Fate and effects of sewage sludge in the coastal marine environment: a mesocosm experiment

Candace A. Oviatt, James G. Quinn, James T. Maughan¹, J. Taylor Ellis, Barbara K. Sullivan, Juanita N. Gearing², Patrick J. Gearing³, Carlton D. Hunt⁴, Peter A. Sampou⁵, James S. Latimer

Graduate School of Oceanography, University of Rhode Island, Kingston, Rhode Island 02882, USA

¹ Metcalf & Eddy, PO Box 4043, Woburn, Massachusetts 01888-4043, USA

² Maurice-Lamontagne Institute, PO Box 1000, 850 Route de la Mer, Mont-Joli, Quebec G5H 3Z4, Canada

³ PO Box 205, Luceville, Quebec G0K 1E0, Canada

⁴ Battelle Ocean Sciences, Ocean Science & Technology Dept, 397 Washington Street, Duxbury, Massachusetts 02332, USA

⁵ PO Box 775, Horn Point Laboratories, Cambridge, Maryland 21613, USA

ABSTRACT: A mesocosm experiment to assess the fate and effects of sewage sludge in the coastal marine environment was conducted over 4 mo during the summer of 1984 with 9 different treatments of sewage sludge and nutrient additions. Evidence from settling rate studies, accumulations of carbon and pollutant organics and ratios of $\delta^{13}\text{C}$ in the sediment indicated up to 83 % of the sludge particulate inputs settled to the bottom and up to 51 % accumulated there. While no toxicity due to organics or metals was apparent, prolonged hypoxia and even anoxia occurred in upper treatment levels. At summer temperatures, sludge particulate inputs in excess of $1 \text{ g C m}^{-2} \text{ d}^{-1}$ caused these hypoxic conditions. In contrast to low primary production in low and medium sludge treatments during the last 2 mo of the experiment, the highest sludge treatment had high production. We attribute these patterns to excessive grazing pressure from zooplankton and benthic fauna in the lower 2 treatments and hypoxia-retarded grazing in the upper treatment. In the upper treatments, peaks of production stimulated peaks of excessive respiration suggesting that respiration of sludge particulates was enhanced or co-metabolized with the production of fresh organic material.

INTRODUCTION

A large variety and number of studies have been conducted to assess the impact of sewage sludge disposal in the marine environment (Bascom et al. 1978, Bascom 1980, 1982, 1984, Geyer 1981, Ketchum et al. 1981, 1985, Mayer 1982, Myers & Harding 1983, Wolfe & O'Connor 1987). In the late 1920's little was known; for example, Rawn (1960) observed: 'Investigations of the behavior of existing outfalls along the Pacific coast failed to fill us in on desired information about the relation between depth and quantity factors, although we gave it quite a try. I recollect without much pleasure, the many hours bobbing about in rowboats in sewage fields along the southern California coast.' Today, tracer studies, water quality and sediment monitoring studies, biotic impact studies, and labora-

tory toxicity tests have been carried out to quantify inputs, assess distribution and fate, and quantify impacts on primary producers and biota. Without arguing the relative merits of any of these efforts, it is clear that neither distribution and fate, nor impacts, have been quantified to the extent that all parties can agree on where or how much sewage sludge can be safely disposed of in the coastal marine environment (Erdheim 1985). In southern California, the decision has been made to cease disposal through an outfall located in relatively deep but poorly oxygenated waters, and to dispose of sludge materials on land (Brooks 1983). In the New York Bight, the decision was made not to dispose of sewage sludge on land but to continue dumping at sea (Erdheim 1985). However, the nearshore 12 mile site is being phased out in favor of the offshore 106 mile site (Suszkowski & Santoro 1986).

Due to the dynamic nature of the marine environment and to alternative potential pollutant sources, it has been difficult to determine sewage sludge fate after disposal. Generally, it can be traced for a matter of hours in the water column and over a few kilometers on the bottom (Bascom et al. 1978). Quite often sludge 'disappears' after disposal. A recent finding at the New York 12 mile site is that sediments collected semi-annually since 1977 show no visible accumulation of sludge on the bottom, no increase in total organic carbon, and low bacterial counts for total and fecal coliform (Levine 1986); thus the author concludes that sludge is rapidly remineralized. However, it has been shown that the most affected zone is not under the sludge dump site, but 10 to 15 km west of the disposal site in a depositional environment on the margin of the Christiansen Basin (Boesch 1982, Steimle et al. 1982). Burnett & Schaeffer (1980) used stable carbon isotope ratios to locate sludge carbon in the sediments of the New York Bight. This technique also has the advantage of often enabling the detection of sludge carbon actually assimilated by organisms (Rau et al. 1981).

Due to the number and variability of sewage sludge components, it has also been difficult to detect consistent effects on biota in the field. Those field studies which have examined the effects of sewage on system response have indicated that nutrients, rather than toxicants, have the dominant effect (Pearson & Rosenberg 1978, Neilson & Cronin 1981, Mearns & Word 1982). Due to the difficulties of plankton sampling, the toxicity to copepods observed in the laboratory (at concentrations ranging from 0.03 to 0.8 % whole sludge) has never been observed in the field (Miller et al. 1987). A wide range of macrobenthic infaunal responses have been reported for sewage sludge. Several investigators report severe inhibition of the total benthic assemblage (Rowe 1971, Pearce 1972, Anger 1975). Others noted domination and large density increases by a few species (McIntyre 1977, Botom 1979, Mearns & Word 1982, Young & Young 1982, Steimle 1985) or increased density of virtually all indigenous species (Dauer & Connor 1980). Zones with presumably different sludge loading rates have been proposed to account for the wide range in macrobenthic response observed (Anger 1975, Bascom et al. 1978, Pearson & Rosenberg 1978, Dauer & Connor 1980).

Due to the large number of past and continuing efforts to assess sewage sludge disposal, it is presumptuous to think that one more effort will solve all problems. However, in the controlled mesocosm experiment described here, there has been an opportunity to quantitatively assess the fate of sewage sludge components, their effects on plankton and benthic macrofauna, and the levels of sludge addition which cause detrimental effects.

The experiment

Nine 13 m³ mesocosms at the Marine Ecosystem Research Laboratory (MERL) were used to observe the response of temperate, estuarine ecosystems to 3 sewage sludge loading rates over a 3 mo summer period (14 Jun to 29 Sep) in 1984. The lowest daily addition rate (1S) was roughly equivalent, with respect to nitrogen and phosphorous content, to the final calculated dilution at the New York 106 mile site (O'Connor et al. 1985). The medium rate (4S) was roughly equivalent to nitrogen dilution in sewage sludge after discharge at the New York 12 mile site and to sewage effluent dilution at the Hyperion (Los Angeles) 5 mile outfall (Hatcher et al. 1981). The highest rate (8S) was most similar in nitrogen to the West Point, Seattle outfall (Mearns 1981). The Joint Water Pollution Control Project effluent outfall in southern California had higher nitrogen values than the 8S treatment (Mearns 1981). For comparison of the relative effects of nutrients and toxics, 3 other mesocosms received inorganic nutrients (N, P, Si) with the nitrogen additions comparable to the amounts in 1S, 4S, and 8S sludge additions. Finally, 3 mesocosms with no additions acted as controls for the experiment.

METHODS

Mesocosms. The mesocosms are simplified models of lower Narragansett Bay (Table 1). Temperature, light, mixing, water turnover and sediments are similar to the natural system (Pilson et al. 1979, Nixon et al. 1980, Oviatt et al. 1986a). The facility design, operation and analytical methods are described in Lambert & Oviatt (1986).

Treatment additions. Sewage sludge was collected weekly from the Cranston Rhode Island Waste Treatment Facility, Cranston, Rhode Island, USA. The facility is an activated sludge plant with separate primary

Table 1. Summary of MERL mesocosm design parameters. Tank construction is fiberglass reinforced resin with a white interior to maximize sunlight reflection. A 50 cm diameter plunger with a 60 cm vertical excursion mixes the tank at a rate of 5 cycles min⁻¹ for 2 h every 6 h. Sediments are from central Narragansett Bay, Rhode Island

Tank diameter	1.83 m
Tank height	5.49 m
Water surface area	2.63 m ²
Water depth	5.00 m
Water volume	13.1 m ³
Sediment area	2.52 m ²
Sediment depth	0.37 m
Water column turnover	27 d

Table 2. Composition of Cranston sewage sludge and inputs to all treatment tanks. Additions to sewage sludge treatments reflected the week-to-week variation of the sludge batches. S: sludge treatments; N: nutrient treatments

