benthos (Pihl et al. 1996, 1999). Jones and Pinn (2006) found that after a month of approximately 75 % macroalgal cover, all species in the sediment declined and many organisms started migrating out of the sediment and moving into the mats. However, biomass was not monitored in these studies. Scanlan et al. (2007) use cover categories of <5 %, 5–15 %, 15–25 %, 25–75 %, and >75 %. In the SCB, placement of ten segments (37 %) in the moderate eutrophic condition was driven by cover between 25 % and 50 % with biomass less than 70 g dw m^{-2}. Placement of these segments in an "actionable" category may be overly conservative. It may be advisable to consider refinement (collapsing 5–25 %) and an additional cover category between 25 and 75 %.

ASSETS (Bricker et al. 2003) and IFREMER (Souchu et al. 2000) thresholds are based on the paradigm of light limitation of benthic primary producers, particularly seagrass, although references to other adverse effects are made. However, this paradigm is not necessarily relevant in all systems, and discussion on different thresholds for systems without seagrass is worth considering. In the SCB, 26 % of estuaries had seagrass habitat and 36 % had brackish water submerged aquatic vegetation. Both ASSETS and IFREMER assessment frameworks have similar "no effect" levels of chlorophyll *a*—less than 5 and 7 μg L^{-1}, respectively. Moderate effects range from roughly 7 to 10 μg L^{-1}, similar to the criteria established for Yaquina Bay in Oregon, 3–5 μg L^{-1} (Brown et al. 2007) and Florida estuaries, <3.8–11.0 μg L^{-1} (Janicki et al. 2000, 2009). Above 20 μg L^{-1}, submerged aquatic vegetation show declines (Stevenson et al. 1993), and phytoplankton community shifts from diverse mixture to monoculture (Twilley 1985). At 60 μg L^{-1}, chlorophyll *a*, high turbidity, and low bottom water dissolved oxygen have been observed in estuaries (Jaworski 1981; Bricker et al. 2003). Estuaries with closed inlets are typically brackish and can become dominated by cyanobacteria under high nutrient loading (Paerl 2008); for these estuaries, studies of the relationships between chlorophyll *a* and cyanobacteria blooms can be illustrative (Walker 1985; TetraTech 2006). Cyanobacteria blooms less likely to occur when summer mean chlorophyll *a* concentrations are less than 5 μg L^{-1} while, at concentrations of 10 μg L^{-1}, such blooms are still rare. These values are comparable to no effect levels in seagrass dominated habitats as described by ASSETS and the IFREMER. Similarly, concentrations of 20 μg L^{-1} suggests cyanobacteria blooms will occur about 15–20 % of the time, which has been suggested to be the maximum allowable level consistent with full support of contact recreation use (Walker 1985). Thus, while there are a few studies that provide a clear picture of biomass dose versus eutrophic response for phytoplankton, there appears to be some scientific consensus around ranges of thresholds (Borja et al. 2011a).

We found that the use of the WDF framework for DO (Best et al. 2007) versus ASSETS thresholds has an effect on the results of the assessment. Relative to the Best et al. (2007) framework, 78 % of segments upgraded eutrophic condition category using ASSETS, with 40 % changing two or more condition categories. We feel the use of the EU-WFD framework in SCB estuaries was well-founded. The thresholds proposed by Best et al. (2007) are similar to those calculated for California species (5.7 mg L^{-1} as chronic effects criteria protective of 95 % of the non-salmonid population and 2.8 mg L^{-1} as acute effects criteria; Sutula et al. 2012). For salmonids, Sutula et al. (2012) calculated 6.3 mg L^{-1} as chronic effects criteria and 4.0 mg L^{-1} as acute effects criteria. Relative to ASSETS, the WFD framework has the advantage of reconciling a threshold protective of all life history stages for salmonids from 7 mg L^{-1} in freshwater to 5.7 mg L^{-1} at marine salinities. The ASSETS upper threshold of 5.0 mg L^{-1} is roughly equivalent to this threshold at full strength seawater but does not take into account effects of salinity (Bricker et al. 2003), an issue in estuaries. Thus, applying ASSETS to river mouth and lagoonal estuaries with a closed inlet, habitats that are typically brackish and that currently or historically support salmonids in southern California, could be under-protective. In addition, Sheldon and Alber (2010) revealed some confusion in the literature over the definition of hypoxia, often cited as <2 mg L^{-1}, but the units used to describe oxygen concentrations cited criterion for hypoxia of 2 mL L^{-1} (Diaz and R. Rosenberg 1995) is actually equivalent to approximately 2.8 mg L^{-1}, which is the acute criteria calculated by Sutula et al. (2012).

Uncertainties in How Data Are Used to Make an Assessment We found that categorization of estuarine eutrophic condition was sensitive to the format of the data as well as the spatial and temporal integration of the data (Nobre et al. 2005). Most expert discussion of assessment frameworks tends to focus on the thresholds, with less attention paid to specifying the spatial and temporal density of data and how to use it to make an assessment. Data format and integration were found to impact the condition categories of estuaries for each indicator, regardless of whether the ASSETS or EU-WFD frameworks were applied.

