Regional assessment of marine and estuarine sediment toxicity in southern California

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Abstract

Sediment toxicity was investigated at 222 stations in the Southern California Bight (SCB) during 2008. This represented the first time that assessment methods established by California's new Sediment Quality Objectives (SQO) program were employed in a survey of this scale. The goal was to determine the extent and magnitude of sediment toxicity in the SCB, how toxicity compared among specific environments, and whether toxicity has changed over the last decade. Two toxicity tests were used: the 10 d amphipod whole sediment survival test with *Eohaustorius estuarius* and a 48 hr embryo development test with the mussel *Mytilus galloprovincialis* exposed at the sediment-water interface. Less than 1% of the area of the SCB was found to be toxic to the amphipod test. No toxicity was found at offshore stations, but 14% of embayment areas were toxic to the amphipods. Estuary and marina locations had the greatest areal extent of toxicity for both tests. The two toxicity methods agreed that sediments were not toxic at over half of the stations tested. The mussel test showed a greater magnitude of response than the amphipod. Sediment toxicity was shown to have declined in both extent and magnitude from levels measured in 1998 and 2003.

Introduction

More than $35 million (US) is spent monitoring the ocean ecosystems of the Southern California Bight (SCB) every year, but little sediment toxicity monitoring is conducted to assess impacts of pollutants. A number of individual programs have studied selected areas of the Bight. The Bay Protection and Toxic Cleanup Program and the National Oceanic and Atmospheric Administration's National Status and Trends Program found many toxic sites in embayments of the SCB (Fairey *et al*. 1998, Anderson *et al*. 2001). Other programs, such as US Geological Survey's BEST and US Environmental Protection Agency EMAP programs have also found sediment toxicity in the SCB (USGS 2000). While these programs identified toxic hot spots, they used a variety of methodologies that make spatial and temporal comparisons difficult.

The State of California recently adopted Sediment Quality Objective (SQO) for the protection of benthic communities in bays and estuaries (SWRCB 2008). This establishes a framework useful in evaluating the status of specific sites (Bay and Weisberg 2009). The SQO enables the consolidation of many surveys from a single agency or a single survey from many agencies into an overarching assessment of status for the entire region. However,

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this has yet to be done in the SCB where more than 4 billion liters per day of treated wastewater and more than 100 billion liters of untreated runoff is discharged every year (Lyon and Stein 2009).

The SQO uses a multiple line of evidence approach to assess sediment quality (SWRCB 2008). The SQO program incorporates three lines of evidence (chemistry, biological assemblage and sediment toxicity) and distinct approaches for data interpretation (Bay and Weisberg 2009). The toxicity line of evidence provides both a measure of exposure and response, helping to assure that SQO-based impairments are focused on the impacts from pollutants. In order to gain confidence in the outcome, SQOs require the use of two toxicity methods at each station. The use of a dual-species approach on a large scale project is daunting, necessitating the use of multiple laboratories to provide capacity. The logistical and quality assurance challenges are immense because the data must be comparable both within and among laboratories.

The objectives of this study were to answer three questions: 1) What is the extent and magnitude of sediment toxicity in the SCB? 2) How does the extent and magnitude of sediment toxicity compare among specific habitats? 3) How do the sediment toxicity results compare to previous regional surveys of the Bight? To answer these questions, a probabilistic regional survey of sediment toxicity was conducted at 222 sites throughout the SCB, distributed among five different environments. Stations from previous surveys were resampled to make temporal comparisons.

