DREDGED MATERIAL
RESEARCH PROGRAM

TECHNICAL REPORT D-78-24

PHYSICAL AND CHEMICAL CHARACTERIZATION OF DREDGED MATERIAL INFLUENTS AND EFFLUENTS IN CONFINED LAND DISPOSAL AREAS

by

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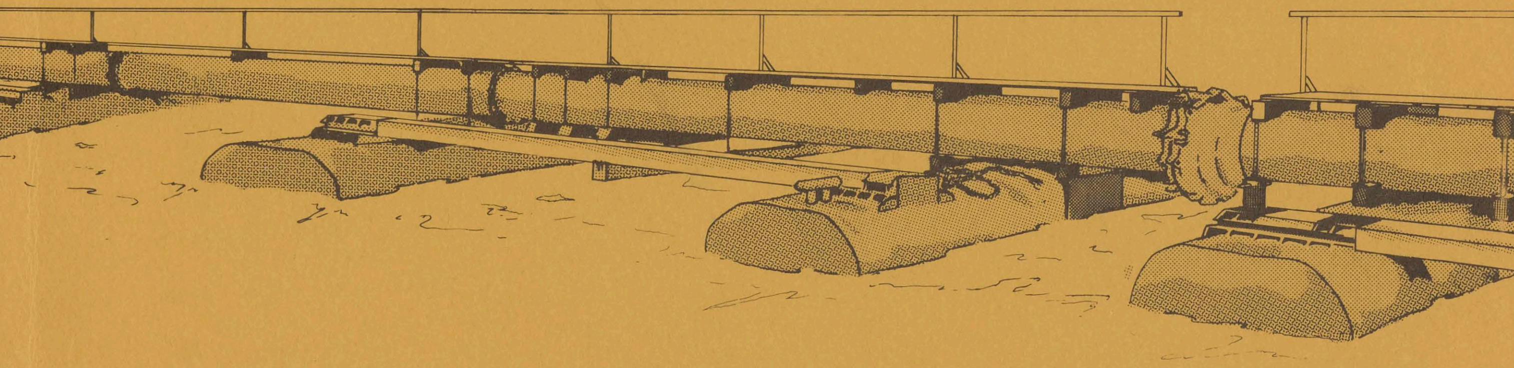

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1. The technical report transmitted herewith involves the results of a study undertaken as Work Unit 2D01 of Task 2D, Confined Area Effluent and Leachate Control, of the Corps of Engineers' Dredged Material Research Program (DMRP). Task 2D was included in the DMRP's Environmental Impacts and Criteria Development Project (EICDP), which was, in part, concerned with the evaluation of potential pollution problems associated with different modes of dredged material disposal and with establishment of criteria to mitigate or abate any observed environmental impact.

2. The objectives of Work Unit 2D01 were to determine the changes in the levels of various physical and chemical parameters resulting from the land containment of hydraulically dredged material; to ascertain the impact of effluents on receiving waters through the monitoring of surface background water, which was collected outside the effluent mixing zone; to determine the influence of vegetation and other site-specific variables in land containment areas on effluent water quality; to attempt to define contaminant trends which are common to most land containment areas; and to attempt to correlate physical and chemical parameters to determine the major factors controlling contaminant mobility. The parameters measured included nutrients, heavy metals, oil and grease, chlorinated pesticides, PCB's, and various field and laboratory physicochemical measurements.

3. The results showed that most heavy metals, oil and grease, chlorinated pesticides, and PCB's are almost totally associated with settleable solids in influent, effluent, and surface background water samples. Occasionally high concentrations of a soluble-phase contaminant in effluent samples were often associated with equally high levels in the receiving water. The contaminants that seemed to show the greatest potential environmental impact in this study include total and soluble ammonium, soluble manganese, total mercury, and dissolved oxygen; however, rapid dilution of effluent discharge in the receiving waters should greatly decrease the observed contaminant levels. A slight net increase

in pH was observed in the monitored containment areas, which would promote ammonia toxicity problems if the pH of the effluent discharge approaches or exceeds 8.5; generally pH values remained below 8. Dissolved oxygen averaged 5.3 ppm in effluents with a range from 0.6 to 12.5 ppm. Although not considered a potential problem, total lead and soluble-phase zinc and copper were sporadically high in the containment area discharges compared to background water levels.

4. Only net nitrate-nitrite nitrogen, zinc, calcium, copper, and potassium concentrations were found to increase in the soluble phase of dredged material during land containment, but the increases were either small or levels were higher in the collected background water. Soluble cadmium levels generally showed little change during slurry confinement, but there were indications that cadmium dissolution from effluent solids, after reaching the receiving waters, could be accentuated as a result of land containment of the dredged material. Many soluble-phase metal contaminants (those passing through a 0.45-μm filter) were noted to be associated with colloid-sized solids. Findings of this study indicate that thick, actively growing vegetation in land containment areas promotes the removal of ammonium-N and orthophosphate-P, as well as aiding in the filtering of fine suspended solids from the dredged slurry.

5. The information and data published in this report strongly indicate that the land containment of dredged material should not impact the environment if the site is managed to limit residence to the maximum time for effective solids removal. Prolonged residency, especially if soluble nutrient levels (e.g., -N and P) are high, may induce an unstable eutrophic environment. Such an environment could promote rapid shifts in pH and oxidation-reduction in the water, which may accentuate the dissolution of chemical contaminants from the solids. If settling of fine suspended solids cannot be attained with residency of one or two days, other measures should be considered, including multiple pond treatment or flocculant use. Vegetated areas should be utilized during the active growing season, with cropping being considered during periods of dormant growth.

6. The data in this publication should be considered, in context with past and future findings, for assessing environmental impact, establishing control methodology, and developing mitigative measures for confined land disposal areas. The results, conclusions, and recommendations should aid those persons with CE dredged material disposal, water-quality monitoring, environmental impact reports, and regulatory programs.

JOHN L. CANNON
Colonel, Corps of Engineers
Commander and Director

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Nine dredged material land containment areas, located at upland, lowland, and island sites, were monitored during hydraulic dredging operations in fresh- and brackish-water riverine, lake, and estuarine environments. Influent-effluent sampling at the diked disposal areas showed that, with proper retention of suspended solids, most chemical constituents could be removed to near or below background water levels.

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Most heavy metals, oil and grease, chlorinated pesticides, and PCB's were almost totally associated with solids in both the influent and effluent samples. The only chemical parameters which failed to show average decreases of less than 90 percent in total effluent samples include: titanium (89 percent), manganese (88 percent), potassium (78 percent), magnesium (64 percent), ammonium nitrogen (57 percent), mercury (46 percent), op′ DDE (46 percent), and pp′ DDE (21 percent). Effluent mercury and titanium frequently appeared to be mainly associated with <0.45-μm filterable particulate matter; DDE was higher in the background water. Total lead and manganese were 125 and 74 times higher in effluents than in surface background water collected near the effluent mixing zone. The lead release was very site-specific and usually its effluent-background water ratio was similar to other low-level heavy metals. Nonfilterable solids decreased by 97 percent during dredged material containment but average effluents contained 47 times the average levels in the background water samples. Land containment results in a net decrease of most soluble (<0.45-μm) chemical constituents. Parameters which showed net increase in effluent samples include, in decreasing order, nitrate-nitrite nitrogen, zinc, calcium, copper, and potassium. Only nitrate-nitrite nitrogen showed an increase of greater than about 15 percent, and surface background water contained higher levels. Soluble phase chemical concentrations in the effluents were generally similar to the surface background water levels for most sites. The soluble phase chemical constituents showing average effluent levels greater than 1.5 times the background water concentrations include: manganese (25x), chromium (6x), arsenic (4x), vanadium (4x), organic nitrogen (3x), oil and grease (3x), alkalinity (3x), zinc (2.5x), iron (2.5x), copper (2.5x), soluble total phosphorus (2x), orthophosphate phosphorus (2x), and chloride (2x). (The data for chromium, arsenic, and vanadium were collected from a limited number of sites.)

The parameters which appear to have the greatest potential impact as a result of land disposal of dredged material are ammonium, soluble manganese, total mercury, and dissolved oxygen. However, none of these should present serious problems after dilution of the effluent discharge in the receiving waters. If the pH of the effluent is above 8.5, ammonia toxicity could feasibly develop in the mixing zone. A slight net rise in water pH was observed during land containment of dredged material, with increased residency in nonvegetated disposal areas tending to accentuate this trend. The highest levels of ammonium were experienced while dredging near heavily vegetated areas, with a direct correlation observed between organic and ammonium nitrogen in the effluent samples. Actively growing vegetation in disposal areas appeared to be efficient in removing ammonium to low levels and also soluble phosphorus, if at a high initial concentration. Additionally, thick vegetation in disposal areas appeared to be proficient in filtering out the suspended solids in the slurries. Dissolved oxygen averaged 5.3 ppm in the effluents, with a range from 0.6 to 12.5 ppm. The lowest dissolved oxygen values were observed when suspended solids and/or nutrients (ammonium, orthophosphate) were high in the effluents. (Soluble organic carbon levels appeared to remain at rather stable concentrations in the dredged material during land containment and were thus not considered to represent a valid indicator of oxygen demand in the system.)

Geochemical phase partitioning of influent and effluent solids indicated that carbonate solids of several heavy metals tended to form during land disposal, along with noticeable increases in metal fixation with iron and manganese precipitates. Metal adsorption onto suspended particles (exchangeable phase) also increased slightly, along with a small increase in cation exchange capacity of effluent solids. Cadmium, although not observed to change in the soluble phase during land containment, increased dramatically in the exchangeable and carbonate phases of the solids; this could cause a potential release problem in the receiving waters.

SUMMARY

Introduction

Recent legislation has given the Corps of Engineers greater jurisdiction over lands adjacent to navigable waterways, including wetlands and drainage systems from upland areas. At this time, there is also increasing emphasis on the land disposal of especially highly contaminated or toxic dredged material, instigated by growing concern about the pollution potential of open-water disposal operations. Irrespective of any potential environmental impact created by the disposal of dredged material in aquatic systems, one must keep in mind that land disposal also produces effluent and leachate discharges, which can perhaps irreversibly impact more sensitive wetland or upland habitats.

There have been, until now, only limited studies concerning the pollution potential and physicochemical changes which are induced by the disposal of dredged material in land containment areas. Some research has suggested that the mobility or availability of many harmful chemical constituents can be accentuated by changing environmental conditions. The placement of reduced subaqueous sediments on aerobic upland soils, in small confinements which are subject to often rapid changes in oxidation, pH, salinity, temperature and other physical and chemical conditions, certainly cannot be considered as a static environment. However, other studies have failed to show any significant releases of contaminant species in disposal area effluent discharges. Due to the paucity of information available and conflicting findings, a comprehensive field study concerning the impact of land disposal seemed warranted.

Nine different confined land disposal areas were monitored in different geographic settings. Freshwater sites were located at Grand Haven, Mich. (dredging in the Grand River); Richmond, Va., at the Deepwater Terminal (James River dredging); and at Vicksburg, Miss. (dredging in Brown Lake). Brackish water dredging locations, with salinity fluctuations from 3 to 20 ppt, are listed below according to increasing salinity: Wilmington, N. C., at Eagle Island (dredged material from the Cape Fear River at the Anchorage Basin); Houston, Tex., at the Clinton disposal area (Houston Ship Channel dredging); Sayreville, N. J. (dredged material from south channel of the Raritan River); Lake Charles, La. (dredging in the Calcasieu River Ship Channel); Southport, N. C., on Oak Island (dredged material from an open-water ship channel in the lower Cape Fear Estuary); and Seattle, Wash. (dredged material from slip 1 on the Duwamish Waterway). Dredged sediments varied from mixed coarse sands and gravel to predominantly silt and clay. The organic matter content also varied in conjunction with the textural changes. These sites were chosen on the basis of high concentrations of contaminants in the dredged sediments, including oil and grease, chlorinated pesticides, PCB's, nutrients, and heavy metals. Dredged sediment characteristics that govern the mobility of contaminants include the texture, oxidation-reduction status, pH, sulfide and organic matter contents, and solids: water ratio of the bottom sediments and dredged slurries. Disposal area characteristics were also considered, such as the effective size, potential slurry residence time, degree of ponding, extent of vegetation cover, and past history of each site.

The relationship between slurry residence time and effluent quality was evaluated by considering all of the sites, especially comparisons between cross-dike and final effluent samples collected concurrently at the Sayreville and Seattle disposal areas. The effect of increased residence time on ammonium and phosphate release, in conjunction with pH changes, was evaluated by continuous monitoring of effluents from the Vicksburg disposal area during and after completion of the disposal operations. The effects of sediment texture and slurry solids content on effluent quality were based mainly on samples collected at the Richmond site, due to very wide variances in these parameters. The heavily vegetated containment area at Southport was monitored primarily to assess what influence actively growing vegetation might have on effluent quality; for comparative purposes, the Wilmington site contained thick dormant and dead vegetation during the winter collection. Salinity effects were evaluated by comparing trends at all of the disposal areas because of the wide range of salinities encountered. However, other physicochemical variables, such as those prevalent in freshwater versus marine environments (e.g., variance in sulfide levels) were always considered in context with the salinity comparisons.

Contaminant availability and toxicity are regulated by the chemical compounds with which they become associated, as different compounds have unique properties which control the solubility and mobility of a given contaminant. Although the number of discrete compounds is immense, the association of a contaminant with a general group of chemical complexes can be determined by subjecting the sediment to different specific chemical extractions or digestions, which in this study will be referred to as "geochemical phase partitioning." The association of metals with the soluble, exchangeable, carbonate, and easily reducible complexes was determined for influent and effluent solid phase samples from Wilmington, Richmond, Lake Charles, and Seattle; the organic-sulfide phase was also determined in the Seattle samples. The metals which are mainly bound in very stable crystalline matrices were included in a final total acid digest of samples from the four sites.

Additionally, an evaluation of the standard elutriate and diluted sediment pore-water tests was performed using sediments and water from the Seattle dredging site and comparing the contaminant release with effluent water composition from the adjacent disposal area during actual disposal operations. The determination of the general association of heavy metal contaminants with three different suspended particle size fractions was also performed with effluent solids from Seattle. Thus, several interrelationships could be investigated for the Seattle site.

Methods and Materials

More than 50 different physical and chemical parameters were determined in total samples, less than 0.45-µm filtrates, greater than 0.45-µm nonfilterable (suspended solids), and/or centrifuged solids of influent, effluent, and surface background water samples. Influents were generally collected beneath the end of the dredge discharge pipe in the turbulent mixing pool; effluents were obtained either at the outfall pipe beneath the sluice or from the back side of a weir structure; surface background water was collected adjacent to but outside of the effluent discharge mixing zone. Daily compositing was performed in most cases to obtain more representative samples. Three to four samples were collected from each monitoring station at the sites; six samples were obtained at the Southport disposal area but these were divided into an initial and final set for comparative purposes in the vegetation interaction study.

Salinity, conductivity, dissolved oxygen (DO), slurry pH, and water temperature were measured in the field for influent, effluent, and background water samples; disposal area sediment pH and oxidation-reduction potential (Eh) were also obtained at each of the nine sites. Additional physicochemical parameters were determined in the laboratory; namely, mechanical particle size analysis (percent sand, silt, clay), Coulter Counter suspended particulate size determinations, total solids, nonfilterable solids, settleable solids, volatile solids, cation exchange capacity, alkalinity, and chemical oxygen demand (COD). The concentrations of 20 nutrients and metals were determined. These included total and organic carbon; organic, ammonium, nitrite, and nitrate nitrogen; total and orthophosphate phosphorus; sulfide; calcium; magnesium; potassium; sodium; iron; manganese; zinc; cadmium; copper; nickel; lead; mercury; chromium; titanium; vanadium; and arsenic. Chlorinated hydrocarbon determinations were made for PCB's, DDT, DDE, DDD, dieldrin, aldrin, lindane, heptachlor, heptachlor epoxide, and chlordane. Also, oil and grease, sulfate, and chloride measurements were made.

Results

Data comparisons of acid digests (bulk analyses) for total influent and effluent samples from the nine disposal areas showed that total concentrations of most chemical constituents showed significant decreases. The following parameters displayed decreases of $\geq$ 95 percent in total samples:

op´ and pp´ DDD (~100%); chromium (~100%); op´ DDT (98%); suspended/nonfilterable solids (97%); total solids (96%); total organic carbon (96%); zinc (96%); cadmium (96%); copper (95%); nickel (95%); and arsenic (95%).

Chemical parameters in total samples which exhibited removal efficiencies of 90 to 95 percent included the following:

organic nitrogen (94%); oil and grease (94%); iron (94%); total phosphorus (93%); vanadium (93%); lead (92%); PCB's (91%); sulfide (91%); calcium (90%) and pp´ DDT (90%).

Total effluent samples showed decreases of $\leq$ 90 percent for the following parameters which are arranged in the order of decreasing removal:

titanium (89%); manganese (88%); settleable solids (85%); potassium (78%); magnesium (64%); ammonium nitrogen (57%); mercury (46%); op´ DDE (46%); and pp´ DDE (21%).

The removal efficiency for the chemical constituents present in sample filtrates passing a 0.45-µm pore size filter (soluble phase) are listed below, in the order of decreasing removal:

arsenic (85%); orthophosphate phosphorus (77%); iron (77%); oil and grease (72%); organic nitrogen (63%); soluble phosphorus (62%); manganese (38%); ammonium nitrogen (35%); alkalinity (30%); organic carbon (30%); cadmium (25%); vanadium (17%); nickel (14%); chloride (6%); magnesium (4%); sodium (4%); titanium (3%); lead (0%); and mercury (0%).

Soluble phase physicochemical parameters which showed increases in concentration as a result of confined land disposal included the following:

nitrate + nitrite nitrogen (94%); DO (40%); conductivity (24%); salinity (17%); zinc (16%); calcium (13%); chromium (13%); water temperature (13%); copper (11%); and potassium (2%).

Comparisons were also made between the levels of different parameters in surface background water and effluent samples. The constituents which were 50 fold or more higher in effluent samples included:

settleable solids (690x); soluble titanium (280x); total lead (125x); total manganese (74x); and soluble ammonium nitrogen (50x).

Parameters which were 25 to 50 times lower in background water (high in effluents) were as follows:

suspended/nonfilterable solids (47x); total phosphorus (45x); total ammonium nitrogen (37x); total titanium (36x); total organic nitrogen (26x); soluble phase manganese (25x).

The remaining constituents which were higher in the effluent samples (0 to 25 times greater) are listed below in decreasing order, divided into total and soluble phase counterparts:

Total samples: mercury (24x); arsenic (24x); organic carbon (23x); cadmium (17x); total solids (16x); PCB (9x); pp´ DDT (> 7x); copper (7x); chromium (7x); sulfide (4x); zinc (4x); potassium (3x); calcium (2.5x); nickel (2.5x); op´ DDT (> 2x); vanadium (1.5x); magnesium (1.5x); sodium (1.5x); op´ DDD (1x); and pp´ DDE (1x).

Soluble phase samples: chromium (6x); arsenic (4x); vanadium (4x); organic nitrogen (3x); oil and grease (3x); alkalinity (3x); zinc (2.5x); copper (2.5x); iron (2.5x); soluble phosphorus (2x); orthophosphate phosphorus (2x); chloride (2x); salinity (1.5x); conductivity (1.5x); sodium (1.5x); magnesium (1.5x); potassium (1.5x); organic carbon (1x); nickel (1x); lead (1x); and mercury (1x).

The only parameters which were higher in the surface background water samples included DO, nitrate + nitrite nitrogen, and op´ DDE, with effluent samples containing 54, 49, and 13 percent of the background water levels, respectively. The pH of influents, effluents, and background water showed a statistically significant but small increasing trend, with respective values of 6.6, 7.15, and 7.5.