	Sludge ^a	1S ^b	4S ^b	Inputs 8S ^b	1N ^b	4N ^b	8N ^b
Solids	2.88 ± 0.97	191	767	1530	0	0	0
Nitrogen	142 ± 32	3.5	13.9	28.4	2.9	11.5	23.0
Phosphorus	8.5 ± 4.1	0.21	0.85	1.71	0.22	0.90	1.80
Silica	2.76 ± 0.7	0.07	0.28	0.56	0.21	0.82	1.64
Carbon	1040 ± 406	26	106	214	0	0	0
PCBs ^c	0.35 ± 0.12	0.066	0.267	0.533	0	0	0
PAHs ^d	45.8 ± 19.4	8.7	35	70	0	0	0
C ₁₀ BZT ^e	180 ± 103	34.4	138	276	0	0	0
Cl-BZT ^e	7.88 ± 6.49	1.5	6	12	0	0	0
Phthalates ^f	613 ± 255	117	470	939	0	0	0
Coprostanol	2040 ± 765	390	1570	3120	0	0	0
C ₁ -BZT ^e	3840 ± 2300	734	2950	5880	0	0	0
Cd	0.2 ± 0.1	0.0049	0.0222	0.0442	0	0	0
Cu	44.4 ± 15.3	1.07	4.72	9.43	0	0	0
Pb	10.8 ± 3.7	0.258	1.15	2.29	0	0	0
Zn	64.4 ± 20.8	1.56	6.821	13.63	0	0	0
Cr	6.2 ± 3.7	0.134	0.643	1.283	0	0	0
Ni	6.5 ± 2.0	0.157	0.687	1.373	0	0	0
Mn	5.37 ± 1.76	0.128	0.565	1.127	0	0	0

^a Units from top to bottom are % solids, mmol l⁻¹, µg g⁻¹ dry weight, and ppm wet weight total metal ± standard deviation

^b Units from top to bottom are g tank⁻¹ 99d⁻¹ (assuming 1 ml = 1 g wet weight), mmol m⁻² d⁻¹, mg tank⁻¹ 99d⁻¹, and mg m⁻² d⁻¹

^c Polychlorinated biphenyls measured as Aroclor 1254

^d Sum of 16 EPA priority polycyclic aromatic hydrocarbons (mainly naphthalene, phenanthrene fluoranthene and pyrene)

^e Benzotriazoles (see text)

^f Sum of 3 phthalates (see text)

and secondary clarifiers. The sludge from both clarifiers and the grease sump goes to anaerobic digesters. The sludge intended for addition to the mesocosms was collected directly from the digesters before dewatering on Friday and held refrigerated over the weekend until used during the next week, starting on Monday.

The sludge was added daily 7 times a week for 99 d during the morning mixing cycle (Table 2). Pre-mixing of the sludge with about 200 ml of tank water (30 ‰ salinity) reduced aggregation when it was added to the mesocosms.

The 3 nutrient treatments received reagent grade inorganic nutrients (NH₄Cl, KH₂PO₄ and Na₂SiO₃) during the morning mixing period (Table 2). The amounts were approximately 1, 4 and 8 times the average areal inorganic nutrient input to Narragansett Bay from sewage effluents, and equivalent to loadings used in a previous eutrophication experiment (Oviatt et al. 1986a).

Sludge components. Percent solids, particulate organic carbon, stable carbon isotope ratio (δ¹³C)*, particulate and dissolved nitrogen, particulate and dissolved phosphate, particulate and dissolved silicate, some organic components (polychlorinated biphenyls,

polycyclic aromatic hydrocarbons, substituted benzotriazoles, phthalates, and coprostanol) and metals were analyzed on each weekly batch of sewage sludge. Detailed methods of analysis for each parameter are given in Lambert & Oviatt (1986), Oviatt et al. (1986b), and this paper.

Settling behavior. In August 1984, 3 sewage sludge settling rate experiments (Expts 1 to 3) were conducted in a tank with no sediment but with the standard MERL mixer configuration. All tank water was filtered through a 0.45 µm filter to reduce background particulate concentrations. In each experiment the mixing cycle was maintained and sludge was added 30 min after the beginning of morning mixing. Mixing continued for another 90 min. Water column sampling (6 replicates) was conducted through the mixing and non-mixing cycle for 285 min. In Expt 1 the water column was also sampled after 24 h with normal mixing cycles.

$$\delta^{13}\text{C} = \left(\frac{R_{\text{spl}}}{R_{\text{std}}} - 1 \right) 1000, \text{ where } R = \frac{^{13}\text{C}}{^{12}\text{C}} \text{ for the sample (spl)}$$

and a standard (std). Values of δ¹³C in this paper are reported against the standard PDB (belemnite from the Pedee formation in North Carolina)

In Expts 1 and 3 the sludge was mixed with seawater and shaken before being added to the tank (in the same manner that it was added to sludge treatment tanks). In Expt 2 sludge was added directly to the tank with no pre-mixing.

System measurements. System production, system respiration, C-14 production, chlorophyll *a*, total diatoms, dissolved inorganic nutrients, zooplankton biomass, benthic fauna, and particulate carbon in water and sediments were measured at various intervals over the course of the experiment as indicated in Lambert & Oviatt (1986), Oviatt et al. (1986b) and Keller et al. (1987).

System production and system respiration were measured weekly by dawn-dusk-dawn oxygen measurements. Oxygen concentrations were measured by Winkler titration. Dawn minus dusk measurements were corrected for diffusion to provide a measure of system production (Oviatt et al. 1986a). Diffusion was calculated according to the following formula:

$$D \text{ (g O}_2 \text{ m}^{-2} \text{ d}^{-1}) = f (0.01) \times [\text{O}_2(\text{mg l}^{-1}) - \text{O}_2 \text{ saturation (mg l}^{-1})] \quad (1)$$

The gas exchange coefficient (*f*) was determined experimentally with changes in temperature (P. Rocque & S. Nixon unpubl. data):

$$f = 0.276 (\text{Temp } ^\circ\text{C}) + 0.3258 \quad (2)$$

In a similar manner, dawn oxygen concentration was subtracted from dusk oxygen concentration to provide a measure of system respiration.

Primary production was also measured by C-14. For this procedure, water samples were collected from 5 depths (0.1, 0.5, 1, 2.5 and 4.5 m) in each tank, inoculated with $\text{NaH}^{14}\text{CO}_2$, and suspended in bottles at the 5 depths. A dark bottle was placed at mid-depth to correct for dark uptake. After incubation from 1000 to 1400 h, samples were filtered (1 μm) and the filters were counted in a scintillation counter. Depth-integrated production was calculated from all samples and daily rates were calculated by assuming 57 % of the daily production occurred during the incubation period (Oviatt et al. 1986a).

Biomass of phytoplankton was determined weekly by chlorophyll analysis (using 1 μm pore size filter) according to Yentsch & Menzel (1963), and Lorenzen (1966) as modified by Lambert & Oviatt (1986).

Diatom concentration was determined each week according to Hasle (1978). Pooled samples from 3 depths were preserved in Lugol's iodine solution. The sample was allowed to settle overnight and counted on an inverted microscope; most counts had a precision of 10 % coefficient of variation (C.V.).

Dissolved inorganic nutrients (NH_3 , NO_2 , NO_3 , PO_4 and SiO_4) were determined on water collected weekly

and pooled from 3 depths, using a Technicon Auto-analyzer (Lambert & Oviatt 1986).

Zooplankton were analyzed weekly from quantitative net tows (64 μm mesh), filtering about 0.20 m^3 water. Biomass (dry weight) was determined on the >150 μm fraction to reduce phytoplankton contamination. When samples were still heavily contaminated (as were 13 of 150) with chain-forming diatoms, the biomass was estimated from a regression developed from the relation between zooplankton numbers and biomass of uncontaminated samples ($r^2 = 0.778$) (Lambert & Oviatt 1986). Samples were rinsed with deionized water to remove salt, frozen, freeze-dried, and weighed to the nearest 0.1 mg. For summary data analysis, mean values for the entire experimental period were calculated from weekly values by calculating time-weighted means.

Benthic fauna samples were collected prior to treatment (11 Jun 1984), about mid-way through the experiment (18 Aug 1984), and at the end of the experiment (19 Sep 1984). For each date, 10 random sediment samples were collected with a flow-through corer (5.067 cm^2) (Frithsen et al. 1983), and sampling was to a depth of about 10 cm. The samples were sliced to yield depth fractions of 0 to 2 cm and 2 to 6 cm, and each fraction was preserved in 10 % buffered formalin containing rose bengal.

After 2 mo (to ensure animal uptake of stain and stabilization of formaldehyde dry weight), the samples were washed through a nest of sieves (0.5 and 0.3 mm) into gridded Petri dishes. Animals were identified and enumerated under a stereomicroscope. Bivalve biomass was determined by measuring shell length and applying organic carbon to length relations (Frithsen et al. 1986). Biomass for other organisms was determined by obtaining dry weights (60 $^\circ\text{C}$ to constant weight) and applying dry weight to organic carbon relations (Frithsen et al. 1986).

At the end of the experiment the sediments from each mesocosm were passed through a sieve (0.318 cm) and a fraction of the retained animals was frozen for subsequent organic component analyses. The remaining animals were preserved in 10 % buffered formalin and later sorted to remove the amphipod *Leptocheirus* sp. The amphipods were enumerated, and dry and ash weight were determined.