Methods

Sampling

Between July 1 and September 30, 2008, 222 stations were sampled between Point Conception, California and the United States/Mexico border (Figure 1). Thirty stations were on the continental shelf with the remainder being in embayments. A Generalized Random Tessellated Stratified design (Stevens 1997) was used to create a spatially balanced random sampling plan in five environments (strata): mainland continental shelf, marinas, ports, bays and estuaries. This design was intensified for sampling in targeted areas and by resampling stations from previous surveys. Intensified sampling was

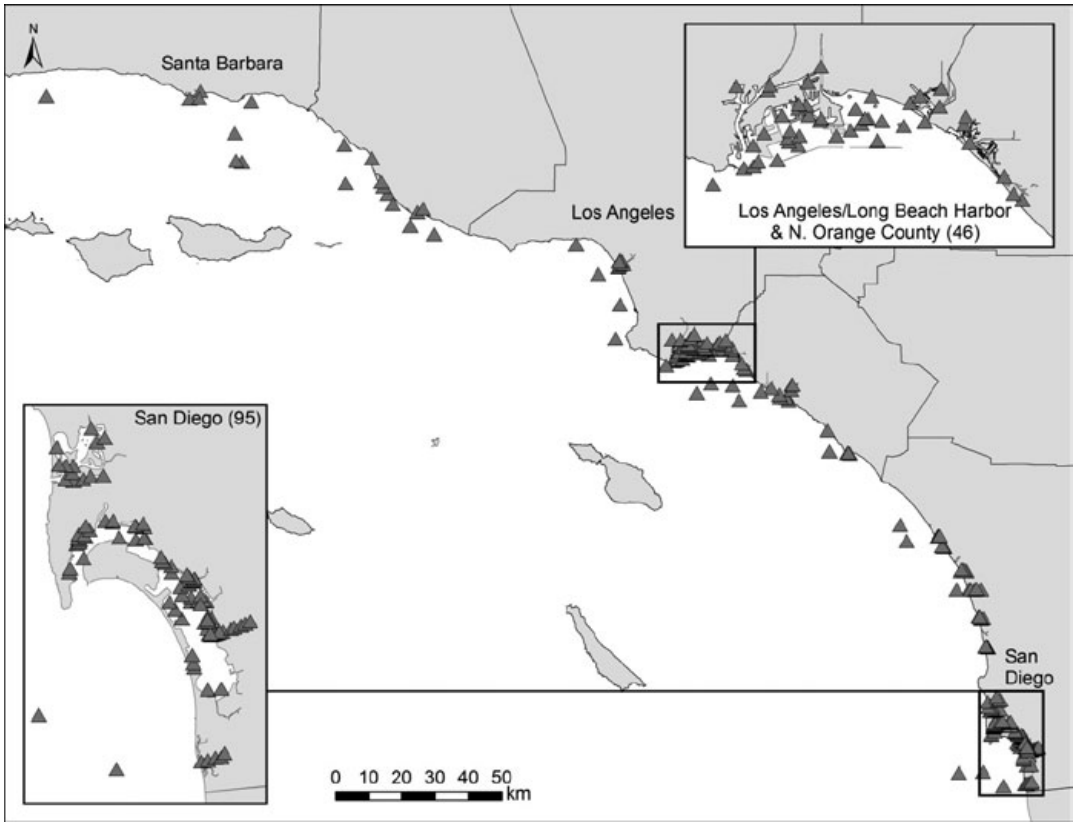

Figure 1. Bight'08 stations that were sampled for sediment toxicity. Values within parenthesis are the number stations in the region pictured.

performed in the San Diego Bay region by increasing inclusion probabilities in that area. To assess temporal trends, 25% of the samples were collected from stations previously sampled in 1998 and 25% from 2003.

Sediment samples were collected with a 0.1 m^2 modified Van Veen grab. A plastic (high-density polyethylene [HDPE], polycarbonate, or Teflon) scoop was used to collect sediment from the top 2 cm from offshore stations and top 5 cm for embayment stations of the undisturbed surface material in the grab. The difference in sample depth between locations is due to the increased sample volume needed to conduct two toxicity tests in the embayments. Contact with sediment within 1 cm of the sides of the grab was avoided in order to minimize contamination. The sediment was placed in clean HDPE containers and distributed to the testing laboratories. Once collected, the samples were stored in the dark at 4° C in the laboratory for no longer than four weeks prior to testing.