The geochemical phase partitioning data for influent and effluent solids showed that some metals exhibited noticeable phase changes during migration of suspended solids in the dredged slurry across land containment areas, while other metals showed little change. Also, the shifts in respective phases differed for each element. The exchangeable and carbonate phases of metal-bound solids could be considered as more available to organisms as these are most easily removed by mild chemical treatment. About a third of the solids-bound calcium and sodium were removed during extraction of the exchangeable phase, with measurable increases of exchangeable calcium, sodium, copper, and arsenic noted in effluent solids. Exchangeable phase manganese, magnesium, and cadmium were high (∿ 10 percent) in both influent and effluent solids. Most of the metals showed increases in their carbonate phase concentrations as a result of confined disposal. Influent solids generally showed high carbonate phase values for cadmium and manganese, while zinc, cadmium, manganese, lead, copper, and sodium showed major increases in effluent solids. Carbonate phase cadmium, zinc, and manganese composed 57, 33, and 20 percent of the respective concentrations in effluent solids. Iron, manganese, cadmium, and copper increased in the easily reducible phase of the effluent solids, although only manganese showed a major increase. Upon total digestion of the remaining solid phases, most metals (except for iron, nickel, and chromium) showed noticeable decreases in the digests. A limited amount of data on the organic-sulfide phase (Seattle site) suggests that the concentration decreases were mainly associated with reductions in solid phase organic and/or sulfide complexes during disposal area detention under oxidizing conditions. Metals showing major phase changes included manganese (easily reducible phase), cadmium (carbonate and easily reducible phases), zinc (carbonate phase), lead (carbonate phase), copper (carbonate phase), sodium (exchangeable phase), and calcium (exchangeable phase). Metals showing little change in phases during the solids detention times included chromium, nickel, mercury, potassium, and magnesium. Iron could not be properly evaluated because of its high total concentration. Arsenic showed a large increase in an exchangeable phase extract of an effluent sample from the Seattle site.

The particulate fractionation study of effluent solids from the Seattle site indicated that a significantly large quantity of particulate potassium (58 percent) and chromium (52 percent) in effluents from this site could pass through a 0.45-μm membrane filter. Large amounts of particulate copper (39 percent), nickel (33 percent), and vanadium (30 percent) also seemed to be included with the 0.45-μm filtrates (soluble phase). For most of the metals, about 10 percent of their total solids concentration was present in the less than 0.45-μm filtrate digests. Zinc, sodium, iron, calcium, and manganese were mainly associated with the greater than 8-μm particulate fraction of the Seattle effluents.

The overall effect of residence time on effluent water quality could not be thoroughly evaluated. However, the data indicated that other variables (e.g., organic matter content) were more important. Increased residence time may affect various important physicochemical variables in divergent ways, with either uniform or fluctuating changes (e.g., for pH, Eh) occurring over time; the nature of these changes is dependent on many other factors. Sediment texture showed some correlations with nutrient and metal release for the Richmond site. Sandy (porous) sediments which contain a high organic content released more iron and manganese than fine-grained sediments of similar organic content. Zinc release was greatest from the fine-grained sediments, while effluent cadmium concentrations were notably lower than comparable background water concentrations during the dredging of fine-grained solids, indicating a scavenging effect. Ammonium nitrogen concentrations in influent slurries from the Richmond site were closely related to the total organic nitrogen content of the sediments.

The standard elutriate and diluted pore water tests, each using a 1:4 ratio of sediment and water from the Seattle dredging site, were conducted to evaluate their applicability to predicting effluent water quality from land containment areas. Overall, the standard elutriate test made a better prediction, most probably because the required settling time allowed for a closer duplication of the land disposal environment. However, at this site in particular, release of some metals from the dike material (especially iron, manganese, and zinc) complicated the interpretation of the tests. There was little difference in organic carbon, nitrate nitrogen, mercury, chromium, and arsenic concentrations in filtrates from both tests. Most of the other parameters showed contrasting values.

The heavily vegetated Southport disposal area, elicited almost complete removal of visible suspended solids. The initial effluent samples, collected when only about 10 percent of the site was ponded and most of the slurry was migrating as overland flow through the plant growth, were of very high clarity with almost no visible turbidity. Effluents collected after most of the shorter vegetation had been buried and about half of the site was ponded, showed a higher solids content, although it was also very low (0.17 mg/ℓ suspended solids). Most of the effluent samples contained solids levels similar to the surface background water. Effluents from the unvegetated Seattle site also showed comparable solids removal. However, an organic polymeric flocculent had been added to this two-compartment disposal area, and flow rates were critically monitored as part of a special PCB cleanup operation. Ammonium nitrogen and soluble phosphorus seemed to be removed at above normal levels from the vegetated area at Southport, despite relatively high influent levels for each. Most of the trace metals in total samples decreased in direct proportion to the suspended solids removal. However, only soluble phase iron and manganese showed noticeable overall declines. The very high effluent levels of soluble phase zinc, cadmium, copper, and nickel were reflected by equally high background water concentrations.

Conclusions and Recommendations

The foregoing general findings, in conjunction with a critical analysis of the data, prompt several conclusions. Generally, the removal efficiency for most heavy metals closely paralleled the removal of the solids during dredged slurry containment in land disposal areas. However, different metals seemed to have varying affinities for different particle sizes, and if the particulate phase passes through a 0.45-μm filter, the associated metals will more than likely be regarded as soluble phase constituents. Total mercury in effluent samples decreased by only 46 percent, which indicates that it was often associated with a fine particulate fraction and/or one of low specific gravity (e.g., organic suspended solids). Other heavy metals which showed removal efficiencies appreciably less than for the solids removal included arsenic (85 percent) and titanium (89 percent) although there were only limited data for these elements.

The particulate fractionation of effluents from the Seattle site, which consisted mainly of oxidized iron hydrous oxide precipitates, also showed that appreciable quantities of some metals, especially chromium and potassium, were associated with particles which could pass through a 0.45-μm filter. The mercury analyses, which used two methods (the cold vapor technique and high temperature charring in a Zeeman spectrophotometer), also clearly showed that much of the filterable mercury was associated with very fine particulate matter. Soluble phase titanium in the Sayreville samples mainly originated from the settling of very fine aerosol particles into the water near the dredging site. Thus, the filter size and the instrument employed for analyses are of great importance in determining "soluble phase" concentrations. The impact that fine particulate matter has on aquatic life is not well documented, but the findings of this study suggest that perhaps unfiltered effluents should be included in any predictive test of effluent analysis rather than 0.45-μm filtrates. However, bulk analyses (acid digests) of bottom sediments or of influent samples are not recommended as they generally show a poor relationship to contaminant mobility.

Most of the chlorinated hydrocarbons (pesticides, PCB's) showed very efficient removal when proper solids retention was maintained in confined disposal areas. An exception was DDE, which showed a very poor removal efficiency. However, the dredging site bottom water appeared to be the source for the DDE and not the bottom sediments, since comparable DDE concentrations were observed in surface background water samples. Oil and grease generally were removed efficiently during dredged slurry containment. However, sediments with high contents of petroleum residues seemed to settle more slowly, often creating highly fluid oil-water-sediment suspensions near the bottom of ponded areas. Poor management may result in the release of these suspensions, resulting in a poor effluent quality.

Analytical data for influent and effluent sample filtrates showed that soluble phase ammonium nitrogen was released in high concentrations from some bottom sediments. Ammonium release was most frequently directly related to organic nitrogen concentrations (Kjeldahl nitrogen less ammonium nitrogen) in the bottom sediments. Soluble phase ammonium nitrogen concentrations in disposal area influent samples averaged 20.8 mg/ℓ with maximum levels of 70 mg/ℓ. Generally, an equivalent amount of ammonium nitrogen was exchangeable from the influent solids. A very rapid initial decrease in soluble phase ammonium was noted in most sites displaying a short slurry detention. This was attributed to sorption by disposal area solids in contact with the slurry and was most pronounced in the presence of fine-grained sediments. Actively growing vegetation in disposal areas seems to accentuate the removal of soluble phase ammonium in the slurry (also of soluble phosphorus if present in high concentration). Although ammonium nitrogen was removed from the dredged slurry by 57 percent during residency, effluent levels often remained at levels which could warrant concern, especially if high pH conditions exist in the disposal area or discharge zone which promote the formation of highly toxic, un-ionized ammonia.

Soluble manganese was the only heavy metal that was consistently released at above ambient background water concentrations in effluent filtrates while zinc and copper were significantly higher in effluents from many of the sites. High soluble phase effluent manganese, which averaged 1.45 mg/ℓ (maximum: 8 mg/ℓ), resulted from high influent concentrations and a poor removal efficiency. Copper and zinc had a tendency to increase slightly in the soluble phase of some dredged slurries during residency in land containment areas, although there were only sporadic occurrences of effluent concentrations which would warrant concern. Generally, the soluble phase concentrations of most heavy metals were closely reflected by concentrations in respective background water samples. Soluble phase cadmium showed no major change during confinement, but there were major shifts of cadmium into the easily reducible and carbonate phases of effluent solids. This suggests that cadmium had a greater potential for availability as a result of land containment of dredged material.

The results of the diluted pore water and standard elutriate test comparisons suggested that, with modifications, an elutriation of bottom sediments could approximate effluent quality. These modifications include: (a) using a sediment-water dilution ratio which would more closely approximate the dilution actually attained during the disposal operation (if this information is available); (b) attempting to more closely duplicate conditions in the containment area during the course of the predictive test; and (c) using the total supernatant rather than a 0.45-μm filtrate, as is prescribed by the standard elutriate test procedures. Disposal area conditions could be estimated if the disposal area to be used is known. Generally, the present elutriate test (30 minutes of shaking; 60 minutes of settling) does not allow for a sufficiently long period of settling to duplicate dredged material residency in most land disposal areas.

PREFACE

This report presents the results of a study concerned with the characterization of influents, effluents, and surface background water at nine diked land containment areas throughout the United States. The investigation was conducted as part of the Corps of Engineers' Dredged Material Research Program (DMRP) under DMRP Work Unit 2D01 entitled, "Physical and Chemical Characterization of Dredged Material Influents and Effluents in Confined Land Disposal Areas," Environmental Impacts and Criteria Development Project (EICDP). This study was conducted during the period October 1975-September 1977 by EL. Sample collection and data analysis for the Seattle site were performed by the U. S. Environmental Protection Agency (EPA) Region X Laboratory, Seattle, Wash. Data analyses for the Sayreville, N. J., Houston, Tex., and Grand Haven, Mich., sites were accomplished by the Environmental Engineering Department, University of Southern California (USC), Los Angeles, Calif. Additional analyses were conducted at U. S. Testing Co., Memphis, Tenn. The investigation was conducted by Messrs. R. E. Hoeppel and T. E. Myers and Dr. R. M. Engler, with joint effort from the Ecosystem Research and Simulation Division (ERSD) and Environmental Engineering Division (EED), EL. The study was under the general supervision of Dr. R. M. Engler, Project Manager, EICDP, and Dr. John Harrison, Chief, EL.

The collection of field samples, field measurements, and general site surveys were conducted by Messrs. H. L. Horstmann, J. Eggleston, R. A. Shafer, T. E. Myers, and R. E. Hoeppel from EL, and by Messrs. D. M. Stewart, Jr. and C. R. Herrington from the Hydraulics Laboratory at WES. Supervision of field sampling for the Seattle site was under Dr. J. N. Blazevich, EPA Region X Laboratory, Seattle, Wash. The vegetation survey for the Sayreville site was performed by Dr. D. Sistrunk, Salisbury St. College, Salisbury, Md.; the vegetation survey at the Oak Island site, Southport, N. C., was made by Dr. R. Jones, Louisiana Tech University, Ruston, La. Additional field monitoring help was given at the Grand Haven site by Mr. B. Sabol, Detroit District; and at the Wilmington, N. C., site by Mr. C. Schillinger, Wilmington District. Preparation and supervision of the analysis of samples from Wilmington, Richmond, Lake Charles, Vicksburg, and Southport were by Mr. T. E. Myers and Ms. K. Myers, EED, EL, WES. Preparation and analysis of samples from Sayreville, Houston, and Grand Haven were primarily directed by Mr. M. Knezevic, Environmental Engineering Department, USC. Mr. A. Gahler and Dr. J. N. Blazevich of the EPA Region X Laboratory were in charge of preparation and analysis of samples from the Seattle site. Coulter Counter analyses were performed by Mr. J. Carroll, EL. Dr. C. B. Loadholt, Medical University of South Carolina in Charleston, performed the statistical analysis for the study.

Cooperation and assistance were received from the following Corps of Engineers personnel: Messrs. T. Clark, D. Suszkowski, and S. Caffiero, New York District; Messrs. B. Hopkins (Fort Point Area Office), J. Bissel, and E. Cobb, Galveston District; Messrs. B. Sabol, D. Billmaier, and R. Kettleman (Grand Haven Area Office), Detroit District; Mr. T. Russell and Ms. C. Correale, Wilmington District; Messrs. R. Wescott and R. Whitehurst, Norfolk District; Messrs. A. J. Heikamp, B. Vick, and R. Hargrave (Lafayette Area Office), New Orleans District; Messrs. L. Juhnke and R. Parker, Seattle District; and Messrs. J. M. Peterson and T. A. Leggett, Engineering and Construction Services Division, WES.

Additional assistance in data interpretation and review of the report was obtained from a number of scientists, including Dr. K. Y. Chen, Head, Environmental Engineering Department, USC; Dr. J. N. Blazevich, EPA Region X Laboratory; Drs. D. Wilson, R. M. Engler, and C. R. Lee, EL, WES.

Directors of WES during the conduct of the study and preparation of the report were COL G. H. Hilt, CE, and COL J. L. Cannon, CE. Technical Director was Mr. F. R. Brown.

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CONVERSION FACTORS, U. S. CUSTOMARY TO METRIC UNITS OF MEASUREMENT

U. S. customary units of measurement used in this report can be converted to metric (SI) units as follows:

Multiply	By	To Obtain
acres	0.4047	hectare
cubic yards	0.76455	cubic metres
feet	0.3048	metres
gallons (U. S. liquid)	3.7854	liters
gallons per minute	3.7854	liters per minute
inches	2.54	centimetres
miles (U. S. statute)	1.609	kilometres
yards	0.9144	metres
Fahrenheit degrees	5/9	Celsius degrees or Kelvins*

---

* To obtain Fahrenheit (F) temperature readings from Celsius (C) readings, use the following formula: $F = (C^{o} \cdot 9/5) + 32$. To obtain Kelvin (K) readings, use: $K = C^{o} + 273.15$.

PHYSICAL AND CHEMICAL CHARACTERIZATION OF DREDGED MATERIAL INFLUENTS AND EFFLUENTS IN CONFINED LAND DISPOSAL AREAS

PART I: INTRODUCTION

Background

1. The Corps of Engineers has regulated dredged material disposal since the passage of the River and Harbor Act of 1899. Although permit issuance under that Act was originally dictated by economic and navigational concerns, the recent emergence of an environmentally aware public has prompted the Corps to also assess the ecological impact of its operations before permit issuance. Long-term ecological benefits, heretofore neglected, are now a major consideration in all Corps cost-benefit analyses. Recent legislation, especially the water quality and ocean dumping laws of 1972 (Public Laws 92-500 and 92-532), has more fully defined Corps responsibility. Section 404 of Public Law 92-500 has specifically expanded the Corps' jurisdiction over wetland and marsh ecosystems adjacent to navigable waterways.[1]

2. Approximately 400 million yd^3* of dredged material is removed annually from navigable waters in the United States. Whereas less than two fifths of the dredged material was placed on land a few years ago,[2] a rapid shift toward confined upland disposal is now occurring. Recently, the Corps has decreased its ocean disposal of dredged material by about 12 percent annually.[3] In the Great Lakes, where 115 harbors and channels are maintained for navigation, the trend is toward confined disposal despite cost increases of nearly 350 percent.[4] Average total expenditure for Corps dredging now exceeds $200 million with annual increases of 10 to 15 percent forecast for the future.[5]

3. Environmental impact must be considered when choosing a mode of dredged material disposal. Unfortunately, many past decisions have been based on minimal definitive information. This is particularly

---

* A table of factors for converting U. S. customary units of measurement to metric units is presented on page 21.

true for the disposal of dredged material on the land. Since upland areas along navigable waterways are becoming scarce, there is increasing economic pressure to create fast land by disposing dredged material on wetlands and marshlands. If the present trend in upland disposal is to continue, additional costs for long-distance pumping or hauling to inland sites will have to be realized. Dredged material, low in available toxic metals and organic contaminants but high in fixed nutrients, could be employed in marshland creation or for improving agricultural soils. However, present criteria emphasize that contaminated sediments be given top priority for confined upland disposal.[6]

Dredged material composition and modes of dredging and disposal

4. Dredging involves the removal of sand, silt, clay, and various organic materials from underwater surfaces by some excavation mode and subsequent conveyance and disposal of the material.[7] The inorganic and organic composition of the bottom sediments to be dredged is somewhat site specific, often varying greatly in particle size and organic content within a very small area. The physical and chemical properties of the solids and associated water are dependent on the complex interactions of often rapidly changing environmental factors as well as the natural levels of metals, nutrients, and other chemical species. Chemical changes, in turn, can govern physical changes, such as pH, oxidation-reduction (Eh), and sorption phenomena.

5. Research defining the physical and chemical composition and time-dependent changes of bottom sediments and pore water under natural conditions in different environments has been fairly extensive,[8-15] although data are frequently collected from many single-site studies using a variety of experimental techniques. The overall findings indicate that the interactions between solids and the aqueous phase are extremely complex. Some recent data are also available on the effects of rapid dispersion and resedimentation of solids in a water column under laboratory conditions.[16-18] Additionally, there have been several studies concerning the fate of contaminants associated with

naturally suspended and transported solids.[19-23] All of the above-mentioned phenomena are somewhat applicable to the disposal of dredged material in upland containment areas.

6. Hydraulic pumping of a solids-water slurry is at present the major means of conveyance of dredged material to confined disposal areas, usually by direct pipeline transport from a hydraulic dredge or by pipeline pumpout of a modified hopper dredge. To avoid the costs and problems of long-distance transportation, the majority of land disposal areas are located immediately adjacent to the waterways being dredged.[2,4,7] Commonly 75 to 95 percent of the transported volume of dredged material is bottom water from the dredging site, while the remainder of the transported volume consists of the dredged sediment and sediment pore water. After deposition of the slurry at one end of a diked land disposal area, the sand and larger sized solids rapidly settle near the discharge pipe, often creating mounds of alluvium and inducing turbulent overland flow of the slurry. The silt, clay, and low-density organic particles then settle more slowly, mainly in ponded regions within the containment area. The resultant stratification is thus both vertical and horizontal, with gravel and sand (also clay balls, common with new-work dredging) depositing near the influent source. Sediment stratification is further complicated by occasional movement of the discharge pipe and use of several effluent sluice or weir structures. As a result, effluent solids may occasionally contain sand suspended from the bottom by turbulent flow.

7. The solids, especially the fine-grained particles, are usually poorly consolidated after resettling within a containment area. Dredged sediments high in petroleum residues are especially likely to form low-viscosity fluid layers near the bottom of containment area ponds. Thus, discharge of the supernatant, combined with the use of surface skimmers to remove surface films, gives an effluent of low turbidity and solids content. However, not all turbidity is associated with the suspension of sediments. Chemical precipitation, especially of iron complexes in aerated water or of organic complexes with changes in salinity or pH,

may create additional turbidity which can last for a considerable length of time.

Previous confined disposal area studies

8. The recent trend has been toward containment of contaminated dredged material in upland areas. However, extensive confined disposal area effluent[24-37] and leachate[38-40] monitoring programs have been conducted only sporadically, and specific studies have usually been limited to a single site. Presently, many disposal practices are based on the more extensive land treatment studies conducted with municipal wastewater and landfill leachates, which typically contain higher dissolved organic loadings, lower levels of suspended solids, and other characteristics dissimilar to dredged material slurries.[41]

9. Previous studies of confined upland disposal areas have demonstrated several important correlations. First, most trace metals, oil and grease, and pesticides show a strong affinity with solid particles[17,29,31,34,35] even though there appears to be no significant correlation between solids composition (bulk chemical analysis) and levels of dissolved contaminants in the effluents.[36,37,42]

Several studies[25,34,35,37] have indicated that most soluble heavy metal concentrations in confined disposal area effluents are similar to those in the ambient surface water near the discharge site. However, nutrients such an ammonium[17,25,36,37] and orthophosphate[25,35,36] could occasionally be released at levels warranting concern. Long retention of dredged material slurries within containment areas has been inferred to improve effluent water quality by increasing solids removal,[28,29,34] but other investigators[36,37] have shown experimentally that an extensive slurry residence time may release higher concentrations of some soluble contaminants. Dissolved oxygen values for effluent seem to be very site specific, with both increases[27,35,37] and decreases[33] observed over levels in the receiving water. Generally, pH values increase slightly, especially in large disposal areas,[25,35-37] although when the ambient water is high in pH (above 8.5) there is a decreasing trend toward neutral pH.[27] Dissolved sulfides in effluents do not seem to be a problem under routine disposal conditions.[33,35] Bacteriological analyses[24-26,34] indicate that anaerobic pathogenic bacteria and indicator bacteria for fecal water pollution, namely fecal coliforms and fecal streptococci, are not a problem in properly managed disposal area effluents. Although influent dredged material frequently contains very high numbers of potentially pathogenic bacteria,[26,34] periodically high coliform bacterial counts in disposal site water are considered to be more closely related to the bird population in the area.[34] In situ bioassays[33] and benthic fauna studies[29,36] seem to indicate that effluent discharge has minimal impact on organisms, although it is admitted that sublethal effects are difficult to ascertain.