Particulate carbon and nitrogen was measured on sludge batches (weekly), on water column samples (weekly), and on sediment samples (at the beginning, mid-point and end of the experiment). Weighed particulate matter was combusted at high temperatures in a CHN elemental analyzer (Lambert & Oviatt 1986). Precision for carbon was about 10 % C.V. and for nitrogen, 8 % C.V.

Stable carbon isotope ratios. Samples of suspended

particulate organic carbon (POC), net plankton, benthic organisms, sediments, and sludge were analyzed for their stable carbon isotope ratio, $\delta^{13}\text{C}$. Water for POC samples was pumped from the top 1 m of the water column during mixing, filtered onto precombusted, glass fiber filters, and the filters were air-dried before storage. Plankton were size-fractionated (Nitex screens), microscopically examined for obvious detritus, then held in filtered seawater, filtered onto precombusted, glass fiber filters, and air-dried. Benthic organisms were hand-picked from sieved sediment, held in filtered seawater overnight, and either filtered whole (small organisms) or dissected before being dried. Sediments were collected with an open corer, sectioned into 1 cm depth intervals, acidified with HCl, washed, and filtered onto glass fiber filters. Sediments were ground after drying and weighed to obtain density. Sludge samples were acidified with HCl and treated as for sediment samples.

All samples were analyzed by combusting a weighed aliquot to CO_2 in a radio-frequency furnace. The purified gas was measured manometrically (for percent organic carbon) and analyzed on a Micromass 602 isotope ratio mass spectrometer. Further details of these methods can be found in Gearing et al. (1984).

The fraction of sludge present in samples was estimated using a simple mixing equation:

$$f_s = \frac{\delta_m - \delta_{ns}}{\delta_s - \delta_{ns}} \quad (3)$$

where f_s = fraction of organic carbon from sludge; δ_m = $\delta^{13}\text{C}$ of the sample containing a mixture of carbon; δ_{ns} = $\delta^{13}\text{C}$ of a similar sample receiving no sludge; δ_s = $\delta^{13}\text{C}$ of pure sludge. δ_{ns} was taken from control tank samples; δ_s was $-23.45 \pm 0.41\text{‰}$; this value is the average of 12 of the weekly sludge batches measured over the course of the experiment.

Organic components. For the analysis of organic components, samples of sewage sludge and sediment were stored in precleaned glass jars at -20°C . Water samples were filtered through precombusted glass fiber filters ($1.2\text{ }\mu\text{m}$) to separate particulate and soluble components. The volume filtered per sample was about 12 l. The filtrate was spiked with internal standards and extracted once with 300 ml of dichloromethane per 3 l in the original glass collection bottle. The organic extracts for each sample were isolated, combined and solvent-exchanged into hexane; they were concentrated in volume on a rotary evaporator under reduced pressure at $<30^\circ\text{C}$, and stored in the dark at room temperature until analysis. After drying to constant weight at room temperature, the filters for each sample were stored at -20°C . When ready for analysis, filters were shredded into a round bottom flask. Internal standards were added, followed by about 150 ml of

methanol, and the mixture was refluxed for 2 h, cooled, and decanted into a separatory funnel containing about 200 ml of distilled water. The filters were rinsed with petroleum ether and the rinse was combined with the methanol extract; the mixture in the funnel was then extracted 3 times with petroleum ether, and the extracts were combined and reduced in volume on a rotary evaporator and stored in the dark at room temperature. Samples for sludge, sediment and benthic organisms were extracted in the same manner as the filter samples.

All extracts were separated into 3 fractions by silica gel column chromatography using a method modified from Pruell & Quinn (1985). The first portion contained saturated petroleum hydrocarbons, the next portion included the 16 EPA priority polycyclic aromatic hydrocarbons (PAHs) with 2 to 6 fused rings and 2 substituted benzotriazoles (C_{10} -BZT and Cl-BZT), and the third fraction contained the phthalic acid esters butylbenzyl phthalate (BBP), diethylhexyl phthalate (DEHP), and dioctyl phthalate (DOP), coprostanol and C_1 -BZT (a third substituted benzotriazole). In addition, polychlorinated biphenyls (PCBs, measured as Aroclor 1254) were also obtained by silica gel column chromatography separation of the extract.

Each fraction was concentrated on a rotary evaporator and analyzed on Hewlett-Packard gas chromatographs with flame ionization (FID) or electron capture (ECD) detectors. Resolved peaks were quantified against internal standards. Several samples were analyzed without the addition of internal standards to assess background levels of these compounds; all proved insignificant. Peak identification by retention time was confirmed by gas chromatography-mass spectrometry. The precision of the analyses was usually 20 % or less and was measured as the relative standard deviation of replicate analyses on selected samples. The detection limits for the FID analyses were usually between 10 and 100 ng l^{-1} or 10 and 100 ng g^{-1} with respect to 10 l or 10 g samples. In the ECD analyses, the detection limits were between 0.1 and 0.5 ng l^{-1} , or ng g^{-1} . Values below these limits were reported with low confidence down to none detected (ND). All values were corrected for blanks, usually less than 20 % of the sample values.

Metals in sludge. Metal concentration in the sludge were determined from triplicate analysis of each batch added to the mesocosms. A known volume (20 ml) of sludge was weighed in a 50 ml round bottom polycarbonate centrifuge tube. Two ml of concentrated Baker Instra-Analyzed nitric acid was added. The centrifuge tubes were capped and samples autoclaved for 1 h. The digestate was then volumetrically diluted and the metal concentration determined by flame atomic absorption spectrometry (AAS) against a standard curve.

Metals in the water column. Water column metal concentrations were determined weekly from 2 controls and the sludge-amended mesocosms. Duplicate samples from each mesocosm were collected and the dissolved and particulate phases (methodological definition) separated using a pressure filtration through 0.4 μm Nuclepore filters. Dissolved metals (Cu, Ni, Cd) were concentrated using an adaptation of the Co-APDE method of Boyle & Edmond (1977) and analyzed using absorption spectrometry (AAS). Dissolved Mn was determined directly with flameless AAS (Hunt & Smith 1983). Particulate metals were digested in 5.0 ml of 2N HNO_3 and metals measured by flameless (Cu, Ni, Cd) or flame (Mn) AAS.

The average of the duplicate samples was calculated for each week. These results were used to calculate the mean and standard deviation for the mesocosms during this study.

RESULTS

Settling behavior experiments

Sewage sludge settled rapidly in the 5 m deep water column (Table 3). Within 10 min, 22 to 39 % had settled and after 30 min 40 % had settled even though the mixers were on. After 75 min of mixing, 33 to 53 % had settled. During the initial period of non-mixing, about another 9 % settled. Samples taken at the end of the non-mixing cycle were inconclusive. In Expts 1 and 2 the percent settled did not change a great deal, but Expt 3 had water column concentrations equal to initial values, suggesting total resuspension (Table 3). With the lack of mixing, there was no mechanism for resus-

pension, suggesting weighing error for Expt 3. Mean values showed a trend of increasing sedimentation through mixing and non-mixing cycles as did the 24 h sample from Expt 1. Visual observations of the sludge confirmed loss to the mesocosm floor.

Sludge coagulation behavior was generally enhanced in Expt 2 in which the sludge was not pre-mixed with seawater. Very large particles (cms in diameter) formed rapidly and many of these remained floating on the water surface up to 24 h later. Sludge pre-mixed with seawater formed small particulates which did not float.

Biogeochemical cycling of contaminants from sewage sludge

The organic contaminants were measured in each of the 14 batches of sewage sludge (14 total) and the total amount added to each of the 3 treatments was calculated (Table 2). Percent solid material in the sludge ranged from 0.61 to 3.99 %. The major polychlorinated biphenyl (PCB) in the samples was Aroclor 1254 with concentrations ranging from 0.225 to 0.601 $\mu\text{g g}^{-1}$. The total of 16 polycyclic aromatic hydrocarbons (PAHs) ranged from 17.2 to 85.8 $\mu\text{g g}^{-1}$. The major components were naphthalene, phenanthrene, fluoranthene and pyrene. Naphthalene is found in various petroleum products (e.g. gasoline and fuel oils), and the other 3 are major compounds produced in the combustion of fossil fuels and wood. The substituted benzotriazoles (C_{10} -BZT, Cl-BZT and C_1 -BZT) are synthetic organic compounds that are used as ultraviolet light absorbers for plastics and coatings. The concentration of C_{10} -BZT

Table 3. Percent sludge ($\pm$ coefficient of variation $S/\bar{X}$) settling in filtered seawater (0.45 μm) in 5 m tank with no sediment at 22° C. Amount settling was calculated from amount remaining in the water column