Toxicity Testing

The toxicity of whole sediment to amphipods was determined using a 10-d survival test (USEPA 1994, ASTM 2002) with *E. estuarius* (EE) under static conditions. Amphipods and control sediment were collected from a non-contaminated estuarine site (Beaver Creek, OR) by Northwestern Aquatic Sciences (Newport, OR). The amphipods were held under laboratory conditions for 2 to10 d prior to test initiation. Testing was conducted in 1 L glass containers. Sediment samples were sieved through a 2 mm mesh screen and homogenized in the laboratory before testing. Sediment samples were added to the test containers to a depth of 2 cm. Filtered (≤ 20 µm) seawater (32 psu salinity) was added slowly to a final volume of 800 ml. Continuous aeration was provided. Sediments were allowed to equilibrate overnight under these conditions before addition of the amphipods. For each sample, exposure consisted of five replicates, along with two surrogate containers for water quality (initial and final dissolved oxygen, pH, total ammonia and salinity measurements of overlying water and pore water). A negative control (amphipod collection site sediment) was included with each batch of samples tested.

Twenty amphipods were randomly added to each replicate at the start of the exposure. Tests were conducted at 15 ± 2 °C under constant illumination. At the end of the exposure period, the sediment was screened through a 0.5 mm mesh sieve and the number of surviving amphipods was recorded. The mean control survival for each test batch had to be at least 90% for the data to be considered acceptable. The control between replicate coefficient of variation also had to be less than 12% to be acceptable.

Embryos of the mussel *M. galloprovincialis* (MG) were exposed at the sediment-water interface following the methodology of USEPA (1995) and Anderson *et al.* (1996). The MG testing was only performed on the 192 embayment stations. There was no MG testing of the shelf since the SQO program for which the dual-species design was employed does not apply to the offshore environment. Sediment was added to 600 ml glass chambers to a depth of 5 cm. Sediment was passed through a 2 cm sieve and homogenized before addition. Approximately 300 ml of filtered (≤1 µm) seawater (32 psu salinity) was carefully added over the sediment. The overlying water was gently aerated and exposure chambers maintained at 15 ± 2 ºC with a 16 hour light: 8 hour dark cycle. The sediment was allowed to equilibrate overnight before addition of a screen tube. The screen tubes were made of polycarbonate tubing with a 25 to 30 µm mesh polyethylene screen (Anderson *et al*. 1996). A negative control consisting of the exposure container and screen tube, but no sediment, was tested with each batch to verify the test system was not causing toxicity. A second control with 10 ml of laboratory water in glass shell vials was tested to verify organism health.

Approximately 250 fertilized mussel eggs from a stock solution were added to the screen tube to initiate the bioassay. The same volume of embryo stock was also added to five replicate glass vials for determination of the initial number of embryos. Water quality parameters (dissolved oxygen, salinity, pH, and ammonia) were measured on the overlying water at the beginning and end of the exposure period. After 48 hours, the embryos were washed from the screen tube into another vessel for fixing and storage. The embryos were then counted and examined for normal development under a microscope. The percentage of normal embryos relative to the initial number of embryos determines the endpoint, which is termed percent normal-alive (PNA). The mean control PNA had to be ≥70% for each test batch to be considered acceptable.

Toxicity testing was conducted by seven laboratories: Aquatic Bioassay and Consulting Laboratories (Ventura, CA), City of Los Angeles (Playa del Rey,

CA), City of San Diego, Nautilus Environmental (San Diego, CA), Orange County Sanitation District (Fountain Valley, CA), UC Davis Department of Environmental Toxicology, Marine Pollution Studies Laboratory (Monterey, CA), and Weston Solutions (Carlsbad, CA).

Quality Assurance

Extensive quality assurance procedures were instituted to ensure data comparability among the laboratories. Split samples were sent to the laboratories before the start of field sampling. Laboratories had to demonstrate adequate comparability in order to participate in the study. Results of the QA exercise were judged on four categories: magnitude of test response, toxicity category identification, intralaboratory variability, and reference toxicant response. Each of these categories had a scoring system and the combined score was used to judge overall comparability. Details of the scoring system can be found in Bay *et al*. (2011). Additional split samples were sent to the laboratories during the sampling period to ensure results remained comparable and to judge the comparability of one laboratory that was unable to participate in the earlier QA exercise.