Vegetation interactions

10. There are two approaches employed by Corps Districts with respect to the proper design of confined disposal areas, the choice frequently being dictated by the size and location of available land parcels. One design emphasizes barren, extensively ponded disposal areas, either quite large or small with often several internal dikes or compartments. These areas are excellent for the containment of primarily coarse sediments and for initial containment of fine particulates. However, overuse or poor maintenance, resulting in short retention time, may produce a poor quality effluent or dike failure. These areas are often not designed for continuous discharge and require prolonged periods of sediment dewatering for efficient use.

11. The other design alternative allows natural, usually wetland vegetation to remain within a large diked area. Such areas frequently function by slow overland flow of the slurry over the surface of the site, with the vegetation helping to disperse the flow and prevent channeling. Such areas often lack extensive ponding or high dikes and seem to be most efficient for the long-term deposition of fine-grained dredged material, especially slurries containing high concentrations of ammonium nitrogen.[43,44]

12. Vegetation studies[31,43] indicate that plants often take up contaminants more readily from fine-grained dredged material than from coarse sediments. The inverse relationship between particle size and surface area available for contaminant adsorption has been offered as a tentative explanation. However, the availability of many contaminants may be more dependent on the organic content and in situ variations in pH and Eh than on grain size,[10,17,23,45] a possibility which would complicate the above relationships because of the interdependence of the variables. Vegetation can also have a profound effect on microbial reactions in sediments, and exudates may also affect the physical properties of colloids and adsorption phenomena. These factors are not extensively covered in the literature, but they appear to be very complex.[43]

Objectives of the Study

13. Minimal information exists concerning the fate of contaminants entering confined disposal areas. The first step in defining the pollution potential from land containment areas is through influent and effluent monitoring programs under different environmental conditions. Disposal area characteristics, such as the presence or absence of vegetation, salinity regime, and climatic factors, may significantly influence contaminant release. The rate of mobility or fixation of different contaminants is influenced by variations in the physical and chemical conditions within the dredged sediments and disposal area slurries. Therefore, many parameters must be measured to delineate cause and effect relationships. These, in turn, may help define the management practices necessary for pollution prevention and abatement.

14. The objectives of this study were to monitor both the physical and chemical parameters of: (a) influent dredged material collected at the dredge discharge pipe; (b) effluent obtained beneath the disposal area sluice or weir; and (c) surface background water collected near but outside of the effluent discharge mixing zone. Active confined disposal sites were to be monitored at different locations throughout the United States, with the selection being limited to those sites

which receive dredged material known to contain high concentrations of a variety of contaminants. Chemical constitutents to be studied included nutrients, heavy metals, chlorinated pesticides, polychlorinated biphenyls (PCB's), and oil and grease. Only maintenance operations of hydraulic dredges were to be considered for the study.

15. Site selection criteria for the disposal areas emphasized variability in the following features: (a) extent of vegetative cover; (b) amount of ponding within the containment area; (c) soil surface area in contact with the dredged material slurry; (d) residence time of the effluent discharge; (e) salinity fluctuations encountered; (f) climatic factors; (g) past history of the disposal area; and variations in influent dredged material and disposal area sediment composition, including: (1) organic matter content; (2) sulfide concentrations; (3) nutrient content; (4) Eh and pH status; (5) salinity; and (6) dredged sediment texture. Most of the physicochemical measurements (e.g., pH, Eh, salinity, conductivity, dissolved oxygen, and temperature) were to be obtained in the field during sample collection.

16. The impact that most contaminants have on the ecosystem is dependent on their form; namely, the inorganic and organic complexes with which they are associated. Not all forms of an element are equally available or show similar toxicity to living organisms. The contaminant forms and migration potential to other complexes can be roughly calculated by subjecting the solid phase of influents and effluents to chemical extractions of increasing severity. These geochemical phase partitioning analyses, along with the previously stated measurements and observations, were used to help materialize the goals of this project.

17. The major goals of this study were in the order of importance: (a) to determine if effluent discharge from confined disposal areas can impact the quality of the receiving waters; (b) to ascertain which contaminants in dredged material are most readily mobilized during dredging and disposal in land containment areas; (c) to determine if common trends exist toward mobilization or fixation of problem

contaminants associated with effluent solids; and (e) to attempt to correlate physical and chemical parameters to determine the major factors controlling contaminant mobility.

PART II: LITERATURE REVIEW

Introduction

18. Hydraulic dredging, subsequent movement of the dredged sediment-water slurry across a confined upland disposal area, and ultimate discharge of the effluent to a large body of water can change the physical and chemical characteristics of natural water systems. Factors which can cause such changes include: (a) the rapid dilution of the sediments with water and the turbulent mixing which occur during dredging operations; (b) Eh, pH, temperature, and salinity changes which can occur within the disposal area; and (c) further physical and chemical changes in the effluent mixing zone. A continuous cycle of fixation and release of heavy metals, nutrients, and organic contaminants may occur simultaneously with these environmental changes. The formation of soluble inorganic and organic chemical complexes further complicates the situation. The chemical form of a contaminant element may dictate its availability and toxicity to organisms as well as its ability to be fixed or released from the sediment. Chemical changes are also strongly controlled by biological, especially microbial, reactions in the dredged sediments and disposal area environment.

19. Most areas which are dredged, because of their shallow depths and location, are frequently subjected to turbidity currents or other physical perturbations created by ship and barge navigation. Additionally, dredged sediments, because of their frequently close proximity to the land and the high biological productivity of the overlying water (e.g., estuarine, lake, and river environments), are continually being supplemented by both land-derived (terrigenous) and indigenous inorganic and organic solids.[45] These events, by contributing new or recycled contaminants, make it difficult to assess the source or fate of contaminants present in the water column. Pre-dredged water quality is important in hydraulic dredging operations because the dredged volume is primarily composed of water overlying

the dredging site sediments. The depth of dredging is also important because of the frequent variations in sediment and sediment pore water composition with depth.[13,46-48]

20. Many short- and long-term interactions between sediments or suspended solids and various soluble chemical species must be understood before the changes occurring during the placement of dredged material in upland disposal areas can be explained. Unfortunately, many chemical and biological transformations are at present incompletely understood. A brief outline of current knowledge is summarized below. A more detailed discussion can be found by consulting the literature.

Interactions of Sediments with Chemical Constituents

Sources of contaminants

21. Sediments are composed of primary minerals derived from the weathering of igneous and metamorphic rocks, clay minerals exhibiting varying degrees of chemical alteration, and organic matter of varying ease of biodegradability.[22,49,50] Although natural weathering processes contribute significantly to the composition of natural waters, highly concentrated local inputs are often the direct result of man's activities. Airborne particulates and volatile forms of trace metals and organic compounds also contribute to the input of contaminants to the aqueous environment. The role of natural and synthetic organic compounds in the distribution of contaminants is complex and poorly understood.[11] Nevertheless, when the input rate exceeds the natural rate of cycling, adverse ecological effects may result.

Biological activity

22. Nearshore sediments, including those found in most dredging areas, often have a higher organic content which serves as a medium for the growth of microbial populations. These microorganisms are capable of transforming both organic and inorganic compounds at often rapid rates. One important function is the oxidation and reduction of organic and metal-organic complexes for energy-producing and respiratory functions, respectively. The overall effect is the production of reducing conditions in the sediments, resulting from the net transfer of electrons

from reduced substances to oxidized substances. Microbially induced reductions are more important in nearshore environments as a greater quantity of readily degradable organic matter reaches the sediment, thus accentuating biological activity.

23. Microorganisms (bacteria, actinomycetes, fungi, and algae) and benthic macrofauna (worms, clams, crabs, etc.) are very active in most aerobic surface sediments, while large microbial populations often extend into moderately reduced anaerobic sediments which contain readily degradable organic compounds.[14,45] The degradation of large organic molecules results in many small soluble organic molecules (e.g., amino acids, fatty acids, and alcohols) and inorganic compounds (e.g., ammonium, carbon dioxide or bicarbonate ions), which can either diffuse into the deeper sediment or enter the water column. Many anaerobic microorganisms specifically depend on these soluble organic and inorganic molecules for both energy sources and for use in respiratory electron transfer reactions. With increasing sediment depth, the organic compounds tend to become more resistant to attack, the microbial populations correspondingly decrease in number, and the influence of biocatalyzed reductions becomes less important. Also, Eh tends to be controlled to a lesser degree by organic compounds, and microbial oxidation-reduction reactions, involving a number of inorganic (e.g., iron, manganese, and sulfur) compounds, become more important.[14,45,51] At the sediment-water interface there is continuous competition between the biologically induced reductions mentioned above and oxidations promoted by the continuous influx of dissolved oxygen from the water column. Usually, only a thin oxidized crust exists beneath the sediment-water interface, but this zone performs an extremely important role in trace metal and nutrient cycling.[10,14] (This topic will be discussed in detail in later sections.)

24. In summary, a typical sediment column in most dredged areas contains a surface oxidized layer, often less than 1 cm thick. A zone of intense microbial activity exists in and below the oxidized crust to a depth of about 20 cm, where Eh and chemical reactions are greatly

affected by the decomposition of organic compounds. The anaerobic profile beneath this zone is strongly influenced by biochemical electron transfer reactions initiated by inorganic compounds.[10,14,45,51]

25. The sediment hydrogen ion concentration (pH) is also governed by the types of compounds produced by biological activity, and, as hydrogen ion transfer is frequently a means for electron transport, pH and Eh are interdependent.[10,52] The pH of surface and sediment pore water is greatly controlled by the carbon dioxide system (carbonate-bicarbonate-carbon dioxide equilibria). However, various organic compounds contribute to its buffer capacity. Other inorganic complexes, including those of metals and nutrients, are important in removing hydrogen or hydroxide ions from the aqueous medium. Biological activity strongly influences the carbonate cycle, Eh-pH values, and the transformations of organic matter. Thus, biochemical interactions greatly influence the physical environment, and, in turn, these same physical phenomena control the proliferation of different organisms.[53,54]

Regulation of contaminant mobility

26. The sediment environment, including the Eh, pH, and chemical composition of the pore water, is very important in dictating the chemical speciation of an element. Environmental changes may enhance the mobility of a given element if the solid phase compounds formed have a greater solubility than the previous compounds. The mobility of a given element in the water column is also governed by the environment, including the ionic composition and strength, Eh, dissolved oxygen concentration, pH, concentrations and types of organic compounds, and temperature. Positive ions (cations) can form soluble complexes with negative ions (anions), especially in waters of high ionic strength (e.g., the marine environment); soluble organic molecules can form similar complexes at much lower concentrations. The resulting complexes for the same element can be positive, negative, or neutral in charge, depending on the environmental conditions. Thus, an element which forms an uncomplexed cation could exist predominantly as an anionic complex if the conditions are appropriate. Also, complexes of

similar charge can have different compositions and thus display entirely different properties. The availability or affinity of an element to an organism, enzyme, or other solid surfaces is often highly dependent upon the complex to which the element is bound. Increasing temperature generally increases the solubility of a given complex in the aqueous medium, but various competing reactions can complicate this relationship. Thus, there appears to be little similarity between the bulk sediment concentration of an element and its bioavailability or soluble phase concentration in the water column.[55] Other factors must certainly be considered.

27. Recent studies[17,18,55] indicate that the greatest release of sediment-bound elements may occur under changing environmental conditions. This mobility may occur when an ion or molecule is reverting from one solid phase to another under fluctuating Eh, pH, and other physical conditions and is undoubtedly influenced by varying rates of reaction for different chemical species.

28. In sediment systems, equilibrium between the solid and liquid phases for a given compound is rarely attained. Continuous change within the system, different rates of reaction for different complexes, competing reactions, and nonequilibrium reactions (e.g., coprecipitation) hinder our understanding of sediment chemistry.[11,14,56] Although Eh measurements can contribute valuable clues about the functioning of sediment chemical systems, they will not alone give definite information about the chemical species present. They will indicate the potential for the occurrence of important oxidation-reduction reactions, especially those involving the nitrogen, manganese, iron, and sulphur systems. Various organic redox couples cannot be well defined because of their complexity and continually changing states.[14,52,56-58]

Organic matter

29. Nearshore estuarine, lake, and river sediments contain a mixture of reworked plankton debris derived from the water column, recycled benthic faunal and floral debris, and terrigenous plant

material. Most of this debris consists of organic compounds which are either toxic and/or highly resistant to biochemical attack, including certain carbohydrates (e.g., cellulose and chitin) and polyaromatic compounds (e.g., lignins and humic matter). Readily degradable proteins and carbohydrates may escape enzymatic attack by complexing with lignins and other resistant macromolecules or by adsorption onto the surfaces of clays or other particles. Organic matter not protected from biodegradation is broken down into various small soluble molecules, including fatty acids, alcohols, aldehydes, water, and carbon dioxide.[45,59]

30. Most of the organic matter in recent sediments and natural waters consists of complex, heterogeneous macromolecular condensates which bear very little resemblance to the cellular constituents of organisms. These often highly aromatic compounds are collectively known as "humic matter" or "humic substances." The water soluble, lower molecular weight fractions are termed "fulvic acid," even though these compounds bear only a poor resemblance to one another in composition and properties.[15] The composition of humic matter seems to be partly dependent upon the Eh of the medium;[9] the physicochemical properties of a given molecule are highly dependent on pH as each form exhibits a diverse array of functional groups.[15,60]

31. Humic substances are primarily resynthesis products derived from microbial activity using the waste products of metabolism, especially aromatic compounds such as lignins, tannins, and phenols, as starting products. Polymerization, oxidation, methylation, and the attachment of fatty acids, alkanes, protein fragments, and practically any molecule found in the environment results in an almost endless variety of structures.[15] Humic substances can be formed in both aerobic and anaerobic environments, but various oxidations necessary for the formation of large aromatic condensates require aerobic conditions. These compounds may be predominantly formed in and above the aerobic sediments, but they persist in deep sediments because of their resistance to degradation, especially under anaerobic conditions.[9] Sedimentary humic matter is usually concentrated in clay- and silt-sized

sediments in nearshore marine environments and in the central regions of lakes. Streams continually translocate soil humus, which is very similar to nearshore sediment humus.[9] Therefore, humic matter should constitute an important part of the organic matter in dredged material.

32. Humic matter (humic acids, fulvic acids) interacts mainly with divalent metal ions to form complexes of varying stabilities and properties.[61] Important characteristics of humic substances are their ability to form water-soluble and water-insoluble complexes with metal ions and their hydrous oxides, and to interact with clay minerals and many nonhumic organic compounds.[15] The lower molecular weight fractions of humic matter (e.g., fulvic acids) have a greater metal binding capacity than the higher molecular weight fractions. Likewise the smaller, more highly charged fulvic acids are more soluble, though solubility decreases with increasing metal content. Solubility also depends on other environmental variables, including Eh, pH, salinity of the medium, and form and type of metal being bound.[9,60]

33. Natural accretion of water-soluble humic matter probably involves a combination of biological- and sunlight-promoted chemical oxidation processes, and the interactions of the organics with ions and mineral substrates, causing flocculation, cross-linking, and adsorption to solids.[62,63] While the binding of metals and organic contaminants with soluble organic matter can increase their mobility, similar affinity of contaminants with insoluble organic matter can enhance contaminant removal from water.[64]

34. Different humic substances respond differently with contaminants under varying physical conditions. However, increased salinity[65] and pH, especially in the presence of divalent "bridging" cations such as calcium, promote the flocculation of humic matter.[63] Under acidic conditions, large humus compounds tend to split into smaller fractions,[61,63] although metal binding seems to be impaired at low pH.[63,64]

35. Humic matter can be highly specific for binding with certain elements or complexes of different oxidation states.[9] Additionally, humic substances are capable of various electron transfer reactions and thus can affect Eh.[63,66] The complexation of metals with organic matter seems to be an important step in the humification process, which is dependent on microbial reactions. This is indicated by a general decrease in metal coordination with the organic compounds in more reduced sediments.[8,67]

Chlorinated hydrocarbon transformations

36. Chlorinated hydrocarbons have recently been given considerable attention because of their persistence in the environment and potential for biological magnification. Although these synthetic compounds are usually concentrated near areas of their use, such as in highly industrialized regions or agricultural watersheds, it is now apparent that they are also spread through the hydrologic regime by atmospheric transport.[68] Only a few types of chlorinated hydrocarbons will be specifically discussed; namely, those which may be a problem in most dredged material. These include the chlorinated pesticide DDT (1,1,1-trichloro-2,2-bis [p-chlorophenyl] ethane); its main transformation products DDE (1,1,-dichloro-2,2-bis [p-chlorophenyl] ethylene), and DDD (1,1-dichloro-2,2-bis [p-chlorophenyl] ethane) and PCB's. Accumulation of different DDT metabolites as well as other organochlorine compounds depends on many sediment environmental factors, including temperature, microbial activity, pH, and Eh changes in the substrate.[69]

37. The predominant transformation product of DDT in anaerobic environments appears to be DDD. Its preferential formation, which seems to be microbially induced, is accentuated by both increasing temperature (to 60 C) and high levels of organic matter. Little formation seems to occur at Eh values above +180 mV.[69] The main degradation product of DDT in aerobic sediments is DDE. The reaction is catalyzed by certain minerals and heavy metals, especially by iron oxides, and is promoted by elevated temperature. The conversion to

DDE appears to be microbial at lower pH and temperature and probably chemical at alkaline pH and elevated temperature.[70] The degradation of PCB's, although slow, is facilitated by many microorganisms,[71] especially if the PCB's are emulsified in the water phase.[72]

38. Chlorinated pesticides and PCB's are very poorly soluble in water, with natural soluble concentrations being less than 0.1 ppb. Although only trace quantities are in solution, dissolved organic matter, especially fulvic acids, can significantly increase the solubility of organochlorine compounds.[73] However, essentially all of the DDT metabolites and PCB's are associated with the sediments, where they appear to have an affinity with sediment humic substances.[74] Suspended or settleable crude oils may also have an affinity with chlorinated hydrocarbons and thus they could aid in concentrating these compounds at the air-water interface, in the water column, or in bottom sediments.[75]

Petroleum hydrocarbons

39. Petroleum hydrocarbons are very diverse in both composition and rate of transformation or loss from the environment. The physical form of the various hydrocarbons, which is partly dependent on composition, also dictates their fate. Water turbulence caused by wind may emulsify surface slicks of oil and intensify volatilization of low molecular weight fractions. Nonbuoyant oil residues may entrain suspended particles and sorb (adsorb and absorb) soluble contaminants in the water column. The settleable fractions may then undergo transformations in the sediments, resulting in the formation of additional soluble products.

40. Generally the solubility of petroleum hydrocarbons decreases with increasing molecular weight and decreasing number of attached side chains or polar groups. Likewise, high molecular weight straight chain hydrocarbons become less soluble with increasing water salinity,[76] although the presence of dissolved organic matter (fulvic acids) in the medium can counteract this effect appreciably.[77] Humic substances in sediments are known to bind with hydrophobic organic compounds,

including petroleum hydrocarbons.[77] However, hydrocarbons associated with sediments may decrease their sorptive capacity by masking adsorption sites.[78]

41. The low molecular weight aromatic molecules (e.g., benzene, and naphthalene derivatives) make up a significant part of the water-soluble fraction of both crude and refined oils. Also many aromatics, especially the alkyl-substituted aromatics, are the most toxic and most easily bioaccumulated petroleum fraction.[79] Another important group of water-soluble compounds are hydrocarbons with chain lengths of ^{14}C to ^{17}C, since many hydrocarbons with chain lengths of less than ^{14}C atoms tend to be rapidly volatilized.[78, 79]

42. Aged petroleum, which becomes associated with bottom sediments, contains mainly compounds with very low water solubility; namely, high molecular weight alkanes (e.g., paraffins larger than C12) and polyaromatic compounds (e.g., phenanthrenes, fluorenes).[79] San Francisco Bay sediments were found to contain total hydrocarbons of up to 6000 ppm dry weight, with the aromatic component contributing nearly half of this amount in some instances. Although the soluble fraction contained less than 0.5 ppm total hydrocarbons, the suspended solids in the bay system were calculated to contain over 13.5 metric tons of pollutant hydrocarbons.[80] Microbial degradation of petroleum is dependent upon many factors. Generally, the medium molecular weight alkanes are most readily utilized, and degradation is accentuated by a rich nutrient supply, increased temperature, aerobic conditions, and emulsification of the insoluble fraction.[81] Increased solubilization of a given compound will also increase its degradation.[78]

43. Although petroleum consists predominantly of hydrocarbons, certain metals may be associated with different components at appreciable levels, either from natural origins or present-day sources. Nickel and vanadium are the most common metals, although many others, especially iron, zinc, chromium, copper, manganese, and cobalt, are almost always noticeably present. These metals may be strongly bound (e.g., with porphyrins) or only weakly adsorbed. Thus, their release

and bioavailability are highly variable.[82] Also, metals may affect the biodegradability of petroleum.[83]

Sorption reactions by sediments

44. Availability or mobility of an element in sediments is highly dependent on the mode by which it is bound with the solid phase. Biochemical reaction mechanisms lead to long-term fixation of contaminants. This will be discussed in later sections. However, the rapid, physical sorption reactions of free ions or soluble chemical complexes are often a necessary first step for permitting chemical reactions to occur.