Time ^a		Percent settled ($\pm S/\bar{X}$)			$\bar{X}$	Laboratory expt ^c
		Expt 1	Expt 2 ^b	Expt 3		
0	Amount added	17.22 g d.wt.	16.22 g d.wt.	10.09 g d.wt.		
10 min		39 % (± 8 %)	23 % (± 19 %)	22 % (± 20 %)	28 %	
30 min		31 % (± 12 %)	43 % (± 8 %)	46 % (± 7 %)	40 %	40 %
75 min		39 % (± 8 %)	53 % (± 9 %)	33 % (± 12 %)	42 %	50 %
90 min		38 % (± 7 %)				
105 min			62 % (± 9 %)	46 % (± 14 %)	49 %	
120 min		40 % (± 5 %)				
165 min			77 % (± 13 %)	19 % (± 14 %)	53 %	
180 min		50 % (± 8 %)				50 %
285 min		50 % (± 9 %)	60 % (± 9 %)	0 % (± 12 %)	?	
6 h						50 %
24 h		85 % (± 14 %)				

^a Sludge added after 30 min into mixing cycle; mixers off after 1.5 h and on again after 5.5 h

^b More coagulation because sludge not pre-mixed with seawater

^c In the laboratory, about 40 % (dry weight) of sludge was added directly to a separatory funnel containing seawater

ranged from 50 to 388 $\mu\text{g g}^{-1}$, Cl-BZT ranged from 1.78 to 25.2 $\mu\text{g g}^{-1}$, and C₁-BZT ranged from 1090 to 9340 $\mu\text{g g}^{-1}$. The phthalic acid esters (phthalates) included benzylbutyl phthalate (BBP), diethylhexyl phthalate (DEHP) and dioctylphthalate (DOP). The total of the 3 phthalates ranged in concentration from 325 to 1190 $\mu\text{g g}^{-1}$. Coprostanol, a sterol found in the faeces of man and carnivorous animals and used as a chemical indicator of sewage contamination, ranged in concentration from 760 to 3580 $\mu\text{g g}^{-1}$.

A large percentage of the total input to the 4S and 8S treatments was measured in the sediments at the end of the experiment (Table 4). The exceptions were phthalates (5 % in the sediment) and C₁-BZT (4 % in the sediments). These components and coprostanol had higher absolute concentrations in the water column than other organic components perhaps due to their greater water solubility which is related to their *n*-octanol/water partition coefficients; for example, the value for C₁-BZT (log P = 4.6) indicates that it is relatively water soluble compared to most of the other

components measured in this study (Lopez-Avila & Hites 1980). Even if an average water column concentration is multiplied by the flushing rate of the mesocosm, we cannot account for most of these 3 components (Table 5). The low concentration in the water and sediment and high variability of individual PAHs in the sludge samples increased the uncertainty of the percentage missing of this group; the other 3 components (PCBs, C₁₀-BZT and Cl-BZT) were sequestered in the sediments (Table 5).

The amounts of organics measured in the 8S sewage sludge treatment were compared to literature values for levels causing effects in the water column and sediments (Table 6). With one exception for organics, effect levels were not attained. Concentration of total phthalate in the 8S treatment sediment reached 20 ppm in the sediment (Table 6) and DEHP was as high as 17 ppm. Perez et al. (1983) have noted a reduced flux of ammonia from the benthos when sediment phthalate (DEHP) concentrations reached 5 ppm.

The mean concentrations of trace metals in the dis-

Table 4. Organic components in sewage sludge treatment sediments. (A) Calculated as the difference between 4S and 8S sludge treatments and a non-sludge treatment, 8N, at the end of the experiment. (B) Calculated as a percentage of the organic components added in treatment

Treatment	PCBs	PAHs	C ₁₀ -BZT	Cl-BZT	Phthalates	Coprostanol	C ₁ -BZT
A. Weight in tank sediment on 19 Sep 1984 (mg)							
4S ^a	0.28	ND	106	4.6	ND	385	ND
8S ^b	0.34	20	230	8.6	43	933	225
B. % of input in tank sediment							
4S	106	ND	77	77	ND	25	ND
8S	66	29	83	72	5	30	4

ND: none detected
^a For the first cm a porosity of 75 % and a density of 2.60 g solids cm⁻³ was used, and for the second cm a porosity of 73 % and the same density was used
^b In this treatment porosity increased and density decreased (0 to 1 cm, 88 %, density 2.29 g solid cm⁻³; 1 to 2 cm, 95 %, 2.5 g solid cm⁻³; 2 to 3 cm, 78 %, 2.5 g solid cm⁻³). To compensate, the mass of this treatment was normalized to the mass of the 4S treatment so organic content was calculated for 2.56 cm instead of 2 cm. Organic content of the third cm was not measured and was assumed equal to that of the second cm. The third cm was measured in 4S and this assumption was generally true for that treatment (Oviatt et al. 1986b)

Table 5. Budget for organic components in 8S sewage sludge treatment. Units are mg

Input	PCBs 0.533	PAHs 70	C ₁₀ -BZT 276	Cl-BZT 12	Phthalates 939	Coprostanol 3120	C ₁ BZT 5880
Water column	0.01	0.23	1	ND	12	16	87
Sediment	0.35	20	230	8.6	43	933	225
Total in system	0.36	20	231	9	55	949	312
Flushed	0.05	0.09	4	ND	45	60	319
Total accounted for	0.41	20	235	9	100	1009	631
% Missing	23 %	71 %	15 %	25 %	89 %	68 %	89 %

ND: none detected

Table 6. Concentrations of organic components in the 8S sewage sludge treatment compared to literature values for concentrations having pollution effects (in parentheses)

	PCBs	PAHs	Phthalates
Water column ($\mu\text{g l}^{-1}$) ^a	0.005 (30 ^b)	0.056 (300 ^c)	4 (30 ^d)
Sediment ($\mu\text{g g}^{-1}$) ^e	0.028 (20 ^b)	3 (129 ^f)	20 (5 ^d)
^a Treatment concentrations equal soluble plus particulate and are the average of values from 3 Jul to 4 Sep 1984			
^b Totals; current criteria fail to distinguish among the 209 PCB isomers in regard to toxicity (Reed et al. 1984)			
^c Acute toxicity (EPA 1980d)			
^d Perez et al. (1983)			
^e Values from the end of the experiment, 19 Sep 1984			
^f Oviatt et al. (1982)			

solved and particulate phases were not different between the control and sludge amended systems with the exceptions of slightly lower mean dissolved Cu, slightly higher mean dissolved Ni, and a 2 to 3-fold increase in mean dissolved Mn in the sludge-amended mesocosms relative to the control systems (Table 7). In no case did the dissolved, particulate, or total metals measured exceed the lowest U.S. Environmental Protection Agency's (EPA) Saltwater Water Quality Criteria, either on a weekly basis or as a long-term mean (EPA 1980a, b, c). The implication from these water column metals data is that metal is removed to the sediments.

System response to nutrient and sewage sludge additions

Unlike controls, the higher nutrient and sludge treatments had initially high levels of system production, followed by periods of low production except for the 8S treatment which had continued high production (Fig. 1). During late summer, system production in these treatments was often lower than in control treatments. System respiration, by contrast, was nearly always higher in sludge treatments than in nutrient treatments or in controls (Fig. 1). Respiration in the 8S treatment was higher than in any other treatment or control and exceeded production (Fig. 1).

Despite high production and large carbon additions, it was difficult to measure carbon accumulations or to quantify unambiguously the amount of sludge carbon in the sediments. Organic carbon concentrations indicated no accumulation in the 1S or 4S treatments (Table 8). Stable carbon isotope ratios provided a more sensitive measure of the sludge. Of the added sludge, 20 to 50 % could be found in the top 2 cm of the sediments (Table 8).

The ratio of system production to system respiration was low in the sludge treatments (0.5 in 1S and 4S, 0.7 in 8S) compared with the controls (1.05) (Table 8). Thus, added carbon may have been immediately respired, which would help to account for the high respiration rates observed. High rates of respiration in the 8S treatment drove oxygen concentration values down in late summer (Table 8). The lack of production

Table 7. Average dissolved and particulate metal concentrations (nmol kg^{-1}) found in the water column of the sludge-amended and control mesocosms. N = 14; SD: standard deviation

Tank and treatment		Cu		Ni		Cd		Mn	
		Diss.	Part.	Diss.	Part.	Diss.	Part.	Diss.	Part.
5, Control	Avg.	29.5	7.2	53	3	1.9	0.2	198	196
	SD	8	2.8	17.4	1.1	1.2	0.1	186	75
8, Control	Avg.	26.8	5.3	51.7	1.9	2	0.2	164	172
	SD	4.3	2.9	7.6	1.2	1.4	0.2	175	73
2, 1S	Avg.	19.7	9.2	68.4	2.3	1.4	0.2	521	168
	SD	6.5	2.9	10.3	1.1	1	0.3	227	182
6, 4S	Avg.	19.6	7.4	61.4	1.7	1.1	0.3	491	120
	SD	5.8	4	10	1.3	0.5	0.6	172	140
7, 8S	Avg.	23.3	6.4	61.3	2.4	1.7	0.2	371	140
	SD	3.7	3.1	10.4	1.2	0.6	0.3	167	120
EPA Saltwater Water Quality Criteria (EPA 1980a, b, c.)			46		135		40		NA