The quality assurance exercise before the sampling period found all of the participating laboratories had comparable results for the amphipod test. Substantial differences were found between the laboratories for the mussel embryo test. Test procedures were reviewed, adjustments were made and the exercise repeated. Results of the second mussel embryo exercise found all participating laboratories that were assigned to test samples during the survey had comparable results. Analysis of split samples within the testing period found all the laboratories to be comparable for both the amphipod and mussel embryo tests.

All of the 222 test stations for EE analysis passed test acceptability criteria. However, for the MG analysis there were three test batches that did not meet acceptability criteria. Data for these batches were excluded, leading to missing data for 12 stations. There were 180 successfully tested stations using both the EE and MG tests.

Data Analysis

All test response data was normalized relative to the controls within each test batch. Normalization is the mean test response at a given station divided by the mean control response. Control normalization facilitates comparisons of stations tested at various times throughout the study and by different laboratories. Statistical difference between samples and appropriate controls was determined by a T-test ($p \leq 0.05$), assuming unequal variance as calculated in Microsoft Excel.

The level of toxicity associated with each station was calculated using thresholds established for the SQO program (Bay *et al*. 2009). Each of the two toxicity methods tested had its own set of thresholds (Table SI-1 in Supplemental Information (SI)). The thresholds were used to classify stations as Nontoxic, Low Toxicity, Moderate Toxicity, or High Toxicity for each of the test methods. Each toxicity category was scored with Nontoxic being one, Low Toxicity two, Moderate Toxicity three, and High Toxicity four. The integrated toxicity assessment for the station was calculated by averaging the category score for each method and rounding up if the average fell between two categories.

To simplify description of results the terms "not toxic" and "toxic" are used when making general spatial and temporal comparisons. The term not toxic refers to stations or areas classified as either in the Nontoxic or Low Toxicity categories. The Low Toxicity category was considered to be not toxic because the biological significance and reliability of this category is uncertain. The response in the Low Toxicity category is of low magnitude and may not be greater than test variability. The term toxic refers to samples classified as either Moderate Toxicity or High Toxicity. Use of the terms toxic and not toxic also conforms with previous studies. Results for all four categories are also presented so the results may be compared to other studies where the SQO assessment methods are used.

Analysis of the toxicity data used design-based inference procedures to provide unbiased estimates of area weighted proportions and areal extent (e.g., the number of square kilometers in a toxicity category for a particular stratum). These probability-based areal estimates take into account the relative area each sample site represents. Specifically, the estimates are a weighted average where the weights are determined by the size of each disjoint sampling area divided by the number of samples falling into that area. These "area weights" are the same as the inverse of the inclusion probabilities for that particular sample. The area weighted proportions were computed as a ratio of the sum of the area weights for all sites

for a particular toxicity category and the sum of the area weights for the entire subpopulation or stratum. The areal extent was computed by multiplying the area-weighted proportion by the size of the subpopulation. The local neighborhood variance estimator, which takes advantage of any spatial proximity with the data set, was used to compute standard errors for constructing 95% confidence limits (Stevens and Olsen 2003). Prior to any statistical computation, area weights were adjusted to account for missing data. Area weights for stations within a stratum having missing data were not included in the analysis, resulting in reduced total area being evaluated for toxicity.

Temporal comparisons were made for EE toxicity data. The SQO thresholds were used assign toxicity categories for the toxicity data collected in 1998 and 2003 to facilitate comparisons with the current study. The stations were then designated as not toxic or toxic as previously described.

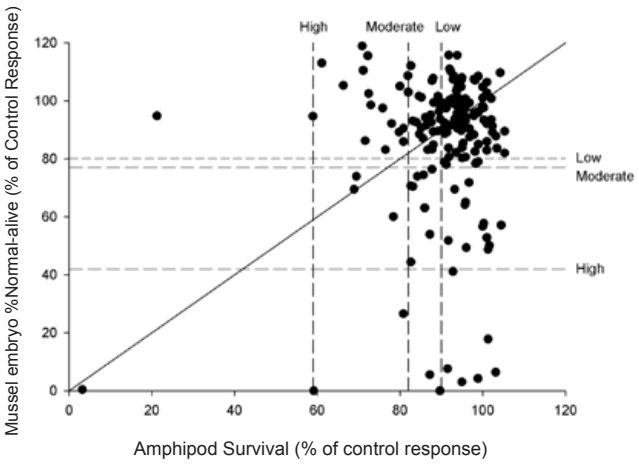

Figure 2. Comparison of magnitude of toxicity between the *Eohaustorius estuarius* (EE) and *Mytilus galloprovincialis* tests for stations where both tests were performed.