How macroalgal abundance data were used to categorize estuaries had a significant effect on condition category. Data management decisions for macroalgae include whether to use wet or dry biomass, whether to use the mean biomass from the three transects, the maximum biomass, or a percentile, and the time period of data integration. Macroalgae biomass was measured in terms of both wet and dry weights. The EU-WFD uses thresholds based on wet weights for practical reasons (Scanlan et al. 2007), although, recent work has argued for use of dry weights (Patricio et al. 2007). We observed that wet weights and dry weights were not necessarily linearly related, with significant scatter particularly for higher biomass samples (R^2=0.691, p<0.0001, least-squares regression). Thus, we felt that dry biomass was a more scientifically defensible approach to assessment of eutrophication to eliminate the error involved in variable water content. Using our preferred data integration period, eight segments changed category when wet versus dry biomass was used, indicating that this is an important consideration for assessment. Management of spatial data also had an effect on condition categorization. We utilized average biomass and cover from all three transects to weigh intertidal area in the segment equally and generate a condition class representative of the entire segment rather than the most severely affected sub-section. Use of a percentile or only using biomass and cover data from the worst of the three transects generated lower scores in half of the segments, demonstrating the importance of variation in spatial scales in assessment. More research is required to identify the amount of estuarine area that can be affected by significant and sustained algal blooms before the ecosystem is significantly degraded. Finally, how temporal data are integrated also affects how estuaries are categorized. As expected, more segments were categorized as having higher eutrophic condition using annual averages compared with peak season or maximum period; what was interesting was the differences between peak season and the maximum period. For some sites, a maximum period with high biomass and cover was averaged with a period of relatively low biomass and cover resulting in a moderate condition category. This approach defined the difference between sites with chronic problems and those with short-duration blooms.

Method of data collection and type of averaging applied to the phytoplankton biomass data set had an impact on the condition categories of a significant number of segments. Half of the systems changed condition category when the discrete data were used versus the continuous data, and 25 % of the systems crossed the good/moderate boundary indicating a change in whether management action would be taken. This is not surprising, given that phytoplankton biomass in estuaries is highly variable on tidal, weekly, and seasonal time scales (Day 1989), so continuous data will always be preferred over discrete grab samples, albeit not always practical. It is worth noting that our discrete data set was insufficient for use in the IFREMER (which requires monthly data collection at minimum). Continuous data could be expressed as either instantaneous 15-min data or as daily averages. We opted to use daily averages to eliminate some high-frequency noise in the data set. However, comparison between the two data sets indicated that there was not a significant effect on how data were categorized with respect to eutrophic condition.

Data management considerations were also important for determining eutrophic condition based on DO. Both ASSETS and EU-WFD (Bricker et al. 2003; Best et al. 2007) utilize a percentile approach to data integration, calculated by ranking data from lowest to highest value, and applying the percentile. The EU-WFD applies a fifth percentile and ASSETS a tenth percentile; the fifth percentile of 9 months of continuous DO data equates to approximately 2 weeks below a designated threshold. Use of fifth and 15th percentile relative to the tenth percentiles changes condition classes in 20–30 % of segments. The use of the percentile approach to integrate duration and frequency of low DO events does not distinguish between high-frequency short-duration events and low-frequency but long-duration events. The effect of these two examples can be very different on biota, depending the timing and number of reproductive cycles in the year, number per brood, and other factors. "Natural" hypoxia in bottom waters of bar-built estuaries (Rabalais et al. 2010) is potentially an issue for application of DO thresholds and has implications for interpretation this assessment. Shallow estuaries are prone to development of density-driven stratification during restrictions or closure to tidal exchange when the estuaries precluding diffusion and mixing of oxygen to bottom waters (Largier et al. 1991, 1996). All of the estuaries that were closed to tidal exchange in this assessment were typified by hypoxic events greater than 1.5 days in duration, with some of the more eutrophic estuaries having hypoxic events up to 36 days. Studies of natural hypoxia in minimally disturbed "reference" estuaries are needed to clarify this issue.

Conclusions

This study has demonstrated that eutrophication is prevalent in SCB estuaries region-wide, regardless of which indicator or set of indicators are used to make the assessment. However, on an individual estuary level, assessment of eutrophic condition is dependent on the indicator(s) used as well as the spatial and temporal scales over which the monitoring is conducted, a finding supported by similar comparisons conducted in European estuaries (Devlin et al. 2011; Garmendia et al. 2012). In this study, we aimed to understand the best way to utilize three indicators of eutrophication in the Mediterranean estuaries of the SCB. Managers who plan to make assessments in similar systems should consider evaluation of assessment frameworks, particularly with respect to: (1) applicability of specific indicators may vary from estuary to estuary; (2) timing of expression of eutrophication is highly variable across estuaries; and (3) effect of sampling and data management approach on assessment outcome.

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