Fig. 1. System production and respiration values from dawn and dusk measurements of oxygen concentration. (a) Control and nutrient treatments; (b) control and sludge treatments

Table 8. Fate of sewage sludge carbon and effects on oxygen concentration

Parameter	Sludge treatment		
	1S	4S	8S
Sludge carbon added (gC m ⁻² 99d ⁻¹)	30	126	254
Excess carbon found in sediment ^a (gC m ⁻²)	b	b	109
Sludge carbon found in sediment ^c (gC m ⁻²)	6	49	129
% Sludge in sediment ^c	20	39	51
System P/R ^d	0.5	0.5	0.7
Mean dawn O ₂ ^e (mg l ⁻¹) (lowest value)	5.7 (4.8)	2.3 (1.6)	1.0 (0.2)

^a In top 1.3 g sediment based on organic carbon concentrations (September minus June values)
^b Not calculated because initial and final volumes not detectably different (Oviatt et al. 1986b)
^c In top 2 cm sediment, calculated from $\delta^{13}\text{C}$ values. None was detected in the 2 to 3 cm interval
^d June to September 1984
^e 14 Aug to 17 Sep 1984

rather than high respiration rates also caused low oxygen concentrations in the 4S treatment (Fig. 1, Table 8). Water column oxygen concentrations in the controls, nutrient treatments and the 1S treatment never were less than 3 ppm; whereas, over the course of the experiment, the 4S treatment had such low values nearly 40 % of the time and the 8S had them nearly 50 % of the time (Fig. 2). In a 28 mo eutrophication experiment with up to 4 times as much nutrients added as in this experiment, low oxygen problems were encountered less than 5 % of the time, further suggesting that the carbon in the sludge treatments was being respired at a rate that the system production of oxygen could not sustain (Fig. 2).

Analyses of oxygen concentration, net respiration and diffusion rates in the 8S treatment indicated the sequence of events in hypoxia (Fig. 3). Peaks of production were followed or coincided with higher peaks of respiration suggesting another source of carbon or co-metabolism of sludge carbon (Fig. 1). During the final 2 mo, there was only one short period of net production

Fig. 2. Frequency of weekly dawn oxygen concentration less than 3 ppm during the 28 mo eutrophication study at 7 nutrient levels, the 3 mo nutrient study at 4 nutrient levels, and the 3 mo sewage sludge study at 3 sludge levels

(Fig. 3). Between peaks of respiration, diffusion of oxygen into the mesocosm helped prevent anoxia and often exceeded 24 h net respiration rates (Fig. 3). These peaks of respiration caused severe declines in the oxygen concentration. In late August one decline resulted in severe anoxia, with a resulting mean oxygen concentration of 0.19 mg l^{-1} on 4 Sep.

Three interpretations were explored to try to explain the lack of sustained high primary production in late summer, which, together with high respiration, caused hypoxia in the 4S and 8S treatments: toxicity of the sludge to phytoplankton; nutrient limitation; and limitation of phytoplankton production by grazing.

Toxicity to phytoplankton would be suggested if the sludge treatments had lower production and/or lower chlorophyll than the corresponding nutrient treatments, but this was not consistently the case (Fig. 4). For both system production and C-14 production, 1S and 4S had lower values than 1N and 4N, but 8S had higher values than 8N (Fig. 4). Production-to-biomass ratios ranged from 1 to 2 in controls, and from 0.5 to 0.7 in treatments, but showed no trend among nutrient or

Fig. 3. Mean oxygen concentration, 24 h net respiration and diffusion rate of oxygen into the water column in the 8S treatment from 23 Jul to the end of the experiment

sludge treatments (Table 9). All ratios were within the range predicted for fast-growing algal populations (Banse & Mosher 1980). Thus there was no evidence of sewage sludge toxicity in production or chlorophyll measurements.

Nutrient limitation probably limited system production only during the first phase of the experiment, when production was high in the upper level treatments. Time series of total dissolved inorganic nitrogen showed low concentrations throughout the experiment for the control, and low initial values with subsequent high values for both nutrient and sludge treatments (Fig. 5). From this analysis it can be concluded that there was no nutrient limitation in the sludge treatments during the second half of the experiment.

In 4X and 8X, nutrient treatments had much higher concentrations of nutrients in the water in late summer than did the sludge treatments (Fig. 5). This pattern suggests that a large portion of the particulate nutrients in the sludge (38 % of total N, and 64 % of total P) were being trapped in the sediments and not immediately remineralized. Nevertheless, benthic remineralization rates of ammonia were higher in sludge treatments (mean range 310 to $690 \mu\text{g-at m}^{-2} \text{ h}^{-1}$) than in corresponding nutrient treatments (mean range 140 to $240 \mu\text{g-at m}^{-2} \text{ h}^{-1}$). Four measurements of remineralization

Fig. 4. Linear functional regressions of mean chlorophyll concentrations and mean daily system productions and of mean chlorophyll concentrations and mean daily C-14 productions by control, nutrient treatments, and sludge treatments

Table 9. Calculated ratios of ^{14}C -measured production to carbon biomass calculated from chlorophyll in each treatment

Treatment ^a	^{14}C prod ^b ($\text{gC m}^{-2} \text{d}^{-1}$)	C/Chl ^c (mg mg^{-1})	Chlorophyll (gC m^{-2})	P/B
C	0.87	36	0.785	1.10
C	1.16	13	0.494	2.35
C	0.96	18	0.565	1.68
1N	1.32	39	1.790	0.74
4N	1.40	38	1.847	0.76
8N	1.44	40	3.184	0.45
1S	0.99	54	1.755	0.56
4S	1.03	61	2.037	0.51
8S	1.83	21	2.581	0.71

^a C: control; N: nutrient treatment; S: sludge treatments
^b 4 h mid-day incubations assumed to equal 57 % daily production
^c Particulate carbon values regressed on mean chlorophyll values using functional regressions for each treatment

rates over the summer failed to show the depressed rates which Perez et al. (1983) was able to attribute to phthalates. The sheer amount of organic matter in sludge (44 % of dry wt. was carbon and 2.5 % of dry wt. was nitrogen) compared to the organic content of the surface 1 cm of control sediment (2.4 % carbon, 0.2 % nitrogen by dry wt.) may have overwhelmed any suppression of rates due to phthalates.

Grazing by benthic macrofauna can limit phytoplankton biomass even in the presence of high nutrient concentrations (Cloern 1982). The same 8 species (*Ampelisca* sp., *Chaetozone* sp., *Mediomastus*

ambiseta, *Mulinia lateralis*, *Nucula annulata*, *Polydora ligni*, *Streblospio benedicti*, *Yoldia limatula*) were the most numerically abundant at the beginning and end of the experiment in all treatments, although the order of species abundance changed. While all 8 species tended to increase modestly in the nutrient treatments and explosively in the sludge treatments, most of the increase was due to interfacial grazers (Fig. 6). By August, spionid polychaete and amphipod grazers accounted numerically for 28 % of macrofauna in the controls; in the nutrient treatments, these groups composed 40 % of the macrofauna; and in the sludge treatments, spionid polychaetes alone account for 65 % of the macrofauna, amphipods being relative rare (Keller et al. 1987). Prolonged and extreme hypoxia in 8S likely caused extreme lethargy and reduced grazing pressure, allowing phytoplankton blooms to occur through the late summer period (Fig. 1, Table 7). Anoxia in 8S in early September reduced abundances of all macrofauna except *N. annulata* (Fig. 6). By contrast, in treatments which did not experience prolonged and extreme hypoxia, grazing pressure from interface grazers may have prevented late summer blooms of phytoplankton (Fig. 7 & 1).

A comparison with nutrient treatments suggested that benthic grazers in sludge treatments were utilizing sludge carbon. Total grazing biomass was calculated to include zooplankton, *Streblospio benedicti*, *Polydora ligni*, *Ampelisca* sp., and some of the rarer amphipods, *Corophium* sp. and *Leptocheirus* sp. This grazing biomass had increased by September in nutrient treatments both in the 1981–83 eutrophication experiment

Fig. 5. Total dissolved inorganic nitrogen concentrations during the sludge addition experiment. (a) control treatments; (b) nutrient treatments; (c) sludge treatments

and in the present experiment (Fig. 7). More grazers were found in sludge treatments than in comparable nutrient treatments, suggesting some species were utilizing sludge carbon (Fig. 7). Analyses of $\delta^{13}\text{C}$ indicated that many animals, including *P. ligni*, may have incorporated sludge carbon from their diets (Maughan

Fig. 6. Abundance of dominant benthic macrofauna in control, nutrient and sludge treatments during August and September. The dominant species were *Ampelisca* sp., *Chaetozone* sp., *Mediomastus ambiseta*, *Mulinia lateralis*, *Nucula annulata*, *Polydora ligni*, *Streblospio benedicti*, and *Yoldia limatula*

Fig. 7. Mean concentration of carbon in summer grazers as a function of treatment level. Data are from this experiment (C and 1X, 4X, 8X) and from the 1981–1983 eutrophication experiment (C and 1X, 2X, 4X, 8X, 16X, 32X nutrient additions). (●) Control mean and nutrient addition means; (○) sludge addition means. Note that both August and September values before and after anoxia are shown for 8S. Biomass of grazers was calculated by adding summer values (mean of weekly values) for zooplankton, and September values for *Streblospio benedicti*, *Polydora ligni*, *Ampelisca* sp., *Corophium* sp., and *Leptocheirus* sp.