Results

Comparison of Toxicity Methods

There were 180 stations where a direct comparison between the EE and MG test methods could be made. Many of the stations had a similar magnitude of response for each test method (Figure 2). There were also a similar number of stations where one test showed greater response than the other. Where there was the greatest difference between the two tests, the MG test was found most often to have the larger response. There was agreement between the tests as to whether stations were toxic or not for 137 (76%) stations. Of these, the tests agreed that 131 stations were not toxic. There was agreement on the SQO category for 94 (52%) of the 180 stations tested with both methods. Both tests found 87 of those 94 stations to be in the Nontoxic category. Of the stations where there was disagreement, more than half indicated the EE test to be in a higher toxicity category (e.g., EE was Low Toxicity while MG was Nontoxic). Where the disagreement was the greatest (i.e. High versus Nontoxic) the MG test indicated a higher toxicity category more times (5) than did EE (1).

Areal Extent of Toxicity

Less than 1% of the SCB's 3884 km^2 was found to be toxic (Moderate or High Toxicity categories) to EE (Table 1). The shelf stratum had no area designated as toxic. The 3758 km^2 in the shelf stratum accounted for the vast majority of the total area (97%) causing its data to drive the EE results for

Table 1. Estimated percentage of area of SCB sediment classified by toxicity category using the amphipod survival test.

Stratum	Nontoxic		Low Toxicity		Moderate Toxicity		High Toxicity	
	Estimate	95%CI	Estimate	95%CI	Estimate	95%CI	Estimate	95%CI
Bay	43.4	15.9	40.4	16.0	16.2	11.9	0.0	-
Marina	61.8	16.8	25.2	15.6	8.6	10.3	4.3	7.4
Port	80.3	12.1	14.2	11.2	5.4	8.3	0.0	-
Estuary	54.9	16.0	23.7	14.4	18.5	13.1	2.8	4.6
Shelf	77.0	14.8	23.0	14.8	0.0	-	0.0	-

Total Areas (km^2): Bay 70.0, Marina 16.8, Port 28.0, Estuary 11.1, and Shelf 3758.3.

the Bight as a whole. Within the embayments, the estuary stratum had the greatest percentage of the area found to be toxic at 21%, followed by bay, marina and port strata, with 16, 13 and 5% respectively. The total percentage of the embayment area found to be toxic by EE was 14%.

The areal results for the MG test were similar to EE for the embayments as a whole, but differed within the individual strata. The total area tested using MG was much smaller than for EE at 124 km^2 (Table 2). The total of the embayments found to be toxic was 13%. The estuary and marina strata had a similar percentage area (29 and 28%) where toxicity was observed. The bay and port strata each had 9% of their areas identified as toxic.

The integrated toxicity results provide the most accurate assessment of the extent of toxicity within the embayments (Table 3). The integrated area designated as toxic for embayments was 13.8 km^2 or about 11% of the area. The marina and estuary strata had the highest percentage of area toxic with 24 and 22% respectively.

Temporal Changes in Toxicity

There has been a considerable decrease in the percentage area classified as toxic (Moderate and High Toxicity categories) over time (Figure 3). The largest decline was from 2003 to 2008 when the total Bight percentage of toxic area dropped from 17.5 to 0.4%. The percentage of toxic area was similar or lower for 1998 compared to 2003 for all strata. The offshore stratum had a similar percent area toxic in 1998 and 2003 (16.1 and 17.0% respectively), while there was no toxicity for 2008. The percentage area identified as toxic decreased by 50% or more in each of the embayment strata between 2003 and 2008. The sum of the percentage of area for the Low, Moderate and High SQO categories is similar across all of the surveys for each of the strata. This pattern indicates that the observed changes over time represent a transitioning from the higher toxicity categories to the Low Toxicity category; the percentage of area in the Nontoxic category has changed little over time.