45. Sediment solids have a great capacity to adsorb ions as a result of actual or induced surface charges. The interaction of a chemical species with water, which is highly pH dependent, is important for most sorption reactions, and resultant metal hydroxide formation can promote chemical precipitation at the solid surface. This time-dependent event prevents reversible desorption.[11] Ion exchange involves generally a reversible sorption of hydrated ions. Certain clays which exhibit a high net negative surface charge (e.g., montmorillonite and vermiculite) have a capacity to sorb considerable concentrations of cations. The affinity of a given cation with a charged clay particle is dependent on the degree of ion hydration and charge. Generally, divalent and trivalent cations compete better for exchange sites on the solid phase than do monovalent cations.[11]

46. Organic molecules can be strongly sorbed at clay and other solid surfaces through either the interaction of charged functional groups or extensive hydrogen bonding; these reactions are strongly influenced by pH.[84] Organic matter, because of its very large effective surface area and number of charged groups, may account for most of the ion exchange prevalent in sediments.[85] The quantity of cations released by a sediment upon the addition of a concentrated salt solution (usually ammonium or sodium acetate) is termed the cation exchange capacity (CEC) of the sediments. The CEC may be interpreted as a rough measure of the proportion of solid phase cations which are readily available to organisms.[86]

47. Soluble anions are generally not readily adsorbed by sediments, despite the presence of some positive exchange sites on clays and organic compounds.[87] Therefore, the movement of chloride, nitrate, and a large proportion of sulfate ions is primarily limited by diffusion rates in sediment-water systems. Soluble phosphate anions are exceptions,[88,89] but their binding is probably dependent on the affinity of phosphate with various precipitates on the surfaces of clays, especially compounds of iron, aluminum, and calcium.[90]

Iron compounds in oxidized sediments

48. Iron can comprise up to several percent of the total composition of sediment.[10] Its ability to form different complexes of varying solubilities with a diverse group of chemical compounds gives iron a very important role in cycling heavy metals, phosphorus, and organic compounds in sediment-water environments. Many iron complexes are rapidly affected by changing Eh and pH conditions, resulting in continuous dissolution, migration, and precipitation of different iron compounds.[14] Additionally, many iron precipitates, especially those formed under oxidizing conditions, possess the properties of large surface area and charge, which give them the capacity to adsorb metal ions or complexes, phosphate, and organic molecules as well as bind with various inorganic and organic solid substrates.[91]

49. Sediments which lie beneath an aerobic water column display a thin yellowish-brown to reddish-brown surface crust colored by the precipitation of various oxidized iron compounds. Beneath this zone, biochemical decomposition by anaerobic microorganisms results in the formation of inorganic and organic sulfide compounds, giving these reduced sediments a greyish to black color. The grey color is mostly contributed by iron disulfide (pyrite), which is more prevalent in older sediments; blackish-colored sediments reflect mainly the presence of iron sulfide and other reduced iron complexes.[14] (The sulfide cycle will be discussed in a later section.)

50. Oxidized (ferric) iron is generally formed in sediments at Eh values more positive than about +200 mV and/or at pH values above 5.0,

mainly as highly insoluble ferric hydroxide ($Fe(OH)_3$), oxyhydroxide (FeOOH), and hydrated oxide complexes.[10,92] At lower Eh and pH values, iron tends to be released into solution as the reduced ferrous ion, which may then form stable compounds with sulfides, carbonates, or other chemical species. The concentration of iron in the interstitial water of reduced sediments is controlled primarily by the carbonate and/or sulfide concentrations in the sediments and also by the rate of upward diffusion of ferrous ions from the reduced zone.[14]

51. Iron rapidly precipitates in aerobic sediments and in an aerated water column, permitting only a trace of soluble iron to persist in most fresh and marine waters of low organic content.[10] Although iron oxidation reactions are rapid and chemically promoted in the presence of oxygen, limited microbial oxidation of iron can occur in almost the complete absence of molecular oxygen if more oxidized chemical compounds are present.[59] Increased solubility of iron found in nearshore waters is usually attributed to its affinity with soluble organic matter, especially humic substances.[93]

Element scavenging by iron oxides

52. The ability of ferric iron to adsorb or complex with trace metals and orthophosphate is an important feature in the elemental cycling in sediments.[14] Concentrations of many trace metals in sediment pore waters are often one to two orders of magnitude higher than those found in fresh[13,48] or saline[46,47] surface waters. The principal controlling factor for the above is the adsorption and coprecipitation of metal ions or metal hydroxides with iron hydroxide and hydrous oxide colloidal precipitates in the water column, or with similar ferric iron coatings on the surfaces of sediment particles.[94] Although the precipitation of these ferric compounds is rapid under aerobic conditions, the adsorption and coprecipitation of trace metals onto these precipitates is much slower. These processes depend on the diffusion or mixing rates, concentration of the given metal ion,[12] and competing reactions in the environment (e.g., dissolution rates from insoluble complexes or complexation with organic molecules).[55] Freshly

precipitated ferric hydrous oxides seem to be better heavy metal scavengers because of their large effective surface area.[94] Thus, dredging operations may accentuate the scavenging of trace metals by iron.

53. Considerable research has recently been initiated concerning the release of phosphate from sediments. Phosphorus has been implicated in stimulating water quality degradation by promoting excessive biological productivity. The importance of ferric complexes in preventing the release of phosphate, especially orthophosphate, from sediment or aiding in resorption of phosphates from the water column is well documented.[12,90,95,96] The release of phosphate from sediments is frequently closely correlated with the release of soluble iron, which is favored by low pH and/or Eh conditions.[90] Generally, if the entire water column above the sediment is kept aerobic (usually with a dissolved oxygen concentration of greater than 1 mg/ℓ) there will be a greater tendency for phosphate fixation rather than release.[13] Phosphate also seems to be adsorbed by sediments under reduced conditions when phosphate in the water column approaches very high concentrations (e.g., 1 to 2 mg/ℓ).[97] Thus, without significant external loading, phosphate concentrations in sediment-water systems should reach an equilibrium. The fixation of phosphate under anoxic conditions also appears to be facilitated by reactions with calcium, iron, and manganese compounds.[98]

Sulfide compounds in reduced sediments

54. The sulfur cycle in sediment-water systems is extremely important in regulating the mobility of trace metals. Soluble sulfate ions, which are in high concentration in seawater and in most estuarine environments, can be reduced in sediments to sulfides under anoxic conditions usually only at negative Eh values (e.g., -150 mV)[99] and after most of the ferric iron has been reduced. In freshwater environments, in the presence of low sulfate concentrations, various sulfur containing organic compounds may become important in sulfide formation.[14]

55. The reduction of sulfates to sulfides is primarily a microbial process accomplished by a small group of anaerobic bacteria present in reduced sediments. A generalized reaction scheme for *Desulfovibrio* species, an important sulfate-reducing genus of bacteria, involves the

oxidation of organic compounds to bicarbonate ions and the use of sulfate as an electron acceptor, resulting in its reduction to hydrogen sulfide.[14] The composition of the organic compounds is important since their rate and mode of degradation influence Eh, pH, and various competitive chemical reactions.[100] The bicarbonate ion produced helps to buffer the environment. This is important since many sulfate-reducing bacteria have narrow pH tolerances, preferring a near neutral pH.[99] Hydrogen sulfide is highly toxic to many aerobic organisms, and, if dissolved sulfides are not rapidly removed, benthic organisms may be affected.[101]

56. Transition heavy metals have a strong affinity with sulfides, forming highly insoluble complexes. Heavy metals which will readily form sulfide complexes include, in their usual order of decreasing solubility: mercury (II), copper (II), lead (II), cadmium (II), zinc (II), nickel (II), iron (II), and arsenic (II). Iron (ferrous) sulfide is by far the most abundant sulfide found in sediments. However, if iron is not plentiful in anoxic sediments, free sulfides may accumulate to toxic levels in the water column. Since iron sulfide forms relatively soluble sulfide complexes (although still poorly soluble), there are generally sufficient free sulfide ions in solution to precipitate most trace metals almost completely. However, if excess sulfide is present, mercury and arsenic may form more soluble polysulfide complexes.[102] Manganese is one abundant metal which does not readily complex with sulfide. (This will be discussed in another section.) There is always competition between free sulfides and organic complexes for trace metals.[103] High levels of soluble organics can greatly change the solubilility of trace metals even in environments generating large amounts of sulfide.[9]

57. Ferrous sulfide is an intermediate acid labile product in anoxic sediments. Over a period of time, free sulfide can be oxidized in aerobic environments to elemental sulfur,[104] which then combines with ferrous sulfide after slow migration back into reduced zones to form iron pyrite (FeS_2).[101] Pyrites can remain quite stable for

prolonged periods of time under aerobic conditions. However, they may eventually be oxidized to sulfate through bacterial action, producing acidic conditions in drained sediments.[10,105] Such a situation could occur when dredged material is placed on the land. Under acidic conditions, trace metals and iron are readily released from acid labile sulfides and carbonates. Thus, the formation of acid conditions in dewatered surface sediments can also lead to the release of contaminants in reduced sediments subjected to percolation of acidic groundwater. The impact that this acidity has on a given soil is dependent on the buffer capacity of the system. Buffer capacity is frequently controlled by the abundance of carbonate solids in the sediments.

Carbonate complexes

58. The carbon dioxide-bicarbonate-carbonate system is to a large extent controlled by biological activity. Equilibrium shifts within this system are pH dependent, with carbon dioxide (carbonic acid) tending to predominate under acidic conditions and carbonate forming at alkaline pH. However, the system is also affected by biochemical reactions. Anaerobic fermentation can release either carbon dioxide or bicarbonate ions, depending upon the types of organic matter and organisms involved.[100] For example, anoxic sediments are frequently buffered to near neutral pH, primarily by the preferential release of bicarbonate ions during sulfate reduction.[101] However, in the zone of accentuated biological activity, above the zone of intense sulfide formation, carbonic and organic acid formation can increase acidity. Thus, one often finds higher concentrations of soluble iron (II), manganese (II), trace metals, and calcium trapped beneath the oxidized crust.[106] These ions are limited in their upward movement to the sediment surface by precipitation in the oxidized crust. Precipitation is also favored in lower horizons because of the presence of a high bicarbonate ion concentration, coupled with increasing pH, which favors carbonate complexation.[101] Carbonate precipitation is also promoted by the microbial formation of methane gas, which selectively removes hydrogen from the bicarbonate ion as well as from some organic compounds in the reduced

sediments.[100] Heavy metals, because of their rapid precipitation with sulfide, especially show decreased mobility in the lower profiles.[101]

59. Metal-carbonate complexes are generally not predominant in environments containing large free sulfide concentrations, such as in anoxic marine or estuarine sediments, as sulfide complexation with most metal ions is favored. However, in the deep reduced profiles where the absence of sulfate ions limits sulfide formation, carbonates may help control sediment pore water ionic concentrations. In freshwater systems where sulfides are less prevalent, carbonate complexation with metals under anoxic conditions may become very important. Soluble iron concentrations in many anoxic freshwater sediments may be controlled by ferrous carbonate and/or bicarbonate equilibria.[14,106]

60. Precipitation of carbonates in the sediment profile can be related to biochemical reactions in the water column. For example, an organism can promote carbonate precipitation if carbon dioxide is selectively used instead of bicarbonate ion in carbon fixation (e.g., photosynthesis) reactions, or by increasing the pH through indirect reactions. Water with high alkalinity and high calcium ion concentrations is particularly susceptible to carbonate precipitation.[59]

Interactions of Chemical Species in the Water Column

Contaminant mobility at the sediment-water interface

61. Previous sections have discussed the net movement of inorganic and organic ions and compounds from the water column and sediment pore water to immobile phases associated with sediment solids. Some of these reactions (e.g., physical sorption) are readily reversed by only minor fluctuations in the environment. Chemical fixation with iron hydrous oxide compounds in oxidized sediments or the sulfide phase of reduced sediments requires often extreme and/or long-term environmental changes to facilitate release. Although drastic changes continuously occur in sediment microsites, the overall sediment system is usually quite static under natural, unperturbed conditions. Many mineral cycles in relatively undisturbed sediment-water systems tend to favor fixation, with usually only trace amounts of contaminants remaining in the water column.[107]

62. Physical disturbance at the sediment-water interface is probably the most rapid method to disrupt the physical and chemical equilibria associated with sorbed and precipitated contaminants. Environmental perturbations of significant consequence can occur without man's intervention. Wave and current action, stimulated by storms or seasonal temperature changes, can result in both a physical disturbance and chemical change at the sediment-water interface.[12,108] River systems are continuously subjected to sediment transport, deposition, and erosion, while nearshore areas may also experience rapid seasonal sediment deposition. Various benthic organisms, especially burrowing worms, can selectively concentrate and release contaminants as well as physically perturb the sediment surface.[109] Attached vegetation in shallow-water and marsh environments can also physically and chemically alter the sediment profile and influence the composition of the overlying water.[10,43,110-112] The activities of man simply intensify the natural processes.

63. The addition of nutrients, organic compounds, and various chemicals to surface waters may do more than just directly impact the environment. It may also indirectly accelerate the net release of contaminants from sediments by generating reduced conditions and/or significant changes in pH at the sediment-water interface. Additionally, if the bottom sediments are mixed in the water column, a net change in water chemistry can result. Changes in sediment pore water composition can eventually affect ionic concentrations in the water column, and the opposite is also true. These results occur in natural environmental cycles, which are accentuated by man's activities through a series of complex interactions.

64. Iron, manganese, many trace metals, ammonium and orthophosphate are among the soluble materials which are trapped within sediments by the presence of an oxidized surface layer at the sediment-water interface. If rapid sediment mixing results from dredging or other perturbations, the highly enriched pore water in the upper sediment profile could be released to the water column or would have potential mobility at the

disposal site. However, many dredging operations exist in environments which are continuously disturbed by current action or ship traffic (e.g., maintenance dredging in rivers and channels). In such situations, a discrete oxidized layer may rarely be present. Thus, pore water in surface sediments associated with most maintenance dredging could very likely have a composition very similar to the overlying water.

65. If the pore water does contain high concentrations of reduced soluble contaminants, the aerobic environment in most upland disposal areas should induce the precipitation of the iron, with coagulation and settling of the precipitate. Resorption of trace metals and other contaminants onto floc surfaces would be promoted, but these reactions could require an extensive period of time, with each compound perhaps reacting differently under existing environmental conditions. Although this major disturbance could seriously impact on the environment, the overall trend would eventually be toward mineral recycling and restabilization of the sediment-water regime.[107]

<u>The manganese dilemma</u>

66. Iron[113] and manganese[114] are elements which are both very common in sediments and also readily affected by changes in Eh and pH. Under reduced or acidic conditions, soluble iron (II) and manganese (II) may accumulate in sediment pore water at very high concentrations. The soluble ferrous iron in reduced sediments is readily complexed with sulfide, whereas manganese is not.[14] The solubility of manganese seems to be controlled by many complex reactions in anoxic environments, but precipitation with carbonate ions in environments having high carbonate alkalinity[115] and adsorption reactions with solids and on precipitates, especially at alkaline pH,[114,116,117] seem to be most important.

67. Iron will rapidly oxidize in the presence of oxygen at natural pH. However, manganese will not be readily oxidized if ferrous iron is also present. This is because the manganese ion has the potential to accept electrons from the reduced iron, and iron will thus be preferentially oxidized.[14] When the upward movement of soluble manganese is controlled by slow diffusion processes in sediment, poorly soluble

manganese oxides readily precipitate in the aerobic environment at or near the sediment-water interface. Continuous reduction and oxidation processes are always occurring because of constant sediment accretion, and thus soluble manganese tends to remain in highest concentration immediately below the oxidized crust.[115,117,118] Therefore, if the upper sediment profile is disturbed during dredging, soluble manganese can be rapidly released in high concentrations.

68. Soluble reduced manganese in the water column will tend to be oxidized at a much slower rate than iron, and the presence of reduced iron or organic compounds may act to further retard its oxidation and precipitation.[14] The very small size of fresh iron and manganese precipitates prevents their rapid settling in containment areas. Therefore they could be readily released in effluent discharges. Also, the iron and manganese flocs can be redissolved if they settle in a reduced environment within the disposal area, thus releasing any sorbed ions. Precipitate dissolution could be accentuated if extensive biological activity develops within the disposal area water.[36,37]

Ammonium and phosphate release

69. The cycling of nitrogen in both freshwater and brackish water environments is important for the growth of both desirable and undesirable biota. Nitrogen has been implicated in creating eutrophic conditions in water systems, but only in brackish water environments, where phosphate is usually in abundant supply, can good correlations be frequently made.[119] In aqueous environments where the pH exceeds 8.5, which is possible in some photosynthetically active lakes, estuaries, or confined disposal areas, an appreciable quantity of ammonium ions (NH_4^+) begin to revert to free ammonia (NH_3), which is very toxic to many organisms.[120] The following discussion will emphasize only those aspects of nitrogen cycling which allow for the accumulation of ammonium nitrogen in sediment pore water. A more detailed discussion of the subject is available in the literature.[120-123]

70. The majority of the nitrogen in lake, river, and estuarine sediments is associated with organic matter, and the highest concentrations exist near the sediment surface, where organic debris accumulates

from the water column. In a cyclical manner, soluble ammonium can be released from organic nitrogen compounds during microbial decomposition. The ammonium ion, which is by far the most abundant combined inorganic form of nitrogen in sediments, can diffuse slowly upward to the sediment-water interface, where it can be microbially oxidized to soluble nitrate ions, but only in the presence of dissolved oxygen. Nitrate can then be either reduced again to ammonium and converted into new organic compounds by many organisms or microbially converted to nitrogen gases (e.g., nitrogen, nitrous oxide) by denitrification pathways, resulting in nitrogen loss to the atmosphere. Denitrification occurs only in anaerobic environments and nitrate formation proceeds only in aerobic areas; thus, ammonium cannot be converted to nitrogen gases unless it first enters an aerobic environment. Therefore, if ammonium diffuses out of reduced sediments and into an aerobic sediment horizon or water column, the total nitrogen in the system will more than likely decrease because of denitrification. Ammonium is subsequently stable in anaerobic sediments.[121]

71. It should be kept in mind that nitrogen loss from anaerobic environments is promoted by aerobic-anaerobic cycling, owing to the high mobility of the nitrate ion and development of an aerobic-anaerobic double layer at the sediment-water interface. However, ammonium is readily adsorbed by sediments and a considerable quantity of nitrogen in anaerobic sediments can be fixed in this manner.[10] Additionally, microbial decomposition and assimilation of organic matter are much slower under anaerobic conditions, a phenomenon which results in a similar lower microbial uptake of nitrogen.[121,122] Anaerobic sediments are also known to contain a much lower biomass than aerobic sediments.[51] These factors help to explain why sediment pore water can contain up to 250 mg/ℓ ammonium,[124] even though concentrations in the water column are significantly less in most instances. When the sediments are physically disturbed, fixed ammonium may be released as a result of environmental changes; also, the mixing allows for more rapid mobility of pore water ammonium to the overlying water column than does the very

slow diffusion process. The oxidation of ammonium to nitrate in confined disposal areas should rarely result in excessive nitrate concentrations because the rapid accumulation of reduced sediments should prompt denitrification in the bottom water, resulting in a net loss of nitrogen.[121]

72. The release of phosphate from sediments has already been indirectly discussed while considering its mode of fixation. Namely, the oxidized iron phosphate precipitates in sediments will release phosphate as the complex becomes unstable under increasingly more reducing conditions. Phosphate can also be released from decomposing organic matter under both aerobic and anaerobic conditions, although oxidized sediments will scavenge much of the soluble phosphate. Increased biological productivity in the water column can promote phosphate release from sediments,[125] although this seems to be indirectly induced by the creation of anaerobic conditions in the water column, thus solubilizing iron phosphate complexes.[126] Because of the slow oxygen diffusion through water, a dissolved oxygen concentration of 1 to 2 mg/ℓ at the sediment-water interface may be required to prevent reduction of the oxidized crust and resultant release of orthophosphate.[13] Phosphate release thus follows closely to that of manganese and iron; disturbance of the surface sediment, such as during dredging, will release the orthophosphate trapped beneath an oxidized crust.