Fig. 8. Linear functional regression of carbon input and concentration of *Polydora ligni* carbon. Carbon input included ^{14}C -measured and added sewage sludge carbon, assuming that all the sludge carbon went to the benthos

Fig. 9. Linear functional regression of mean diatom concentration and concentration of amphipod carbon. The 8S value of zero was omitted from the regression. Amphipod carbon concentration was derived from September data for *Ampelisca* sp., *Corophium* sp. and *Leptocheirus* sp.

1986). The regression coefficient with carbon input (^{14}C -measured production plus sludge carbon) indicated a biomass enhancement of 0.1 g C *P. ligni* for 1 g C ($r = 0.92$, significant at 0.01 level) (Fig. 8). By contrast, amphipods, which were only abundant in the

Fig. 10. Mean zooplankton biomass in control, nutrient and sludge treatments

Fig. 11. Comparison of daily system production and zooplankton biomass of the week following, in the 4S and 8S treatments. ($r = 0.427$, significant at the 0.05 level)

nutrient treatments, were better correlated with mean diatom abundance (Fig. 9).

Judging by their biomass, zooplankton were important grazers of phytoplankton in the 4S and 8S sewage sludge treatments, although they showed little enhancement over controls in the nutrient treatments (Fig. 10). However, the zooplankton in these sludge treatments were not the normally dominant *Acartia* copepod species but instead were polychaete larvae in 4S and the cyclopoid copepod *Oithona colcarva* in 8S. This copepod species has a relatively low metabolic rate compared to other copepods (Lampitt 1978, Lampitt & Gamble 1982) and probably was more tolerant of low oxygen conditions in the 8S treatment. Analysis of $\delta^{13}\text{C}$ showed no incorporation of sludge carbon by

zooplankton, implying that their food source was phytoplankton. With exceptions, zooplankton biomass was reflective of system production in the 4S and 8S treatments, also suggesting that these zooplankton were grazing on phytoplankton (Fig. 11). While zooplankton biomass was equivalent to that of benthic grazers in the control treatments, it ranged from only 1/4 to less than 1/10 that of benthic grazers in upper level nutrient and sludge treatments, respectively (Keller et al. 1987).

DISCUSSION

Fate of sludge

Settling rates, concentrations of organic contaminants, carbon content, and $\delta^{13}\text{C}$ values indicated that much of the added sludge particulates went rapidly to the benthos. Settling rate studies showed that up to 30 % reached the benthos within 10 min during mixing in the 5 m deep water column; up to 85 % reached the benthos within 24 h (Table 3). Up to 83 % of the least water soluble organic contaminants were found in the benthos at the end of the experiment (Table 4). Over 50 % of the sludge carbon could be accounted for in the surface sediments of 8S even though much of the input carbon was probably respired or could have been buried deeper (Table 8). Thus, previous calculations which have utilized a factor of 20 % of the total added sludge falling through the water column at rates of 10 m d^{-1} (O'Connor et al. 1985) may be underestimating the amount of sludge reaching the benthos by a factor of 4. If the settling rates found here were applied to 2700 m at the 106 mile site, 30 % of the sludge would reach the bottom in 90 h; 50 % would reach the bottom in 2 mo. It should be noted, however, that the dilution factor during settling rate studies in the tanks was 0.26×10^5 compared with 5×10^5 predicted for the 106 mile site. Thus more coagulation and settling might occur in the tank than in the field. Coagulation of the sludge did appear to increase settling rates (compare Expts 1 and 2; Table 3).

Sludge toxicity

No evidence of sludge toxicity to phytoplankton, as measured by primary production or chlorophyll, was detected. Nor did the observed concentrations of most sludge components reach levels previously reported to be toxic in either the water column or sediment. Miller et al. (1987) determined that copepods were more sensitive to sludge than other water column organisms. Acute effects on adult copepods of sludge from 3 treatment plants were seen at concentrations ranging from 0.03 to 0.8 % whole sludge in the water (Miller et al. 1987). In this experiment sludge was added at rates which could

have achieved 0.03 % whole sludge after 9 d of addition to the 8S treatment, if one makes the very unlikely assumption that all the sludge remained in the water column. In the case of the 4S and 8S treatments, toxic effects could not, therefore, be ruled out although the settling experiments strongly suggest that most of that sludge did not remain in the water column. Oviatt et al. (1986b) observed no direct toxic effects in extensive studies on copepod biomass and numbers and population dynamics. The change in dominant copepod species in 8S at the end of the summer could be attributed to low oxygen conditions. The toxicity of the sludge to benthic fauna was not examined and the changes observed in community structure could all be attributed to nutrient and carbon enhancement and to low oxygen conditions (Fig. 6, 7, 8 & 9). The failure of amphipods in sludge treatments could be ascribed to low oxygen and where amphipods failed, polychaetes thrived. The fact that toxicity was not observed over the 4 mo period of the experiment does not preclude it occurring over a longer time; most dump sites have a useful life of at least 5 yr, with 20 yr not uncommon.

Hypoxia

Both the 4S and 8S treatments experienced hypoxia during the last 2 mo of the experiment, probably due to the addition of sludge. In 8S, hypoxia appeared to limit grazing pressure on the phytoplankton, allowing further blooms. But even these blooms did not prevent anoxia. Large populations of grazers in 4S and in the nutrient treatments during the last half of the experiment may have prevented the blooming of phytoplankton.

An important question is whether such hypoxia could occur in the field, where a greater advective dilution occurs and the water column depth may be greater. The oxygen reservoir of a 5 m water column is about $37.5 \text{ g O}_2 \text{ m}^{-2}$; in a 30 m water column, it would be about $225 \text{ g O}_2 \text{ m}^{-2}$ and in a 2700 m water column, it would be over $20000 \text{ g O}_2 \text{ m}^{-2}$, assuming originally oxygen-saturated water columns. The net respiration rates in 8S of $1.4 \text{ g O}_2 \text{ m}^{-2} \text{ 24h}^{-1}$ could not be expected to deplete these reservoirs, particularly in surface waters, where, even in the absence of net production, diffusion of oxygen can equal or exceed net respiration (Fig. 3). However, if (1) the sludge settled rapidly below the thermocline, where diffusion is not a factor, and (2) peaks of respiration of 7 to $11 \text{ g O}_2 \text{ m}^{-2} \text{ d}^{-1}$ (Fig. 2) occurred, and (3) the summer oxygen concentrations were only 4 ppm, then water near the bottom could be greatly depleted in less than 3 d with sludge inputs of $2.5 \text{ g C m}^{-2} \text{ d}^{-1}$ (equal to the 8S treatment). Even if only half this amount reached the bottom, hypoxia could result (Fig. 2). In the field, it is common to observe low summer oxygen values in bottom water. The Christiansen Basin and upper Hudson Shelf

Valley benthic environments have been subject to seasonal reductions in dissolved oxygen, often critical to benthic organisms (Steimle et al. 1982). In southern California, where outfalls are located at the bottom beneath the thermocline, low oxygen conditions can only be exacerbated by sludge disposal (Mearns 1981). Unfortunately, no oxygen surveys appear to have been carried out at the Hyperion sludge outfall (90 m depth) to provide evidence of sludge-induced hypoxia, except that values may range from 4 to 5 ppm in bottom waters (Bascom 1980, p. 201). In this area, the shelf drops off rapidly to the Santa Monica Basin. The oxygen concentration at 400 m was 1.4 ml O₂ l⁻¹; below the sill depth of 737 m, it was 0.2 ml O₂ l⁻¹ (Jackson 1982). Jackson (1982) has calculated that sludge disposal at depths greater than 600 m would seriously lower oxygen concentrations.

Benthic fauna

At deeper water sites it seems unlikely that benthic grazers could affect surface water primary production. In most cases a thermocline would provide an effective barrier isolating benthic grazers from surface production. However, similar abundances and genera occurred in the 8S treatment and the depositional area for the New York Bight 12 mile site. In the New York Bight *Nucula proxima* attained densities of 20000 m⁻² (Boesch 1982). In 8S, August samples had a density of *Nucula annulata* of 33100 m⁻². Polychaetes were 57 % of total abundance in the New York Bight and amphipods were less than 1 % (Steimle et al. 1982). In

8S August samples, polychaetes were 87 % of total abundance and amphipods were 0.3 %.