In order to verify temporal trends, comparisons were also made by analyzing data from stations that were sampled in one of the previous surveys and again in 2008. This included 36 stations sampled in 1998 and 55 stations from 2003. Most of the stations from 1998 or 2003 had no change in toxicity in 2008. Given the similarity in results, data from both periods have been combined. The majority (74%) of stations did not change toxicity categories between surveys (Figure 4). Stations that did change went mostly from toxic to not toxic (18 of 24; 75%).

Discussion

The results showed the total area of SCB with toxic sediment to be small. The offshore sediments were not toxic. Within the embayments, especially the marina and estuary strata, both the prevalence and magnitude of acute toxicity was greater compared to offshore locations. The overall temporal trend is of declining sediment toxicity, both in the percentage of area toxic and the magnitude of toxicity.

Most national studies have used only a single species to assess sediment toxicity. Results from the SCB embayments are within the toxicity range found in these studies. A NOAA study using the amphipod *Ampelisca abdita* found a wide range of results in embayments throughout the country, with between 0 and 85% of the area within individual water bodies

Table 2. Estimated percentage of area of SCB sediment classified by toxicity category using the sediment-water interface test with mussel embryos.

Stratum	Nontoxic		Low Toxicity		Moderate Toxicity		High Toxicity	
	Estimate	95%CI	Estimate	95%CI	Estimate	95%CI	Estimate	95%CI
Bay	82.6	10.9	8.7	8.2	8.8	8.0	0.0	-
Marina	72.3	13.1	0.0	-	21.6	12.2	6.1	8.0
Port	87.2	9.0	3.7	3.6	8.7	8.7	0.4	0.7
Estuary	66.9	14.5	3.9	5.1	12.6	12.0	16.6	13.2

See Table 1 footnote for total areas.

Table 3. Estimated percentage of area of SCB sediment classified by toxicity category using integrated results.

Stratum	Nontoxic		Low Toxicity		Moderate Toxicity		High Toxicity	
	Estimate	95%CI	Estimate	95%CI	Estimate	95%CI	Estimate	95%CI
Bay	40.5	16.2	50.8	15.6	8.7	8.0	0.0	-
Marina	50.2	17.9	25.4	14.0	19.8	14.0	4.6	7.8
Port	74.6	12.4	19.6	11.4	5.8	8.0	0.0	-
Estuary	41.6	15.5	36.7	15.1	13.9	11.8	7.8	9.3

See Table 1 footnote for total areas.

exhibiting toxicity (Long 2000). The percentage of toxic area for all areas of that study was 7%, about half of the area found for SCB embayments. A USEPA study evaluated sediment quality in several national coastal areas (USEPA 2008). Results of this study found the percentage of area rated poor for sediment toxicity to be similar to SCB embayments for the West Coast and Gulf coast at 17 and 13%, respectively. The Northeast Coast, Southeast Coast, Alaska and Hawaii all had less than 5% of their areas rated as poor. The EPA's Environmental Monitoring and Assessment Program in 1999 and 2005 found northern California bays and estuaries to have 17% of the area toxic to EE (Barnett *et al*. 2007). This is similar to 14% for southern California embayments found in the current study.

A more direct comparison can be made to the San Francisco's Regional Monitoring Program (RMP) where the toxicity methods used are the same as those used herein. The toxicity observed in SCB embayments appeared to be less prevalent than in San Francisco Bay. The 2008 RMP monitoring indicated that 70% of the stations had substantial toxicity to either EE or MG (SFEI 2010), compared to 28% of SCB embayment stations. A difference in responsiveness of the two test methods was also observed between the regions. While the amphipod test was more responsive in southern California (i.e.