Suspended solids interactions

73. Suspended solids in natural environments consist primarily of very small particles which, because of their large surface area to volume ratio and charge characteristics, can remain in the water column for extensive periods of time.[127] Such particles are said to have colloidal properties and may consist of clay minerals, inorganic precipitates, and organic matter, usually less than 10 μm in diameter.[128] Nearly all of the turbidity associated with low-energy regimes, such as in lakes or estuaries, consists of colloidal particles.[22] However in high-energy regimes, such as in rivers and ship channels, a significant part of the turbidity can be attributed to larger silt- and sand-sized suspended

solids. Since turbidity is an optical measurement, it tells little about the actual size or mass of the suspended solids.[22]

74. Generally, the amount of solids which a given water column can keep in suspension is dependent upon the energy supplied, usually through turbulent mixing. If the energy is dissipated through friction, solids will tend to deposit.[22] These rules are applied to dredging and disposal operations; in confined disposal areas various internal dikes, compartments, or the presence of vegetation can reduce the energy in the system and promote the settling of solids.[6,129] Solids in influents from hydraulic dredging may vary from less than 50 g/ℓ for sandy material to 350 g/ℓ for silt- and clay-sized materials.[129]

75. In the previous discussions, emphasis was placed on the importance of the degree of sediment oxidation and reduction in cycling contaminants in sediment-water environments. The same importance can also be placed on suspended solids. Most naturally, suspended particles are already in a highly oxidized state because they usually have been exposed to an aerobic water column for long periods of time. In contrast, dredged sediment-water slurries are most often obtained from highly reduced, oxygen-free environments. Once in the disposal area, these sediments are usually slowly oxidized, and during residence the turbid water may be observed to change from a dark color to a light brownish or orangish color. This change in color generally indicates that iron oxidation is occurring during slurry confinement, along with other more subtle chemical reactions. Thus, certain contaminants may subsequently revert to different complexes. These changes could modify the mobility and toxicity of a given element or compound.[14]

76. Research concerning changes in the different chemical phases with which elements or compounds become associated during sediment dispersion has been minimal. Most studies examine either the soluble phase or the total sample, but rarely is a detailed analysis of the solids attempted.[23] Studies concerning the association of elements and chemical compounds with the solids in rivers indicate that adsorption and ion exchange reactions may be important in contaminant transport in

the water column.[20,21,23,130] Dissolution of carbonate precipitates with changes in the ionic concentration, especially the alkalinity of the water, may also be important, even where the metals are associated with crystalline carbonate particles.[20,23]

77. Dispersion and resedimentation studies of initially reduced sediments under controlled oxidation-reduction and/or pH conditions suggest that cycling between ferric hydrous oxide solids and insoluble sulfides are important reactions for both suspended solids and bottom sediments.[17,18,131] Organic chelation reactions also seem to be very important,[9,16] but the levels of total organic carbon are generally not directly correlative to the degree of chelation.[16]

78. Dispersion and resettling experiments indicated that iron and manganese are released under anaerobic conditions,[16-18,132] that precipitation of iron in an aerobic water column is rapid,[18] and that adsorption of trace metals and orthophosphate onto these precipitates is a viable means of contaminant removal during dredging and disposal operations.[132] The slow oxidation of manganese[16-18] after release to the water column may be due to the formation of more soluble intermediate products, such as hydroxides.[55] Cadmium appears to become more mobile under more oxidizing conditions in both fresh and marine environments[16-18] with a concomitant increase in exchangeable phase cadmium.[18] Association of cadmium with highly insoluble sulfides is strongly inferred[17,18,132] although organic complexation[18] and rapid oxidation of cadmium sulfide in the presence of oxygen[55] are possible mechanisms which explain its high mobility.

79. Many trace metals in disturbed marine environments appear to be more readily mobilized from solids under conditions of rapidly increasing oxidation,[55] while there is a trend toward greater trace metal mobility under reducing conditions in the absence of high sulfide levels, such as in disturbed freshwater environments.[18] However, there appear to be many site specific variances in the fixation and release of trace metals. Ammonium and phosphate releases in high concentrations from dispersed anoxic sediments are well documented.[16-18, 132]

80. The association of contaminants with suspended particles of different sizes can also be important, not only because of the direct relationship between particle size and settleability but also because solids of different sizes and densities may have different exchange capacities, be composed of different precipitated compounds, or consist of different crystalline phases of the same compound. Increased crystallinity of a precipitate tends to result in a net decrease in surface adsorption of ions by decreasing the surface area, but it also decreases the tendency of the precipitate to resolubilize.[14] Comparisons of different suspended particle sizes with their metal content have been obtained for ocean discharges of wastewater, with some interesting correlations.[133] Although it is presently unknown what relationship these particulates have to effluent particulates from confined dredged material disposal areas, this study showed a poor correlation between turbidity or other gross solids determinations and contaminant availability.

81. Dredging and disposal operations can have a significant influence on the migration and transformations of chlorinated and petroleum hydrocarbons as well as other synthetic organic compounds. Sediments suspended during dredging can increase levels of PCB's and similar solids-sorbed contaminants in the water column.[134] However, in the high-energy regime of rivers and channels, background suspended solids will often mask the input directly caused by the dredging operation.[135] The release of sediment-sorbed organic compounds from confined disposal areas should be directly related to the solids content of the effluent. However, this relationship may correlate better with surface area of the suspended particles than with total solids measurements,[74] and different organic molecules may be preferentially bound to different particle-size or density fractions.[133] Upland disposal of dredged material may increase the rates of transformation or govern the products formed from recalcitrant compounds through increased mixing, aeration, emulsification, and other interactions. However, little information

exists about what these changes might be.

Colloid flocculation

82. Dispersed clays, inorganic precipitates, and fine organic particles can form larger particles (flocs or agglomerates) through the interaction of their net surface or internal electrostatic charges. Mechanisms governing suspended particle flocculation and agglomeration include adsorption of inorganic ions or large organic molecules for charge neutralization, adsorption site masking, and/or adsorption interparticle bridging; formation of chemical precipitates on particulates, thus increasing their size and changing their charge characteristics; and suppression of the thickness of the layer of hydrated ions (double layer) surrounding the particles.[84, 136] Most suspended colloids can be flocculated by brackish water salinity, mainly through the influence of the ionic concentration on charge neutralization and suppression of the double layer of hydrated ions. However, the composition of the particulate fraction and salt medium is very important.[136] Dissolved humic matter may keep clay particles in a dispersed state under conditions conductive to floccualtion, or it may act to bind and aggregate particles.[9] The interaction of organic polymers with clay particles is very complex, and the effect of a given organic compound may vary drastically with just a small change in concentration.[43] Flocculation of metal and phosphate compounds and organic matter can occur when water of low ionic concentration enters a high-salinity regime.[137] Changing electrolyte and dissolved organic (humic) concentrations readily occur in upland dredged material disposal areas, and the resultant effects are probably just as unpredictable as they are in natural systems.

PART III: EXPERIMENTAL METHODS

Disposal Area Locations and Descriptions

83. Nine different confined upland disposal areas were sampled during the period 2 October 1975 to 20 May 1976. The general locations of each site, a site description, and dates on which samples were obtained are given in Figure 1 and Table 1. (Future number references to sites listed in this Part will be the same as those given in Figure 1.) Dredging and field sampling logs for each of the disposal sites are presented in Appendix A.

Sayreville, N. J.

84. Generalized and detailed maps of the dredged material disposal site near Sayreville, N. J., are given in Figure 2. The area is located about ½ mile down the Raritan River from National Lead Industries' titanium oxide plant, and it has been designated National Lead Industries disposal area No. 4.

85. The 44-acre disposal site is divided by spur and cross dikes into three compartments. The north side of the site is bordered by the Raritan River, and approximately 9 ft of dredged material had previously accumulated within the site. Sampling was conducted on 2 October 1975.

86. Approximately 462,000 yd^3[*] of in situ sediment was hydraulically dredged from a brackish water area in the lower Raritan River during the period 10 July to 2 October 1975. About 387,000 yd^3 was dredged from the south channel (mile 5) near the National Lead Industries plant; the remaining 75,000 yd^3 was dredged concurrently from the company's private dock basin. Dredged material was transported directly to the site by a 16-in.-ID pipeline from a cutterhead dredge, which maintained an average daily influent slurry volume of 5,947 yd^3. The disposal area had been previously used during October to November 1973.[**]

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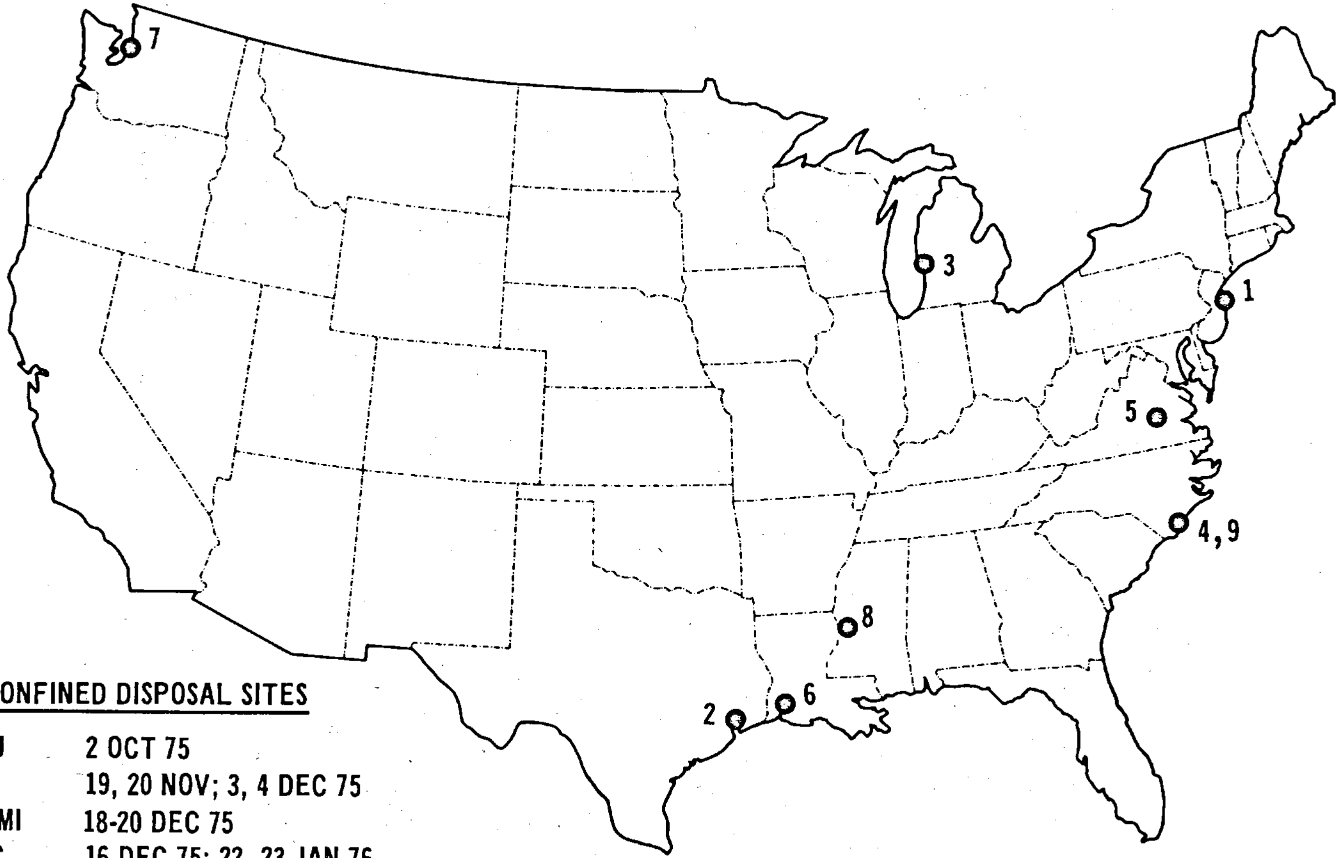
[*] The following U. S. Customary Units will be used in this report because it is considered that these will be more meaningful to the general reader than metric units.

[**] Personal communication, Thomas Clark, U. S. Army Engineer District, New York, August 1977.



SAMPLED CONFINED DISPOSAL SITES

1. SAYREVILLE, NJ	2 OCT 75
2. HOUSTON, TX	19, 20 NOV; 3, 4 DEC 75
3. GRAND HAVEN, MI	18-20 DEC 75
4. WILMINGTON, NC	16 DEC 75; 22, 23 JAN 76
5. RICHMOND, VA	27 JAN; 19-21 FEB 76
6. LAKE CHARLES, LA	4-6 FEB 76
7. SEATTLE, WA	16, 19, 22, 23 MAR; 1, 3-6 APR 76
8. VICKSBURG, MS	17 MAR; 12, 15 APR 76
9. SOUTHPORT, NC	6, 7, 17, 19, 20 MAY 76

Figure 1. Locations of disposal areas and sampling dates

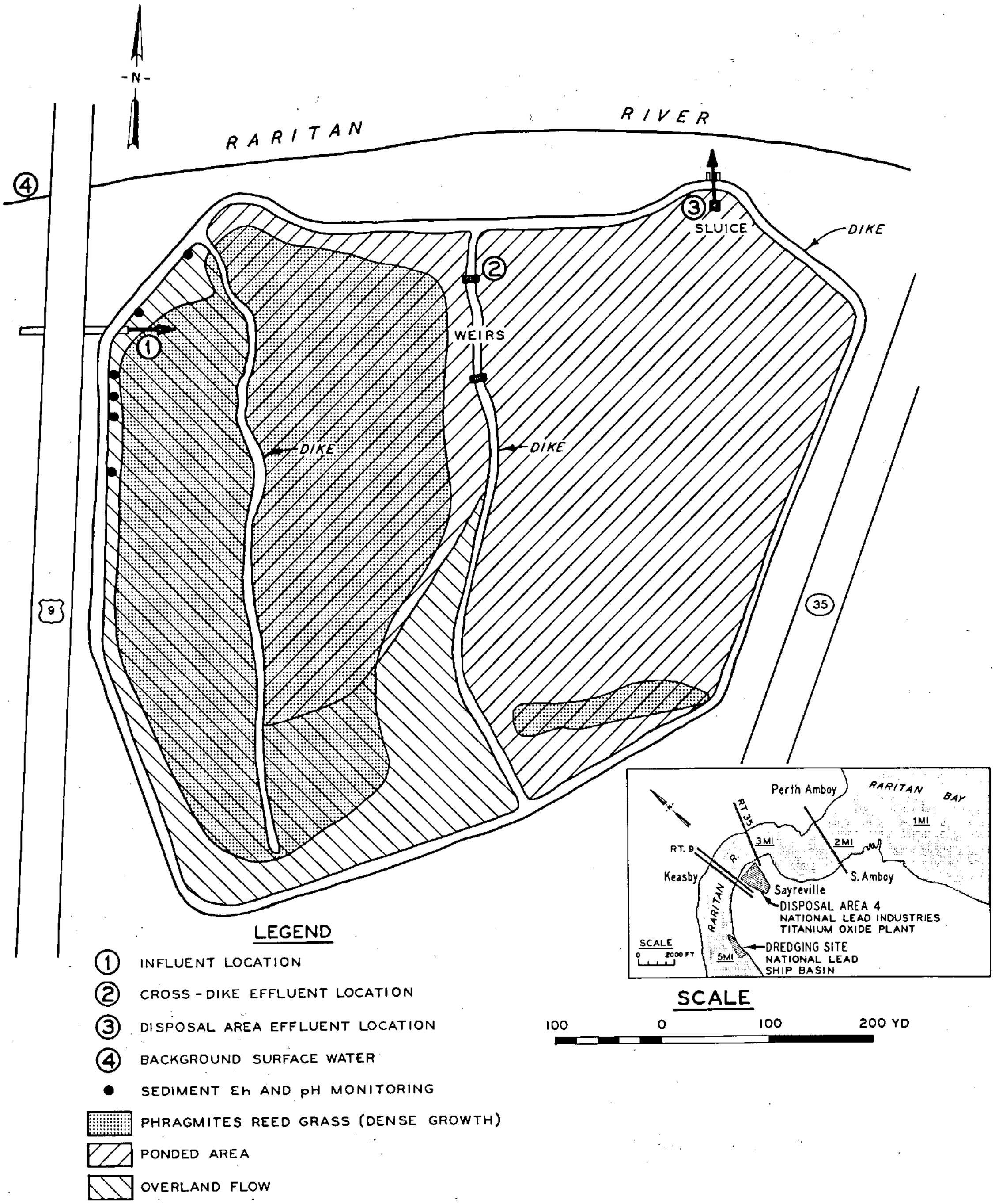


Figure 2. Sayreville, N. J., disposal area

87. Influent was discharged into the northwest corner of the first (westernmost) compartment, and effluent was released back to the Raritan River from a square sluice box in the northern section of the third (easternmost) compartment, after the supernatant topped two 6-ft-wide rectangular weirs. The disposal area was covered with thick pure stands of the common reed grass Phragmites communis. Table 2 gives a detailed description of each compartment of the disposal area, with reference to the slurry movement and vegetation density.

88. Generally, overland flow occurred in compartment 1, with most of the slurry flattening the vegetation in a path along the spur dike. In compartment 2, there was mainly shallow ponding in thick vegetation, as shown in Photo 1. In compartment 3, deeper ponding occurred in very sparse, almost completely buried vegetation (Photo 2).

89. Samples were collected from within the disposal area about 5 hours after dredging had ceased. Therefore, a composite influent water sample was collected from a pond beneath the discharge pipe, and a saturated sediment sample was collected from around the perimeter of the above pond. Slurry samples were also collected at a cross-dike weir between compartments 2 and 3 and from inside the sluice box in compartment 3 (disposal area effluent); additionally, sediment was collected adjacent to the sluice in compartment 3. Flow rates from compartments 2 and 3 were estimated at 200 and 600 gpm, respectively.[138] Residence time or the slurries could not be estimated because of continual removal of weir boards and uneven drainage patterns in the thick vegetation. A background surface water sample was collected about 600 yd up the river from the effluent discharge.

Houston, Tex.

90. The Clinton disposal area in Galena Park, a suburb of Houston, is depicted in Figure 3. The detailed map shows the entire 560-acre site, although only the eastern 280-acre section (cross-hatched area) was being used during sample collection. Samples were obtained on 19 and 20 November and 3 and 4 December 1975.