This experiment defined the levels of sewage sludge addition which cause hypoxia in the MERL mesocosms. At summer temperatures, amounts in excess of 1 g C m² d⁻¹ initially cause changes in the zooplankton and benthic community structure and finally cause hypoxia in the shallow water column. Since the sewage sludge settled more rapidly than predicted, benthic hypoxia will, likely, also occur in deeper water columns with low oxygen concentrations in the bottom waters. These results are consistent with field studies (Steimle et al. 1982). No direct toxic effects were attributed to the sewage sludge treatments. Perhaps, this was due to the short duration of the experiment or to the generally lower concentrations of toxic components (except for PAHs) in Cranston sludge compared with New York/New Jersey sludge (Table 10).

Acknowledgements. Dr Michael E. Q. Pilson was the principal investigator on the grants which supported this project. The efforts and daily fortitude of Glen Almquist, Steve Kelly, Eric Klos, Rick Beach, Aimee Keller, Jeffrey Frithsen, Lynn Beatty, Richard Chinman, Veronica Berounsky, Carin Asjian, Paul Fofonoff, Bob Bergin, Celia Chen, Betty Buckley, Susan Jackson, Tracy Butterworth and Joceline Boucher are gratefully acknowledged. William Zuehlke undertook the sewage settling experiments as a student project. This work was supported by the Environmental Protection Agency Centers Program Cooperative Agreement 810265-03 and the U.S. Department of Commerce National Oceanic and Atmospheric Administration Office of Oceanography and Marine Assessment Grant NA-83-ABD-00008.

Table 10. Concentrations of sewage sludge compounds in whole, New York/New Jersey sludge and in the sludge used in this experiment

	New York/New Jersey sludge ^a		Cranston sludge ^a	
	Wet (µg g ⁻¹)	Dry (µg g ⁻¹)	Wet (µg g ⁻¹)	Dry (µg g ⁻¹)
Total solids	33 000	—	28 800	—
TOC	9 200	180 000	12 000	—
TN	1 500	—	2 000	—
TP	400	—	260	—
Cd	1.5	44	0.2	—
Hg	0.3	8	—	—
Cu	54	1 637	45	—
Pb	33	985	11	—
Zn	93	2 809	65	—
Fe	1 100	22 000	—	—
Cr	26	784	6	—
Ni	7.0	211	6.6	—
V	0.6	19	—	—
As	0.5	14	—	—
Se	0.1	2	—	—
Total PCBS	0.05	1.5	0.01	0.4
Total PAHS	0.7	14	1.3	46

^a O'Connor et al. 1985

^b Average from 14 samples collected over 14 wk (see Table 2)

LITERATURE CITED

- Anger, K. (1975). On the influence of sewage pollution on inshore benthic communities in the south of Kiel Bay. *Helgoländer wiss. Meeresunters.* 27: 408–438
- Banase, K., Mosher, S. (1980). Adult body mass and annual production/biomass relationships of field populations. *Ecol. Monogr.* 50: 355–379
- Bascom, W. (ed.) (1980). Coastal Water Research Project, Biennial Report. Southern California Coastal Water Research Project, Long Beach, California, p. 1–363
- Bascom, W. (ed.) (1982). Coastal Water Research Project Biennial Report. Southern California Coastal Water Research Project, Long Beach, California, p. 1–294
- Bascom, W. (ed.) (1984). Southern California Coastal Research Project Biennial Report. Southern California Coastal Water Research Project, Long Beach, California, p. 332
- Bascom, W., Mearns, A. L., Word, J. Q. (1978). Establishing boundaries between normal, changed and degraded areas. In: Bascom, W. (ed.) Coastal Water Research Project, 1978 Annual Report, El Segundo, California, p. 81–94
- Boesch, D. F. (1982). Ecosystem consequences of alterations of benthic community structure and function in the New York Bight region. In: Mayer, G. F. (ed.) Ecological stress and the New York Bight: science and management. Estuarine Research Federation, Columbia, South Carolina, p. 543–568
- Botom, M. L. (1979). Effects of sewage sludge on the benthic invertebrate community of the inshore New York Bight. *Estuar. coast. mar. Sci.* 8: 169–180
- Boyle, E. A., Edmond, J. M. (1977). Determination of trace metals in aqueous solution by APDC chelate co-precipitation. In: Gibbs, T. P. R. (ed.) Advances in chemistry series, No. 147. American Chemical Society, Washington, D.C., p. 44–55
- Brooks, N. H. (1983). Evaluation of key issues and alternative strategies. In: Myers, E. P., Harding, E. T. (ed.) Ocean disposal of municipal waste water impacts on the coastal environment, Vol. 2. Published by Sea Grant College Program, Massachusetts Institute of Technology, Cambridge, Massachusetts, p. 707–759
- Burnett, W. C., Schaeffer, O. A. (1980). Effect of ocean dumping on $^{13}\text{C}/^{12}\text{C}$ ratios in marine sediments from the New York Bight. *Estuar. coast. mar. Sci.* 11: 605–611
- Cloern, J. E. (1982). Does the benthos control phytoplankton biomass in south San Francisco Bay? *Mar. Ecol. Prog. Ser.* 9: 191–202
- Dauer, D. M., Connor, W. G. (1980). Effects of moderate sewage input on benthic polychaete populations. *Estuar. coast. mar. Sci.* 10: 335–346
- EPA (1980a). Ambient water quality criteria for cadmium. U.S. Environ. Protection Agency Rep. 440/5-80-025 or PB 81-11768 available from Natl Tech. Info. Serv., Springfield, Virginia 22150, USA, p. 1–183
- EPA (1980b). Ambient water quality criteria for copper. U.S. Environ. Protection Agency Rep. 440/5-80-025 or 81-117475 available from Natl Tech. Info. Serv., Springfield, Virginia 22150, USA, p. 1–115
- EPA (1980c). Ambient water quality criteria for nickel. U.S. Environ. Protection Agency Rep. 440/5-80-025 or PB 81-117715 available from Natl Tech. Info. Serv., Springfield, Virginia 22150, USA, p. 1–171
- EPA (1980d). Ambient water quality criteria for polynuclear aromatic hydrocarbons. U.S. Environ. Protection Agency Rep. 440/5-80-025 or PB 81-117806 available from Natl Tech. Info. Serv., Springfield, Virginia 22150, USA, p. 1–180
- Erdheim, E. (1985). United States marine waste disposal policy. In: Ketchum, B. H., Capuzzo, J. M., Bart, W. V., Duedall, I. W., Park, P., Kester, D. R. (ed.) Wastes in the ocean, Vol. 6. Nearshore waste disposal. Wiley-Interscience, New York, p. 421–460
- Frithsen, J. B., Rudnick, D. T., Doering, P. H. (1986). The determination of fresh organic carbon weight from formaldehyde preserved macrofaunal samples. *Hydrobiologia* 133: 203–208
- Frithsen, J. B., Rudnick, D. T., Elmgren, R. (1983). A new flow-through corer for the quantitative sampling of sediments. *Hydrobiologia* 99: 75–79
- Gearing, J. N., Gearing, P. J., Rudnick, D. T., Requejo, A. G., Hutchins, M. J. (1984). Isotopic variability of organic carbon in a phytoplankton-based, temperate estuary. *Geochim. Cosmochim. Acta* 48: 1089–1098
- Geyer, R. A. (ed.) (1981). Marine environmental pollution, 2, Dumping and mining. Elsevier Scientific Publishing Company, New York
- Hasle, G. R. (1978). The inverted microscope method. In: Sournia, A. (ed.) Monographs on oceanographic methodology 6. UNESCO, Paris, p. 88–96
- Hatcher, P. G., Berberian, G. A., Cantillo, G. A., McGilivray, P. A., Hanson, P., West, R. H. (1981). Chemical and physical processes in a dispersing sewage sludge plume. In: Ketchum, B. H., Kester, D. R., Park, P., Kilho (ed.) Ocean dumping of industrial wastes. Plenum Press, New York, p. 347–380
- Hunt, C. D., Smith, D. L. (1983). Remobilization of metals from polluted marine sediments. *Can. J. Fish. aquat. Sci.* 40: 132–142
- Jackson, G. A. (1982). Sludge disposal in southern California basins. *Environ. Sci. Technol.* 16: 746–757
- Keller, A. A., Frithsen, J. B., Oviatt, C. A., Maughan, J. T., Sullivan, B. K., Nixon, S. W., Pilson, M. E. Q. (1987). Marine ecosystems responses to sewage sludge and inorganic nutrient additions: a mesocosm experiment data report. MERL Series, Report No. 6. University of Rhode Island, Kingston, Rhode Island, p. 1–238
- Ketchum, B. H., Kester, D. R., Park, P., Kilho (ed.) (1981). Ocean dumping of industrial wastes. 1st International Ocean Dumping Symposium, University of Rhode Island. Plenum Press, New York
- Ketchum, B. H., Capuzzo, J. M., Burt, W. V., Duedall, I. W., Park, P., Kilho, Kester, D. R. (ed.) (1985). Wastes in the ocean, Vol. 6. Nearshore waste disposal. Wiley-Interscience, New York
- Lambert, C. E., Oviatt, C. A. (ed.) (1986). Manual of biological and geochemical techniques in coastal areas. MERL Series, Report No. 1, 2nd edn. Graduate School of Oceanography, University of Rhode Island, Narragansett, Rhode Island
- Lampitt, R. S. (1978). Carnivorous feeding by a small marine copepod. *Limnol. Oceanogr.* 23: 1228–1231
- Lampitt, R. S., Gamble, J. C. (1982). Diet and respiration of the small planktonic marine copepod *Oithona nana*. *Mar. Biol.* 66: 185–190
- Levine, E. A. (1986). Program for monitoring municipal sewage dumpsite in the New York Bight. In: Champ, M. A., Rugh, W. L. (ed.) Proceedings Ocean 86, National monitoring strategies symposium. IEEE Service Center, Piscataway, New Jersey, and Marine Technology Society, Washington, D.C., p. 760–763
- Lopez-Avila, V., Hites, R. A. (1980). Organic compounds in an industrial waste water: their transport into sediments. *Environ. Sci. Technol.* 14: 1382–1390
- Lorenzen, C. T. (1966). A method for continuous measurement