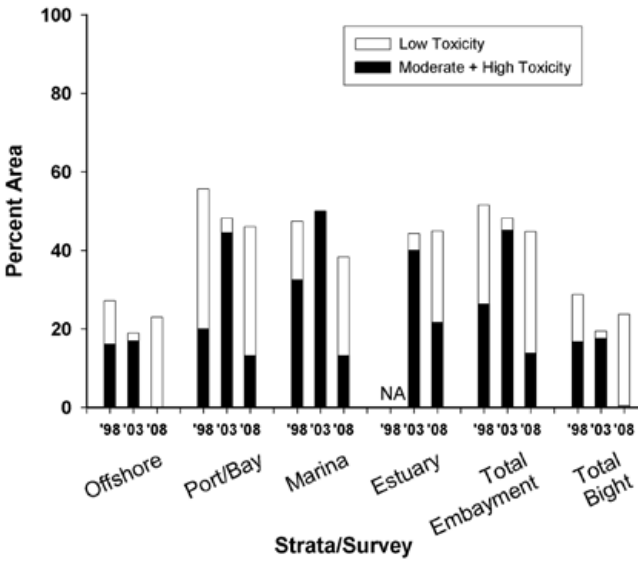

Figure 3. Comparison of percentage areas found to be toxic (Moderate + High Toxicity) with the *Eohaustorius estuarius* survival test by stratum over multiple Southern California Bight regional monitoring surveys.

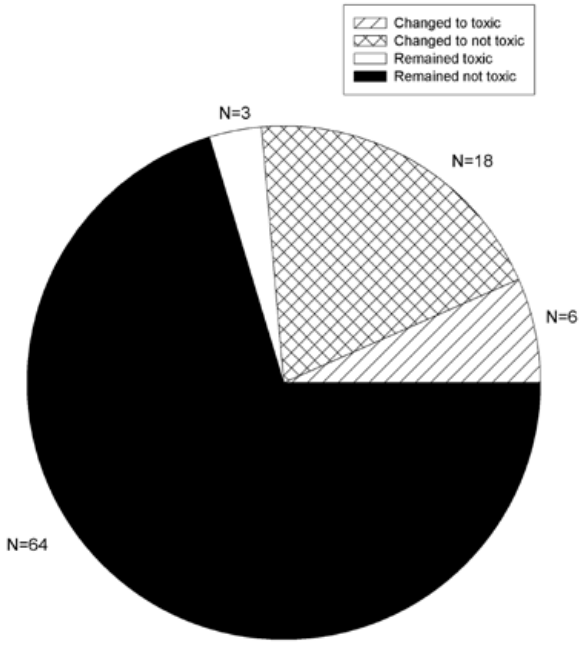

Figure 4. Changes in sediment toxicity to *Eohaustorius estuarius* for stations sampled in 1998 or 2003 and again in the 2008 Southern California Bight regional monitoring surveys. Stations defined as toxic are in the California Sediment Quality Objective Moderate and High Toxicity categories.

more stations in the Low, Medium and High categories), MG detected toxicity more frequently in the RMP. This may indicate a difference in the causes of toxicity between the two regions.

The testing with two toxicity methods at the embayment stations for the current study allowed for a greater degree of confidence in the results. While the two tests agreed on the toxicity level at more than half the stations, there were 43 stations where there was disagreement. No one toxicity test provides maximum sensitivity to the myriad of contaminants that may be encountered in the environment. An example of this is the EE test being insensitive to copper having a water LC50 of 3.7 mg/L (McPherson and Chapman 2000), whereas the MG test is quite sensitive with an LC50 of 7.8 µg/L (Phillips *et al*. 2003). Conversely, the EE test has been found to be responsive to pesticides, such as chlorpyrifos with an LC50 of 0.645 µg/L (Anderson *et al*. 2008), while MG has an EC50 of 153.5 µg/L (Beiras and Bellas 2008). This differential responsiveness increases the confidence in stations identified by both methods as Nontoxic as being truly so. Stations where there was disagreement between the methods likely indicate the cause of toxicity was a contaminant or other factor for which the methods have differential responsiveness.