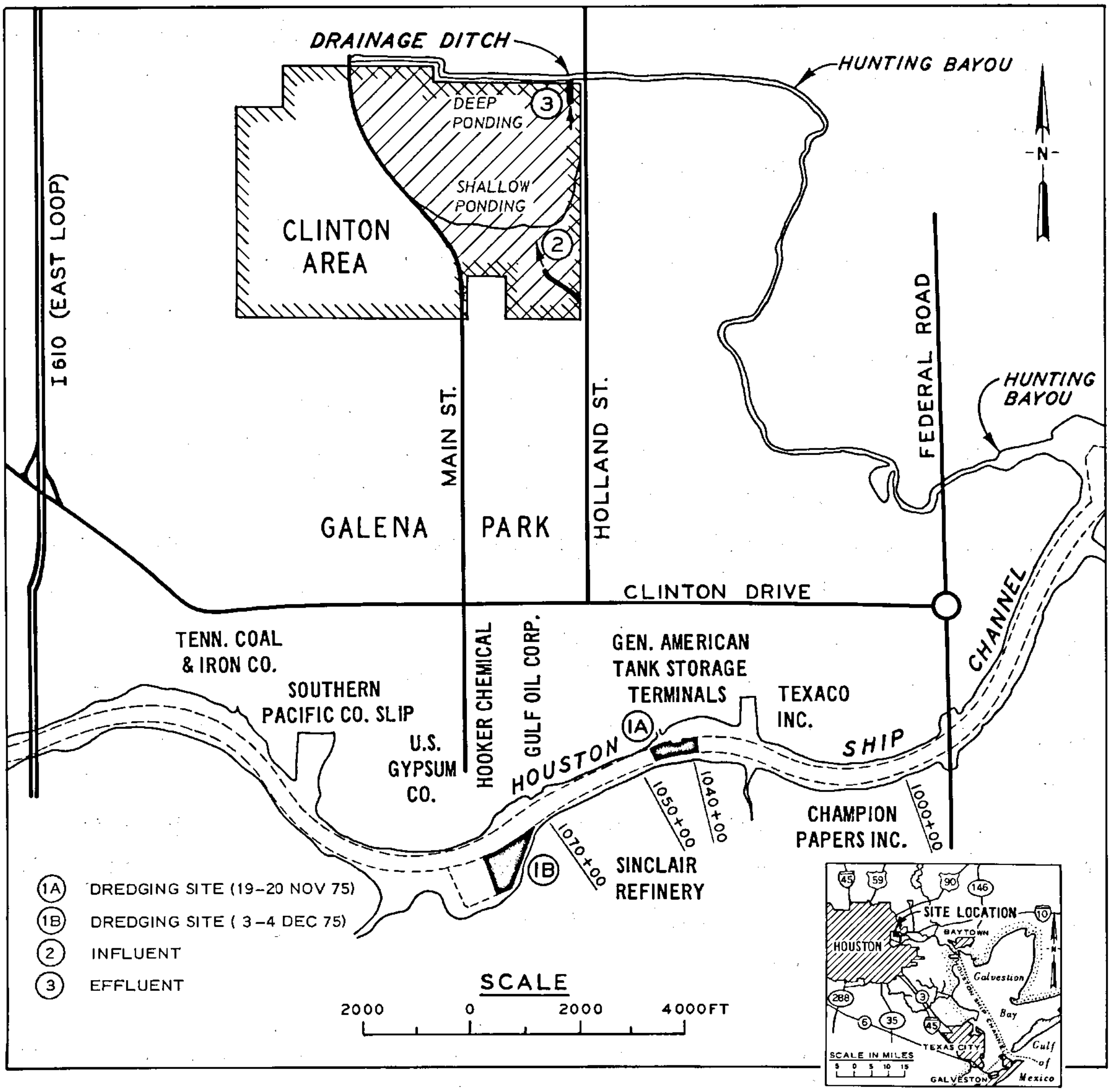


Figure 3. Houston, Tex., disposal area

91. Material from a maintenance dredging operation was being pumped over 1 mile from the brackish water Houston Ship Channel to the disposal area by a cutterhead hydraulic dredge. Influent entered the disposal area by a 30-in.-ID pipeline and flowed about 200 yd over old dredged material, consisting of poorly sorted oily sediment and clay balls, to an approximately 200-acre shallow pond, as shown in Photo 3. Effluent was discharged into a rectangular sluice over two boards, each 14 ft long. Effluent flow rate was supposed to be regulated, permitting flow through a 4-in. gap between the weir boards, with the upper board functioning as a surface skimmer. This would have permitted an average effluent discharge of about 8,000 gpm.[138] However, this flow rate was exceeded during part of the sampling. Information about the influent during the sampling period is included in Appendix A, Table A1. The average influent flow rate of 22,500 gpm[*] seems excessive in comparison to the effluent average. Based on a change in the color and consistency of the dredged material, a residence time of less than 14 hours was estimated for the surface water. This short period of residency was probably prompted by the shortcircuiting of flow along the eastern margin of the site; deeper ponded water probably was replaced much less often due to its greater density. The disposal area was only sparsely covered with vegetation, with most of this being along the western margin, and it probably contributed little to disposal area function. A picture of the ponded area within the site is shown in Photo 4.

92. Effluent was discharged into a ditch, and after a lengthy course through Hunting Bayou, finally returned to the Houston Ship Channel about 2 miles from the dredging site. A surface background water sample was collected on 2 January 1976 in the ship turning basin at wharf No. 10, located 4.5 miles up the Houston Ship Channel from the dredging site. Freshly dredged sediment along the eastern margin of the pond in the containment area was monitored for Eh and pH.

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[*] Personal communication, William Humphreys, T. L. James Company, Kenner, La., August 1977.

Grand Haven, Mich.

93. The Grand Haven disposal area (Figure 4) is located on the north shore of the Grand River, about 2 miles east of Lake Michigan. The site encompasses a diked area of about 6 acres located on the property of Verplank's Coal and Dock Co., in Ferrysburg, Mich. The irregularly shaped site is about 250 yd long and lacks internal dikes or compartments. The substrate consists of sandy dredged material and buried solid waste, mainly the parts of old automobiles and scrap metal (Photo 5).

94. The dredging of about 23,900 yd^3 of in situ sediment from the Grand River (harbor area below the U. S. Highway 31 bridge) was conducted from 13 to 19 December 1975 by the Detroit District's hopper dredge HAINS. The daily dredging log during the sampling period is given in Table A2. At the time of sampling, from 16 to 19 December 1975, the disposal area was covered with ice, which prevented sample collection at the influent pipe. Therefore, samples were obtained at the inflow pipe into the hoppers of the dredge.

95. Influent discharge from the dredge hopper took over half of the 2-hour dredging and disposal period. Occasionally dredging had to cease due to freezing of the influent discharge pipe. The nonvegetated disposal area appeared to be almost completely ponded (Photo 6), and, although residence time was not determined, it was probably in the range of a few hours for most of the slurry. Effluent was discharged back into the Grand River after flowing over the four sides of a square sluice box measuring about 4 ft on each side. The discharge rate, based on one measurement on 17 December, was estimated to be about 5,000 gpm with the aid of a dye marker.* Influent discharge was calculated to average about 2,500 gpm over a 24-hour time period based on data in Appendix A, Table A2. This indicates that effluent discharge probably fluctuated greatly in volume, although influent volume figures could be in error as a result of underestimating the volume of hopper wash water used. A background

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*Personal communication, Bruce Sabol, U. S. Army Engineer District, Detroit, January 1976.

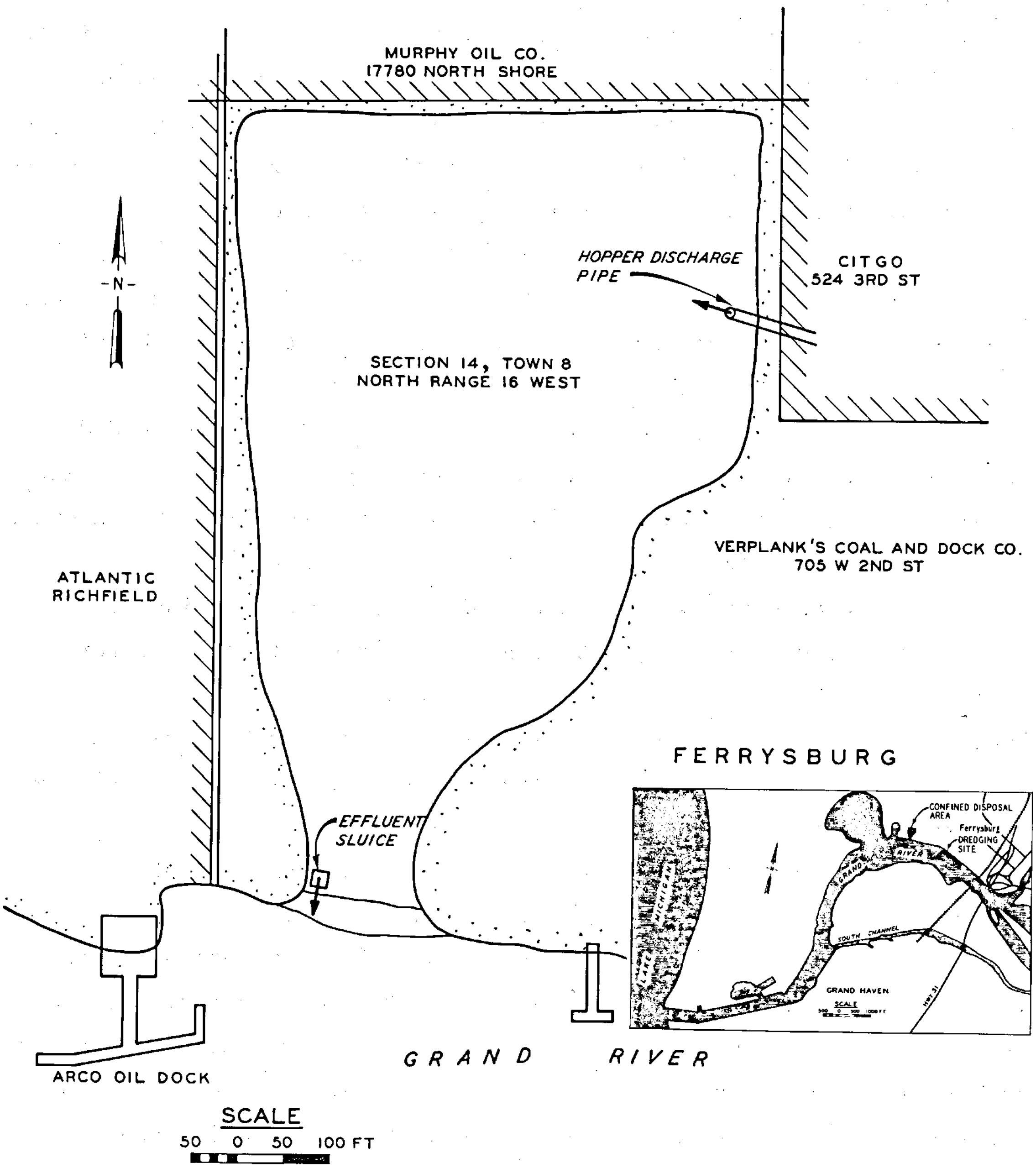

Figure 4. Grand Haven, Mich., disposal area

surface water sample was also obtained about 200 yd up the river from the effluent discharge area. Previous disposal onto the site took place during March 1975.

Wilmington, N. C.

96. The dredging and disposal operation monitored near Wilmington, N. C., involved hydraulic dredging of a brackish water section of the Cape Fear River in the Wilmington Anchorage Basin channel and dock area, and disposal in a large 525-acre confined disposal area on Eagle Island. An area map of the site is shown in Figure 5, and a detailed map of Eagle Island and the dredging locations is shown in Figure 6. Three sample collections were made on 16 December 1975 and 22 and 23 January 1976. Background surface water samples were collected from the river channel area on each of the two trips to the site.

97. Dredged material was removed by a hydraulic cutterhead dredge and transported directly to the disposal area by a 16-in.-ID pipeline. Two different influent discharge locations were used during the sampling period. Initially (16 December), effluent was obtained from leaks between weir boards as the Wilmington District was attempting to fully contain the effluent slurry, but samples obtained during the later sampling trip (22 and 23 January) were composited from three weir overflows. Two of the weirs were D-shaped spillways with 3.5-ft-wide variable-height board faces. The other structure was a square sluice box with two rectangular spillways, each 12 ft wide.

98. The Eagle Island disposal area is a good example of a large overland flow system, using vegetation to disperse the slurry flow velocity and promote solids settling. About 75 to 80 percent of the site was being used in dredged material treatment, with overland flow covering an area of close to 300 acres and with flow distances frequently reaching 1800 yd. A view of the overland flow area is shown in Photo 7. Less than 15 percent of the diked area consisted of shallow ponding, with deep ponding being observed primarily in the borrow pit adjacent to the dike (Photo 8). On the last day of sampling, sediment had accumulated to very near the rim of the weirs. Thick vegetation covered essentially

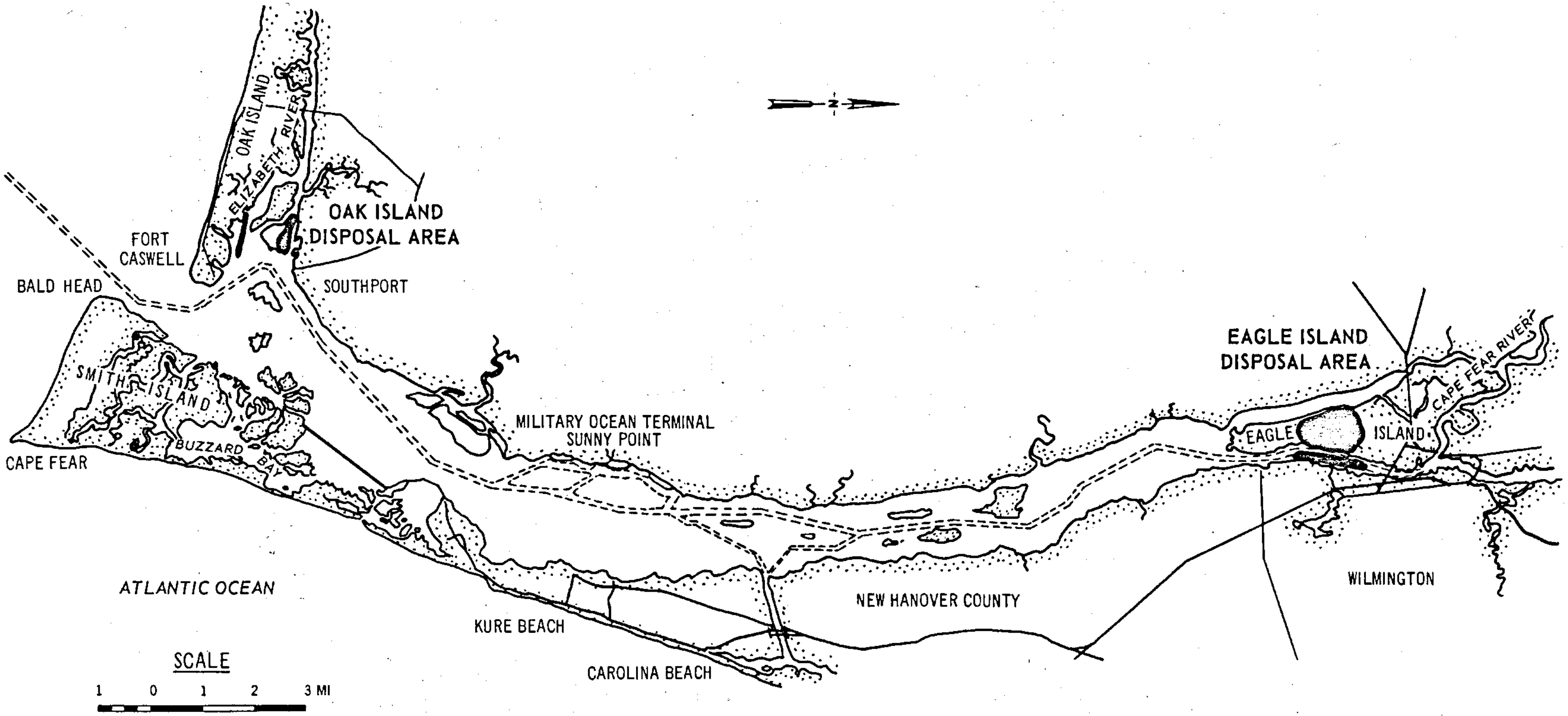

Figure 5. Vicinity map of Wilmington and Southport, N. C., disposal areas

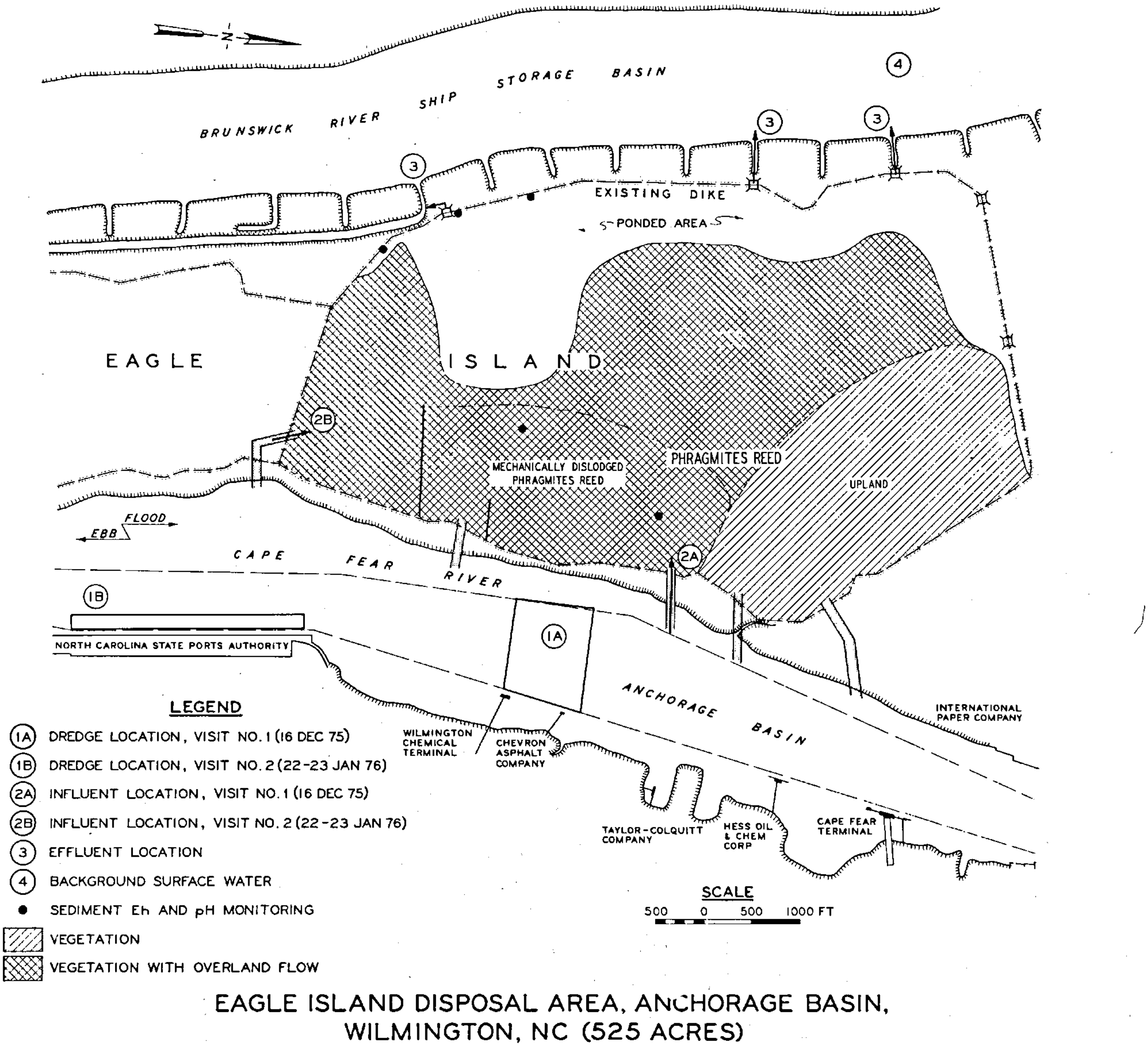

Figure 6. Wilmington, N. C., disposal area

all of the disposal area, but about 20 percent of the vegetation, especially near the influent discharge pipes, had been flattened with heavy machinery. About 90 percent of the vegetation consisted of dead stems and leaves of dormant Phragmites communis; the other emergent plants present were mainly perennial salt-tolerant bushes and trees.

99. The Eagle Island disposal operation was conducted from 30 October 1975 to 23 January 1976, with a total of 947,000 yd^3 of in situ dredged material being excavated. During the 84 days of active dredging, an average of about 755 yd^3 of in situ sediment was pumped each hour for a total of 1253.5 operational hours.* Available detailed dredging information for the days sampled is listed in Appendix A, Table A3. Based on available information, influent and effluent flow rates and slurry residence time in the disposal area could not be estimated. However, residence time was probably in the order of weeks, because of the attempt to totally contain the dredged material.

Richmond, Va.

100. Dredging was conducted in the ship channel on 27 January and 19 and 20 February 1976 and alongside the docks on 21 February 1976 at Deepwater Terminal in the James River just south of the city limits of Richmond. The river at this site experiences tidal fluctuations but is not affected by saltwater intrusions. Dredged material in the channel is mainly sand and gravel, whereas fine-grained sediments are found in the dock area. A three-compartment, 70-acre confinement was used for upland disposal. Two background surface water samples were obtained from the channel area of the James River, and the disposal area was sampled daily on 27 January and 19 to 21 February 1976. Figure 7 depicts the dredging and disposal sites.

101. A total of 281,600 yd^3 of in situ sediment was dredged from Deepwater Terminal between 26 January and 6 March 1976, during 30.7 days of active dredging. Due to the predominantly coarse sediment texture and use of a low-powered hydraulic pipeline dredge, only about

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* Personal communication, Barry Holliday, U. S. Army Engineer District, Wilmington, January 1978.