- of *in vivo* chlorophyll concentration. *Deep Sea Res.* 13: 223
- Maughan, J. T. (1986). Relationship between macrobenthic fauna and organic carbon. Ph. D. thesis, University of Rhode Island, Kingston, Rhode Island
- Mayer, G. F. (ed.) (1982). Ecological stress and the New York bight: science and management. Estuarine Research Federation, Columbia, South Carolina
- McIntyre, A. D. (1977). Effects of pollution on inshore benthos. In: Coull, B. C. (ed.) Ecology of marine benthos. Belle W. Baruch Library of Marine Science, University of South Carolina Press, Columbia, South Carolina, p. 301-318
- Mearns, A. J. (1981). Effects of municipal discharges on open coastal ecosystems. In: Geyer, R. A. (ed.) Marine environmental pollution, 2. Dumping and mining. Elsevier Scientific Publishing Company, New York, p. 25-66
- Mearns, A. J., Word, J. Q. (1982). Forecasting effects of sewage solids on marine benthic communities. In: Mayer, G. F. (ed.) Ecological stress and the New York bight: science and management. Estuarine Research Federation, Columbia, South Carolina, p. 495-512
- Miller, D. C., Marcy, M., Berry, W., Deacutis, C., Lussier, S., Kuhn, A., Heber, M., Schimmel, S. C., Jackim, E. (1987). The acute toxicity of sewage sludge to marine fish, mysids and copepods. In: Wolfe, D. A., O'Connor, T. P. (ed.) Oceanic processes in marine pollution, Vol. 5. Urban wastes in coastal marine environments. Robert E. Krieger Publ. Co., Malabar, Florida, p. 1-32
- Myers, E. P., Harding, E. T. (ed.) (1983). Ocean disposal of municipal waste water impacts on the coastal environment (2 volumes). Sea Grant College Program, Massachusetts Institute of Technology, Cambridge, Massachusetts
- Neilson, B. J., Cronin, L. E. (1981). Estuaries and nutrients. Humana Press, Clifton, New Jersey
- Nixon, S. W., Alonso, D., Pilson, M. E. Q., Buckley, B. A. (1980). Turbulent mixing in aquatic microcosms. In: Giesy, J. P. (ed.) Microcosms, ecological research. U.S. Dept. Energy Symposium Series, Augusta, Georgia, Nov 8-19, 1978, CONF 781101. Natl. Tech. Info. Serv., Springfield, Virginia 22150, USA, p. 818-849
- O'Connor, T. P., Walker, H. A., Paul, J. F., Bierman, V. J., Jr. (1985). A strategy for monitoring of contaminant distributions resulting from proposed sewage sludge disposal at the 106-mile ocean disposal site. *Mar. environ. Res.* 16: 127-150
- Oviatt, C. A., Frithsen, J., Gearing, J., Gearing, P. (1982). Low chronic additions of No. 2 fuel oil: chemical behavior, biological impact and recovery in a simulated estuarine environment. *Mar. Ecol. Prog. Ser.* 9: 121-136
- Oviatt, C. A., Keller, A. A., Sampou, P. A., Beatty, L. L. (1986a). Patterns of productivity during a mesocosm experiment. *Mar. Ecol. Prog. Ser.* 28: 69-80
- Oviatt, C. A., Maughan, J., Sullivan, B. K., Sampou, P. (1986b). Marine ecosystems response to sewage sludge and inorganic nutrients additions. Final Report to NOAA, Marine Ecosystems Research Laboratory. Graduate School of Oceanography, University of Rhode Island, Narragansett, Rhode Island
- Pearce, J. B. (1972). The effects of solid waste disposal on benthic communities in the New York Bight. In: Ruivo, M. (ed.) Marine pollution and sea life. Fishing trading Books, London, p. 404-411
- Pearson, T. H., Rosenberg, R. (1978). Macrobenthic succession in relation to organic enrichment and pollution of the marine environment. *Oceanogr. mar. Biol. A. Rev.* 16: 229-311
- Perez, K. T., Davey, E. W., Lackie, N. F., Morrison, G. E., Murphy, P. G., Soper, A. E., Winslow, D. L. (1983). Environmental assessment of a phthalate ester, Di (2-ethylhexyl) phthalate (DEHP), derived from a marine microcosm. In: Bishop, W. E., Cardwell, R. D., Heidolph, B. B. (ed.) Aquatic toxicology and hazard assessment: sixth symposium. ASTM STP 802, American Society for Testing and Materials, Philadelphia, Pennsylvania, p. 180-191
- Pilson, M. E. Q., Oviatt, C. A., Vargo, G. A., Vargo, S. L. (1979). Replicability of MERL mesocosms: initial observations. In: Jacoff, F. S. (ed.) Advances in marine environmental research. Proc. of a Symposium, June 1977, Rep. EPA-600/9-79-035, U.S. Environ. Protection Agency, Narragansett, Rhode Island, p. 359-381
- Pruell, R. J., Quinn, J. G. (1985). Geochemistry of organic contaminants in Narragansett Bay sediments. *Estuar. coast. Shelf Sci.* 21: 295-312
- Rau, G. H., Sweeney, R. E., Kaplan, I. R., Mearns, A. J., Young, D. R. (1981). Difference in animal ^{13}C , ^{15}N , and D-abundance between a polluted and an unpolluted coastal site: likely indicators of sewage uptake by a marine food web. *Estuar. coast. Shelf Sci.* 13: 701-707
- Raw, A. M. (1960). Fixed values in ocean disposal of sewage and wastes. In: Pearson, E. A. (ed.) Proceedings of the First International Conference on Waste Disposal in the Marine Environment. Pergamon Press, New York, p. 6-11
- Reed, M., Jayko, K., Isaji, T., Rosen, J. (1984). Ocean risk assessment model system (ORAMS): development and preliminary applications. A report for EPA, Contract Number 68-016621 Applied Science Associates, Narragansett, Rhode Island
- Rowe, G. T. (1971). The effects of pollution on the dynamics of the benthos of the New York Bight. *Thalassia jugosl.* 7: 353-359
- Steimle, F. W. (1985). Biomass and estimated productivity on the benthic macrofauna in the New York Bight: a stressed coastal area. *Estuar. coast. Shelf Sci.* 21: 539-559
- Steimle, F., Caracciolo, J., Pearce, J. B. (1982). Impacts of dumping on New York Bight, apex benthos. In: G. F. Mayer (ed.) Ecological stress and the New York Bight: science and management. Estuarine Research Federation, Columbia, South Carolina, p. 213-224
- Suszkowski, D. J., Santoro, E. (1986). Marine monitoring in the New York Bight. In: Champ, M. A., Puch, W. L. (Co-Chairmen). Proceedings Ocean 86, National Monitoring Strategies Symposium. IEEE Service Center, Piscataway, New Jersey, and Marine Technology Society, Washington, D.C., p. 754-759
- Wolfe, D. A., O'Connor, T. R. (ed.) (1987). Oceanic processes in marine pollution, Vol. 5, Urban wastes in coastal marine environments. Robert E. Krieger Publ. Co., Malabar, Florida (in press)
- Yentsch, C. S., Menzel, D. W. (1963). A method for the determination of phytoplankton chlorophyll and phaeophytin by fluorescence. *Deep Sea Res.* 10: 221
- Young, N. W., Young, D. K. (1982). Marine macrobenthos as indicators of environmental stress. In: Mayer, G. F. (ed.) Ecological stress and the New York bight: science and management. Estuarine Research Federation, Columbia, South Carolina, p. 527-542