The two tests used in this study also have differential susceptibility to common confounding factors. The use of two tests with differential sensitivity minimizes the chance that no data will be available from a station if one of these confounding factors is present. The SWI exposure system prevents interference from grain size for the MG test. The EE test has been found to be adversely affected by fine grained sediments (DeWitt *et al*. 1989, Tay *et al*. 1998). The MG test is approximately a factor of 10 more sensitive to ammonia than EE (USEPA 1994, Tang *et al*. 1997). In the current study, analysis of water quality data found no EE samples where ammonia would be expected to influence test results (data not shown). However, the MG test had 10 (6%) stations where ammonia likely affected the results.

The finding that toxicity is more prevalent at embayment stations is consistent with sources of contamination in the SCB. Embayments are impacted by runoff from rivers and creeks. With few exceptions, runoff is not treated in any way. In addition, the use of pesticides that find their way into the runoff continues to increase. Pyrethroids in particular have increased both in usage and prevalence in the environment. A survey of 30 urban creeks throughout California detected pyrethroid-associated toxicity to freshwater amphipods at all locations (Holmes *et al*. 2008). Pyrethroids are also quite toxic to EE (Anderson *et al*. 2008). In the current study, EE found a greater number of toxic stations in the estuary and bay strata than did MG. This may be indicative of organic chemicals, such as pesticides as the dominant cause of toxicity in southern California embayments. In some estuarine locations within the SCB, pyrethroids have been found to be at least partially the cause of toxicity (Anderson *et al*. 2010, Bay *et al*. 2010). A review of California freshwater and estuarine sediment toxicity identification evaluation data concluded pesticides were a likely cause of toxicity at all sites, mostly attributed to pyrethroids (Hunt *et al*. 2010). Conversely, the MG test identified more stations as being toxic in the marina stratum than did EE. Southern California marinas have been found to be contaminated with copper from boat hull paints (Schiff *et al*. 2004) which may be a source of this toxicity.

The trend toward decreasing toxicity over time in the offshore environment is consistent with changes in contaminant input patterns in the SCB. The offshore environment receives a large volume of waste water inputs. In spite of a trend toward increasing volume of effluent, mass emissions of contaminants have trended downward over the past few decades (Lyon and Stein 2009). Some constituents have decreased by orders of magnitude. A decrease in contaminants to the ocean would be expected to result in lower sediment concentrations and less toxicity. A previous study found that there was little toxicity is surficial sediments, but many instances of toxicity in deeper (older) sections of cores taken throughout Santa Monica Bay (Greenstein *et al*. 2003). Sediment contamination with organic chemicals was found to be highest in core sections corresponding to the 1960s -1980s and decreased thereafter (Bay *et al*. 2003). A decrease in the area contaminated by legacy compounds in surficial sediments from 1998 to 2008 has also been found (Schiff *et al*. 2011). It should be noted that each station tested for sediment toxicity in the offshore stratum represented a large area. While this leads to large confidence intervals for the area calculations (Table 3), the lack of toxicity at any offshore station in 2008 when compared to previous surveys is significant.

Measurement of sediment toxicity is only one part of the multiple line of evidence approach

necessary for assessing sediment quality in embayments for the SCB program. To provide an accurate assessment using the California SQO approach, data on chemical concentrations and benthic community condition are also necessary (SWRCB 2008). Chemistry data are needed to confirm the toxic responses are associated with chemical exposure instead of non-contaminant factors (e.g., grain size). Measures of benthic community condition are necessary to confirm toxicity results are ecologically relevant. Integrating all three lines of evidence in a sediment quality triad approach maximizes the strengths and minimizes the weaknesses of individual components (Chapman *et al*. 1997). This integration of data for the Bight'08 study will be the focus of future work.

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Supplemental Data

Supplemental Information avaialable at ftp://ftp.sccwrp.org/pub/download/DOCUMENTS/AnnualReports/2011AnnualReport/ar11_SupplementalInfo_RegionalAssessment.pdf.

Thresholds for SQO sediment toxicity categories can be found in Table SI-1. Results for the individual stations, including location information and SQO categories can be found in supplemental Table SI-2.