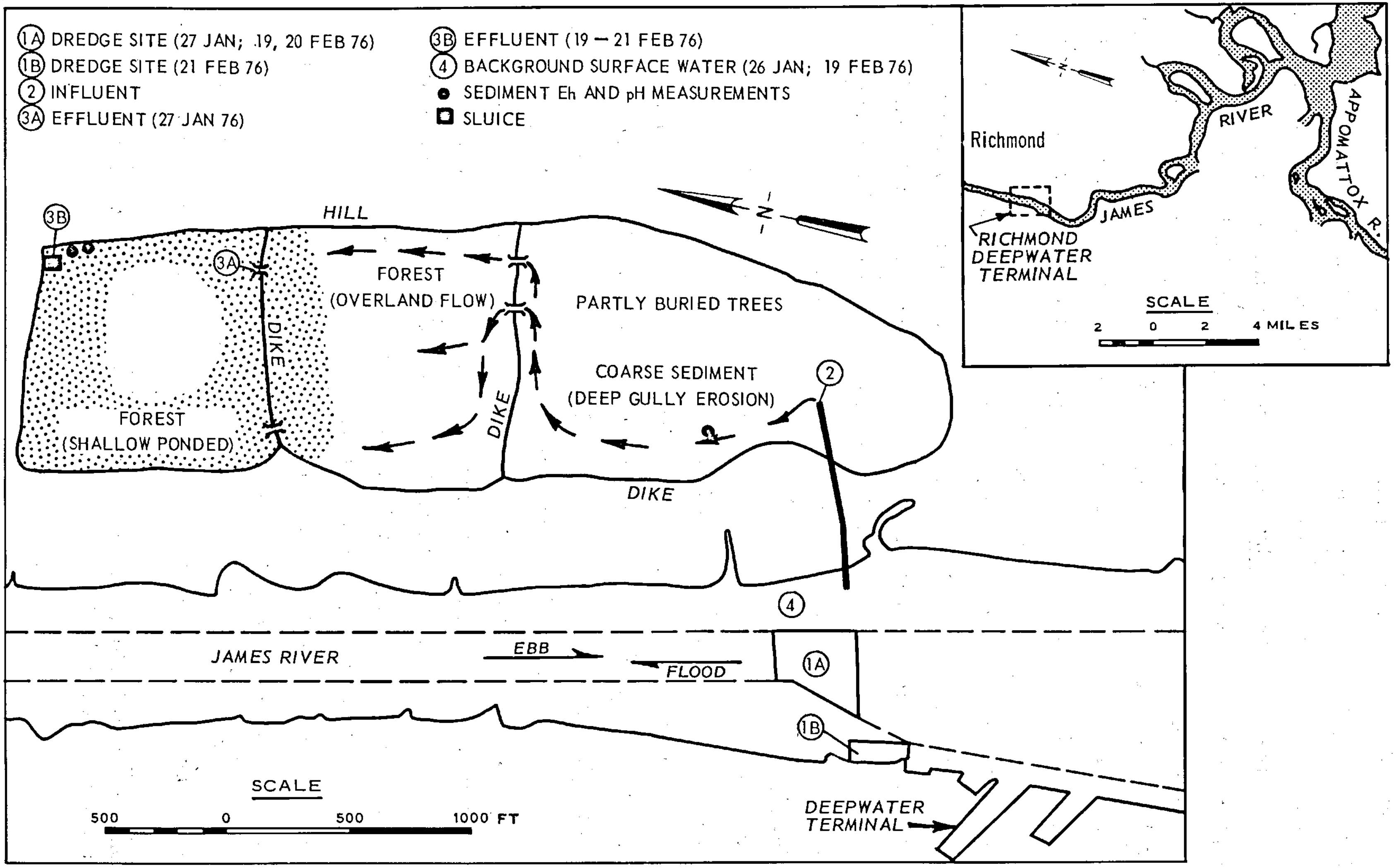

Figure 7. Richmond, Va., disposal area

385 yd^3 of in situ sediment was dredged per operational hour.* The dredging log for the sampling period is given in Appendix A, Table A4. Influent was discharged through a 16-in.-ID pipeline which was equipped with a terminal splash plate about half of the time to help disperse the solids and prevent bed erosion.

102. The first (southernmost) compartment was filled with coarse-grained dredged material which had mounded near the influent pipe and resulted in gullying and turbulent overland flow through a forested area as shown in Photo 9. Only the trees extended above the sediments which had accumulated during more than 20 years of disposal area use. About 3 acres of compartment 1 were covered by the slurry.

103. Less than 75 percent or 15 acres of compartment 2 was covered by the dredged material slurry, and about two thirds of this area consisted of overland flow through a thick undergrowth of primarily small dormant trees and sparse to moderately thick dormant undergrowth. There was only minimal observed interaction between the forest vegetation and the slurry because of the small surface area covered by the plants. Effluent from compartment 2 flowed from a shallow ponded area near the second cross-dike over two weirs, each 10 ft wide. On 27 January, a composite effluent sample was collected from these weirs (see Figure 7), as discharge into the disposal area was temporarily curtailed at this time.

104. Approximately 75 percent or 15 acres of compartment 3 consisted of shallow ponding, with almost no overland flow (Photo 10). Most of this area was covered with sparse forest vegetation. The three composite effluent samples collected on the second trip (19 to 21 February 1976) were obtained at the final sluice box, which had two 14-ft-wide rectangular weirs in use. The average effluent discharge at the sluice box, considering an observed weir head of 4 in. during sampling, was approximately 8,000 gpm.[138] Although no influent rate or

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* Personal communication, Rivers Wescott, U. S. Army Engineer District, Norfolk, August 1977.

accurate residence time could be computed, slurry residence was probably no more than a few hours as a result of poor disposal area use and limited ponding. Nevertheless, the effluent was usually quite clear because of the small volume and coarse texture of the influent dredged material.

Lake Charles, La.

105. The dredging and disposal sites are located about 10 miles down the Calcasieu River from Lake Charles, at the northwestern end of Lake Calcasieu. The disposal area has been designated disposal area No. 22 on the Calcasieu River channel by the New Orleans District, and it can be approached by Highway 27 near Hackberry, La. (See the generalized map in Figure 8). The 185-acre containment area is located on a dredged material island, part of a chain that separates the ship channel from the lake, and it must be reached by boat. Although the islands are generally thickly vegetated with upland shrubs and trees, disposal area No. 22 is only sparsely covered with plants, mainly along the higher elevated western section adjacent to the channel. Sediments in the disposal area are well mixed both horizontally and vertically, with the major part being fine-grained sand mixed with silty layers. Extensive mounds of weathered clay balls are present on the upper western slopes. The area contains no internal dikes. Samples were collected daily from 4 to 6 February 1976, and one background surface water composite was obtained in the ship channel as shown in Figure 8.

106. The amount of in situ sediment discharged into disposal area No. 22 could not be separated from the rest of the large dredging contract. However, dredging log information for the days sampled is given in Appendix A, Table A5. Based on this, about 2,000 yd^3 of in situ sediment was being dredged per operational hour. Influent discharge from the hydraulic cutterhead dredge was through a 30-in.-ID pipeline. The influent slurry then flowed down an eroded gully for a distance of about 300 yd at the northern corner of the site.

107. About 75 percent of the containment area was ponded as pictured in Photo 11. However, on the first sampling day (4 February),

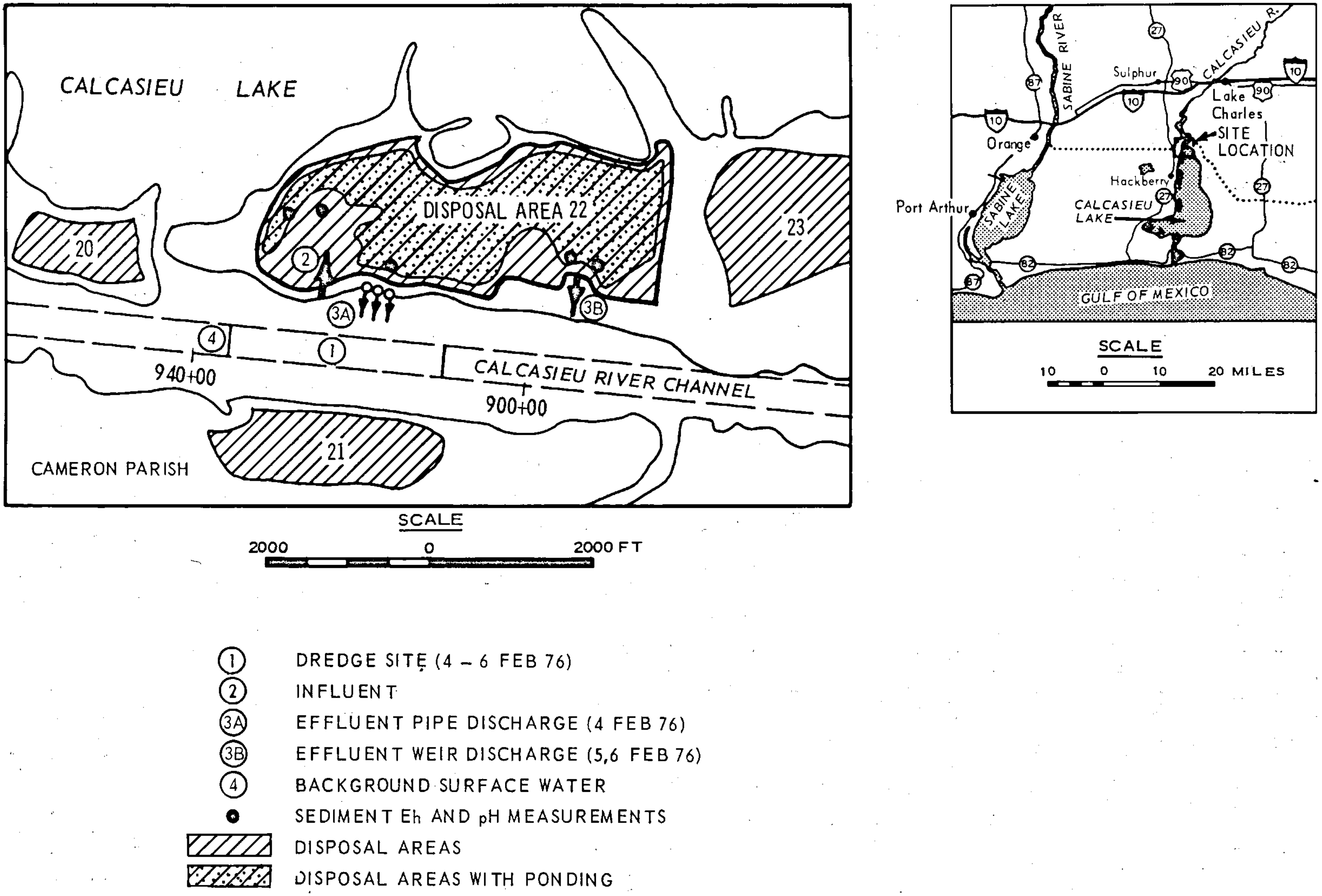


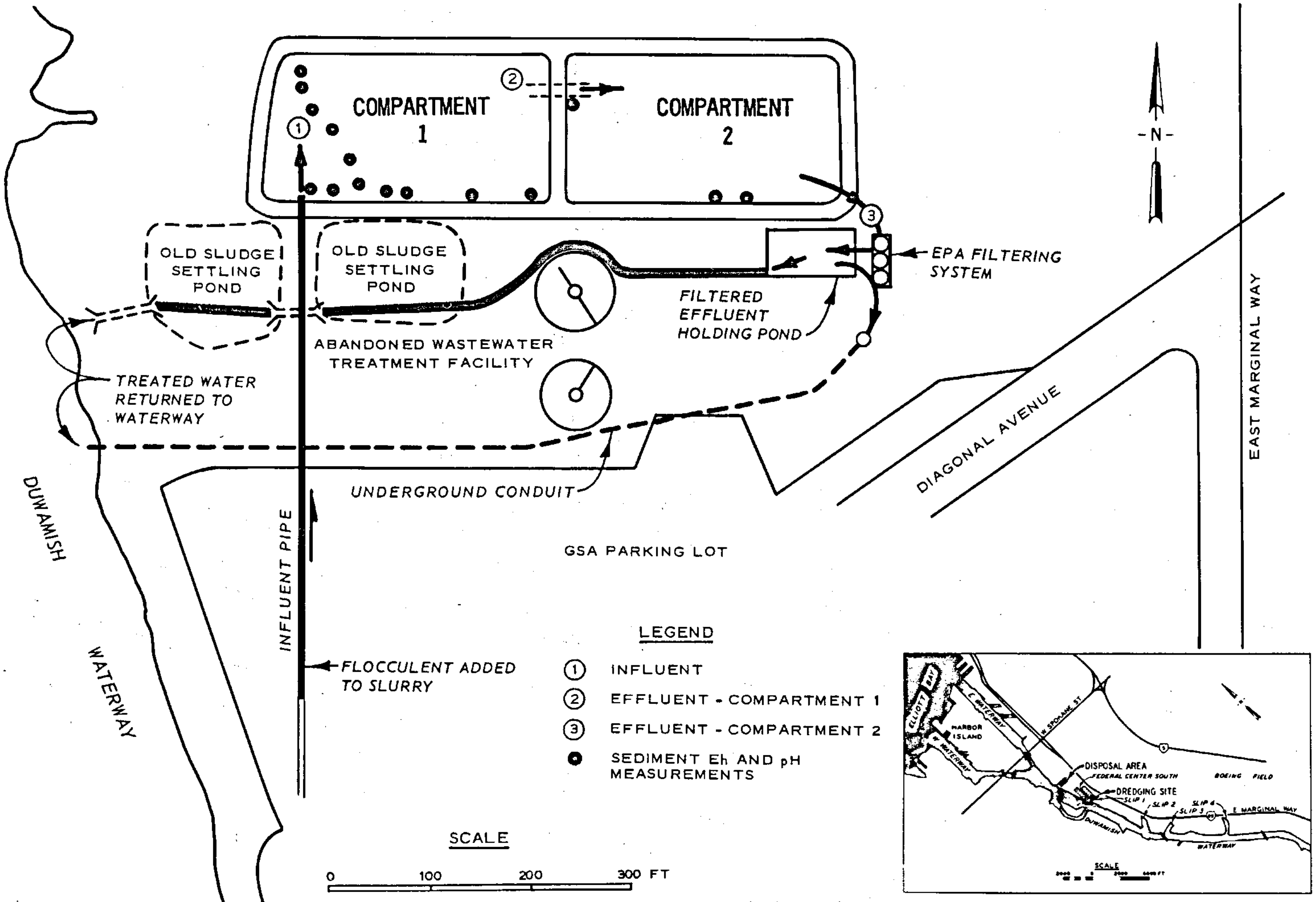
Figure 8. Lake Charles, La., disposal area

the majority of the dredged material slurry flowed around the edge of the ponded area for a distance of less than 500 yd, as the effluent was then being discharged through three 16-in.-ID pipes buried in the dike about 300 yd from the influent pipe. The effluent showed high turbidity as a result of probably a very short residence time. The final two daily effluent composites were collected at a 16-ft-wide rectangular weir located in the far southwestern corner of the site (see Figure 8) and Photo 12). A very long residence time of probably several days was suggested for the second effluent composite (5 February) because it showed low turbidity, and most of the turbidity usually appeared to be due to abundant planktonic algae. The third daily effluent sample appeared to be mixed, with half of the water flowing over the weir showing turbidity, and indicating a directional flow of the slurry water, and simultaneously, the other half of the flow being low in turbidity.

108. Based on dredging log calculations (Appendix A, Table A5), average influent flow rate during the three sampling days was about 28,400 gpm. A very rough calculation for effluent, considering an observed head of 8 in. over the weir on one occasion, would be a discharge of about 13,000 gpm[138] back into the Calcasieu River. However, rapid fluctuations in effluent volume can occur with only a small change in the water level over the weir, and it is doubtful that there would be a 50 percent loss in water volume during residence in the disposal area as temperate, overcast conditions prevailed.

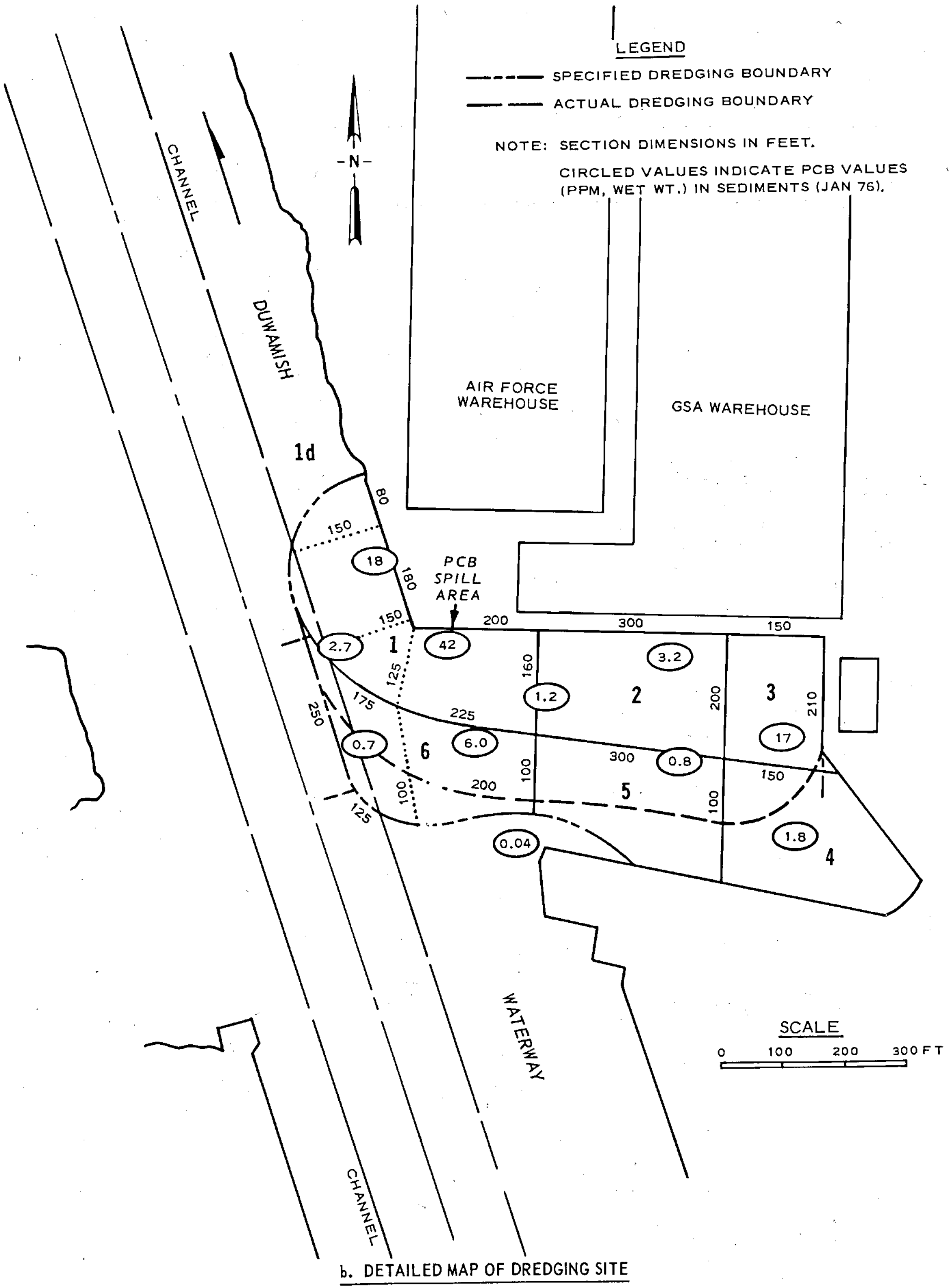
Seattle, Wash.

109. The dredging site was in slip 1 on the east side of the Duwamish Waterway, 2.2 miles above its confluence with Elliot Bay (Figure 9a). The disposal area was constructed on land about 670 yd north of the slip. It was necessary to collect surface and bottom background water samples in the Duwamish Waterway, 0.75 miles up the river from the dredging site, because of significant vertical salinity stratification in the brackish water regime.[139] Sediments were also collected from within the slip before dredging, compositing multiple samples from six defined sections (Figure 9b).



a. VICINITY MAP AND DETAILED MAP OF DISPOSAL AREA

Figure 9. Seattle, Wash., disposal area (sheet 1 of 2)



b. DETAILED MAP OF DREDGING SITE

Figure 9 (sheet 2 of 2)

110. The dredging operation was instigated as part of a joint EPA-Corps monitoring program of an accidental spill of 255 gal of PCB Aroclor 1242 in slip 1 during September 1974.[26,140] Because of the high affinity of PCB's with sediments, a Pneuma Model 600 pneumatic pipeline dredge was used since its functional design retards turbidity at dredging sites. Approximately 7000 yd^3 of in situ sediment was dredged from the slip, originating mainly from the upper foot of dredged area sediments except for an 8-ft-deep hole at the PCB spill site.[26]

111. Sediments were mixed with bottom water in a ratio of about 1 to 6 by volume to deliver a total influent slurry volume of greater than 10 million gal through a 10-in.-ID discharge pipe.[140] The dredging operation took place from 4 to 30 March 1976. The influent flow rate averaged about 2,100 gpm during 81.2 hours of active pumping but only about 750 gpm during the total 223.2 hours of the dredging operation. To improve settling in the disposal area, about 500 gal of a flocculent (Nalco No. 7134) was added directly to the influent discharge pipe to maintain a concentration of about 20 mg/ℓ.[26,140]

112. The dredged material slurry passed through two nonvegetated settling ponds (compartments), each measuring approximately 150 by 280 by 15.5 ft and each separated by a cross-dike containing a 20-in.-diam pipe which was used as a weir structure. Dredging site slurry water did not pass through this weir and into compartment 2 until 12 March. The two compartments contained a total ponded area of about 1.9 acres, with a combined capacity of about 48,000 yd^3. Almost all of the dredged sediment remained in compartment 1 (Photo 13), although reddish-brown turbidity developed in compartment 2 as a result of iron precipitation in the more oxidized water as noted in Photo 14. During periods of quiescence, the bottom of compartment 1 could be seen due to rapid settling of flocculated fine-grained solids and absence of ferric iron precipitation (Photo 15). Approximately 9500 yd^3 of dredged sediment filled compartment 1, thus reducing its initial capacity of about 24,000 yd^3 to 14,500 yd^3 by the end of the dredging project.[26]

113. During the period from 13 March to 7 April, 9,834,000 gal

of effluent water was pumped from compartment 2 through a cartridge filter (containing Filterite No. 264MSO), sand, and activated carbon filters, respectively, and the final effluent was then discharged to the surface water of the Duwamish Waterway through an underground conduit.[26] However, all effluent samples for this study were obtained before passage through these filter systems (Figure 9). Fresh groundwater intrusion into compartment 2 appeared to initially affect the effluent composition, but this contribution became less important as the saline dredging site water flushed out the impoundment. Table 3 gives daily influent and effuent slurry volumes. Influent values were calculated from daily working hours, assuming a constant inflow rate, and effluent volumes are metered flow rates.

<u>Vicksburg, Miss.</u>

114. Brown Lake, a 23.5-acre freshwater impoundment at the U. S. Army Engineer Waterways Experiment Station (WES), was the site of the dredging operation. A small cutterhead dredge was used to remove mainly grey-colored lake bottom silt. This sediment was loess eroded from mixed agricultural and forested land, with some small local inputs of domestic wastewater and urban runoff. Influent was discharged through a 12-in.-ID pipe, with the flow being shunted into either a 5.0- or 9.4-acre confined disposal area (Figure 10). Approximately 226,600 yd^3 of in situ sediment was dredged from 6 March to 15 April 1976, at a rate of 308 yd^3/hr during active pumping.*

115. The 5-acre disposal area, located at the mouth of Durden Creek, was monitored for this study, and composite influent and effluent samples were obtained on 17 March and 12 and 15 April. There were no records as to when or for how long each disposal area had been used, however, pumping into the smaller area occurred about one third of the time, and this area had been used almost exclusively during the first 2 weeks of dredging. The entire diked confinement was ponded except for the constricted region shown in Figure 10. Effluent was discharged over two 6-ft-wide rectangular weirs on a sluice box (Photo 16), but

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* Personal communication, James Peterson, Engineering and Construction Services Division, WES, August 1977.

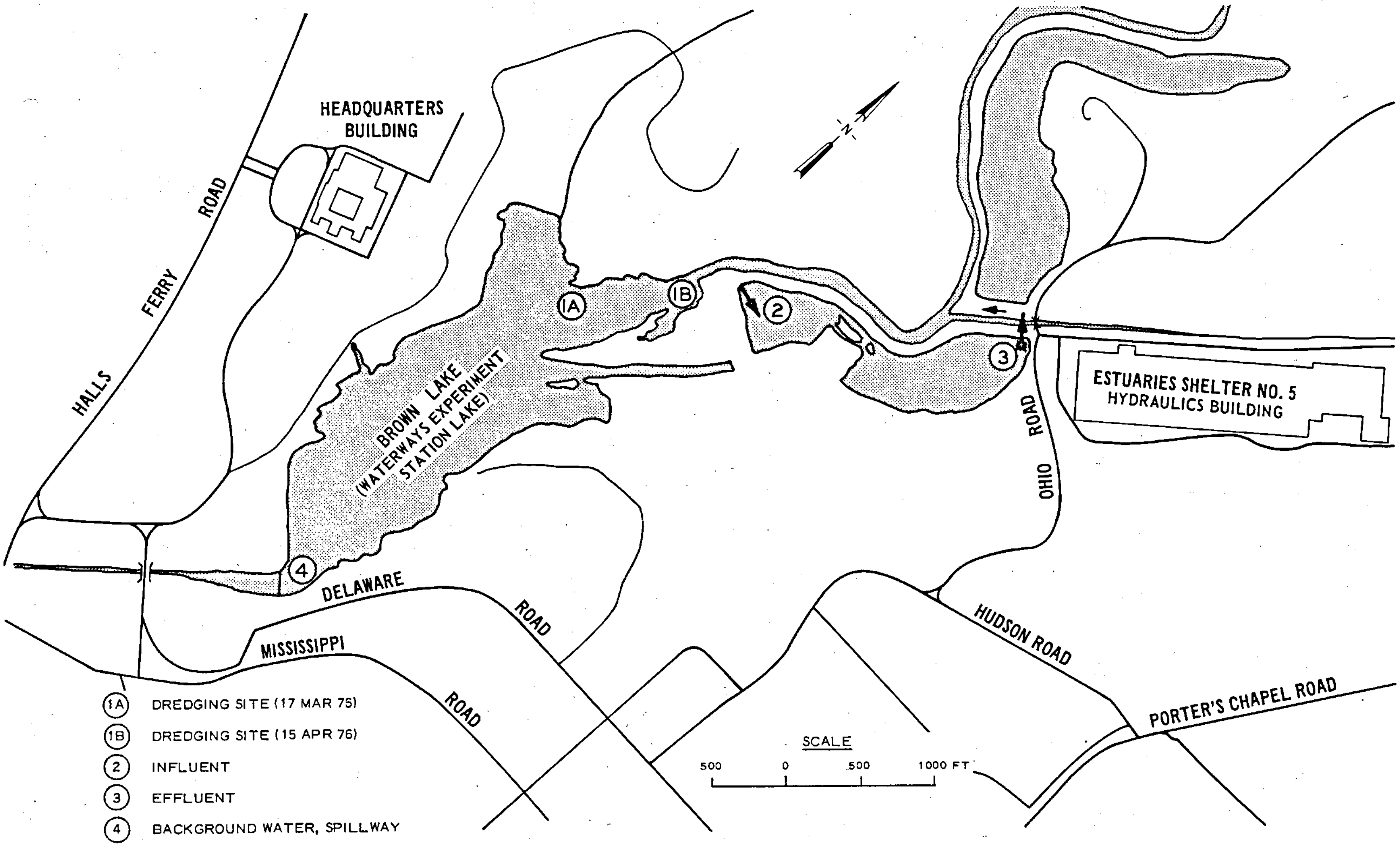


Figure 10. Vicksburg, Miss., disposal area

samples were collected from the end of a 20-in.-diam galvanized metal pipe which drained the sluice. The effluent discharge rate varied greatly during the monitoring program with average and maximum flows of about 500 and 2000 gpm, respectively. The site was drained for a few days beginning 18 March for a dike repair.

116. Effluent was monitored for about 1 week following completion of the dredging project to correlate residence time with nutrient release and other environmental parameters. During this period the effluent discharge rate decreased exponentially from about 300 to 5 gpm.

117. The disposal areas were barren of vegetation immediately preceding and during disposal operations. However, within only about 2 weeks, thick vegetation, composed mainly of tall grasses, cattails, and willows, had completely covered the sites.

Southport, N. C.

118. The monitoring program at this confined disposal area was designed to assess the interactions between vegetation and the dredged slurry in land containment areas. Vegetation could act to filter out suspended solids as well as affect soluble concentrations of different chemical species in effluent discharge water. The 48-acre disposal area is located on a small island opposite the Southport Marina on the Atlantic Intracoastal Waterway, which is at the easternmost end of Oak Island (see Figure 5). The site was ideal for the study as three distinct vegetation communities were located within the diked area. Figure 11 shows the locations of the dredging and disposal sites and the areal extent of the three main types of vegetation.

119. The western third of the containment area, nearest the effluent discharge weirs, was covered with dense woods. The dominant tree in this area was red cedar (Juniperus silicicola), although other common plants in the forested section included: wax myrtle (Myrica cerifera), hackberry (Celtis laevigata), marsh elder (Iva frutescens), marsh pine (Pinus serotina), yaupon (Ilex vomitoria), groundsel tree (Baccharis glomeruliflora), dwarf sumac (Rhus copallina), blackberry (Rubus sp.), pepper vine (Ampelopsis arborea), virginia creeper (Parthenocissus quinquefolia),

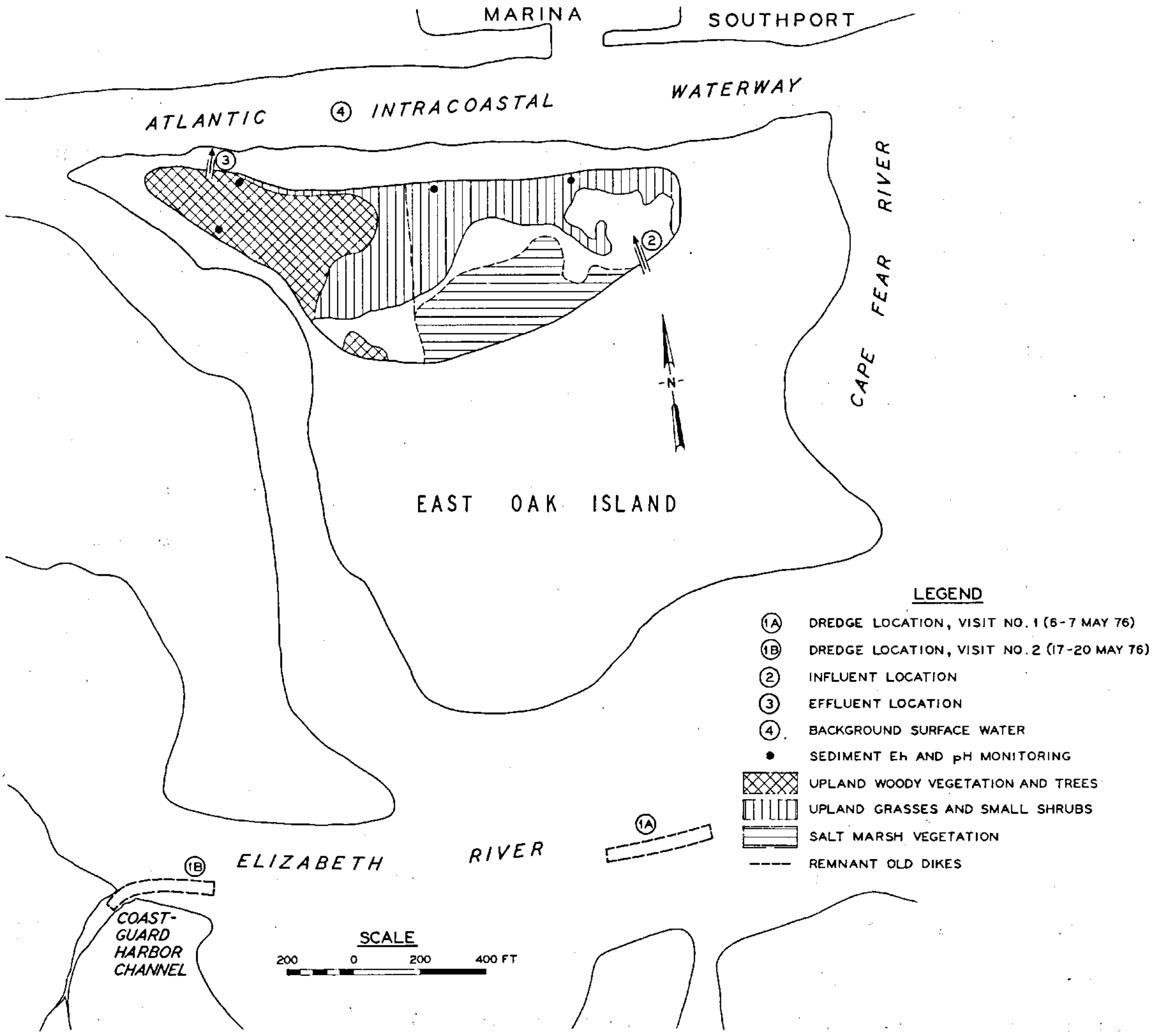


Figure 11. Southport, N. C., disposal area

summer grape (Vitus aestivalis), and greenbriar (Smilax smallii). Many of the trees in low-lying areas were dead, probably as a result of saltwater flooding during previous disposal operations. However, some of the plants (e.g., marsh elder and groundsel tree) are salt tolerant.

120. About 10 acres of the southern section of the disposal area contained natural salt-marsh vegetation as a result of the construction of a new section of dike. This area had never been used for dredged material disposal. The dominant and almost exclusive plant in this area was smooth cordgrass (Spartina alterniflora), which is also dominant in the salt-marsh areas immediately south and east of the diked containment on the island.

121. The remainder of the disposal site was about equally divided between barren areas and regions of sparse to dense stands of salt-tolerant but more drought-resistant vegetation. There existed a dense band (190 stems/m^2) of slatmeadow cordgrass (Spartina patens) along the eastern border of the wooded area and a band of mixed grasses (S. patens, Distichlis spicata, and S. alterniflora) along the southern border of the forest in old dredged material (Photo 17). There also existed scattered thick stands (250 stems/m^2) of common reed grass (Phragmites communis) and S. patens in parts of the eastern section near the influent discharge pipe. Other species scattered about sandy areas in this region included: seaside goldenrod (Solidago sempervirens), sand grass (Triplasis purpurea), evening primrose (Oenothera laciniata), and sweet clover (Melilotus albus).

122. The dredging operation took place between the main ship channel in the Cape Fear Estuary and the Coast Guard Boat Harbor up the Elizabeth River (Figure 11). A cutterhead dredge using an 18-in.-ID discharge pipe was used to excavate 160,000 yd^3 of in situ sediment between 5 and 21 May 1976. Influent and effluent samples were collected shortly after water began to flow over the D-shaped, 3.5-ft-wide weir on 6 and 7 May. In order to determine if vegetation interactions are important, the site was sampled again during the last few days of the disposal operation, on 17, 19, and 20 May. It was

assumed that burial of part of the vegetation by fresh dredged material would reduce the surface area of vegetation in contact with the slurry. However, one complication dealt with the degree of ponding during each trip to the site. Initially, only about 10 percent (5 acres) of the disposal area was ponded, but there was maximum contact between dense ground vegetation (e.g., grasses) and overland flow of the dredged slurry, as shown in Photo 18.

123. During the final collection, about half (20 acres) of the disposal area was ponded, but only the trees remained above the newly deposited disposal area sediment (Photo 19). Overland flow occupied most of the confinement area which was not ponded.

124. The effluent discharge, estimated at about 500 gpm, appeared to be comparable during the duration of both sampling trips to the site. Effluent water samples were obtained from the end of the discharge pipe draining the sluice (Photo 20). Background surface water samples were collected for analyses on each of the two trips. These were obtained from the middle of the Atlantic Intracoastal Waterway, about midway between the Southport Marina and the effluent mixing zone (Figure 11). Dredging log information for the sampling days is listed in Appendix A, Table A6.

Field Sampling Procedures

Sample collection

125. Field sampling methodology included the collection of at least three replicate influent and effluent dredged material slurry samples from each field site. A minimum of three subsamples were composited over a period of one to several hours in order to obtain a more representative aliquot. Where possible, replicate samples were obtained on separate days and trips to each site. This scheme allowed for inclusion of diurnal effects and daily dredging site sediment variability, while permitting adequate statistical evaluation. However, several changes in the influent or effluent collection sites also resulted which complicated sampling procedures but provided a better idea of actual problems in confined upland disposal operations. Surface

background water samples were collected from outside of the effluent discharge mixing zone, but sites were chosen which seemed to be representative of ambient water quality. Separate background water samples were collected on each trip to a given site. Emphasis given to extensive subsampling was in the order of influent, effluent, and background water, respectively.

126. Collapsible 1-gal polyethylene containers were employed for collection of samples used for heavy metal, nutrient, and oil and grease analyses. These containers were prewashed with 0.1 M hydrochloric acid and rinsed twice with deionized, distilled water. Usually, four of these containers were used; 50 ml of chloroform was added to one container as a preservative of the contained nutrients. Samples for chlorinated hydrocarbon analyses (pesticides, PCB's) were collected in ½-gal glass wide-mouthed jars. The containers were prewashed with hexane solvent, rinsed twice with deionized, distilled water, and combusted at 350°C for 30 minutes in a muffle furnace. The jar lids were lined with aluminum foil to further prevent analytical interferences. All containers were completely filled to exclude air, and the polyethylene containers were collapsed as aliquots were removed.

127. All of the influent samples were obtained at or directly beneath the end of the influent discharge pipes, except at site 3 where influent samples were obtained from the inlet pipe into the hopper of the hopper dredge. Influent slurry subsample collection and distribution to the multiple containers presented some problems in providing a representative and uniform sample. Due to the occasional inaccessibility and difficulty of obtaining samples directly from the discharge pipe, influents were obtained from within the mixing pool beneath the influent pipe. At the Seattle site, as a result of a low discharge rate and volume, samples were obtained directly from the end of the discharge pipe. Turbulence was noted to keep the majority of coarse solids in suspension within the mixing pools and a more uniform sample could be obtained because of recirculation of a larger slurry volume within the pool.

128. Two methods of influent sample collection were employed. For sites 2 and 8, a single polyethylene container was filled repeatedly, and, after stirring, a measured volume was added to each final container containing the composite sample. The containers were filled by alternating their order of filling after each completed filling cycle so as to equalize the larger solids content. Influent samples were composited at the other sites by filling several polyethylene containers of equal capacity simultaneously in adjacent areas of the mixing pool. Each collection vessel was then added separately to individual sample containers. This procedure was repeated until the sample containers were filled.

129. Collection of effluent posed less of a problem owing to its greater uniformity. Effluent samples were obtained either from the end of the effluent discharge pipe of a sluice box (sites 3-5, 8, and 9) or from flow over a weir structure (sites 1 and 6). At site 2, effluent was obtained by submerging a small capped container just beneath the surface oil film of the pond, adjacent to the sluice box. This was done to eliminate contamination from a kerosene base odor suppressing chemical which was being added inside the sluice box. Since the effluent was being pumped out of site 7, effluent samples were collected from an outlet valve, located before a series of in-line sand and activated charcoal filtering systems. Background surface water collection usually did not include subsampling. However, multiple analyses were frequently run on samples from different containers. For site 7, bottom background water samples were also collected to determine contaminant stratification in a salt wedge estuary.

130. Predredging bottom sediments were also obtained at the slip 1 dredging site in the Duwamish Waterway near Seattle, Wash. (Site 7). Samples were obtained from six sections in the slip that appeared to vary in chemical composition (Figure 9b), although sections 3 and 4 at the back of the slip were later combined because of noted physicochemical similarities in the sediments. Surface sediment samples (upper 2 in.) and 2-ft-long cores were obtained from each section with a Van Veen grab

sampler and a Phleger coring device, respectively. Six sediment samples from each section (eight samples for the larger combined area) were then composited, placed in air-tight containers, and stored at 4°C until analyzed. The sediments were used in conducting a standard elutriate test, and were prepared for interstitial water and bulk sediment (acid digest) analyses.

131. All samples were packed in ice immediately after collection and shipped by air freight to an analytical laboratory. Samples from sites 1-3 were sent to the Environmental Engineering Laboratory at the University of Southern California in Los Angeles, while the samples from sites 4-6, 8, and 9 were shipped to the Environmental Effects Laboratory at WES. Sample collection and preparation for site 7 were performed by the personnel of the Environmental Protection Agency (EPA) Region X Laboratory in Seattle, Wash. Samples were stored in environmental chambers maintained at 4°C. Samples were not frozen so as not to alter the structure of the solids.

Field measurements

132. Measurements of dissolved oxygen (DO), temperature, salinity, conductivity, and pH of influent, effluent, and background water samples were usually obtained concurrently with the subsample collections. Thus, from one to four measurements were usually made of each parameter at each collection site on a given day of sample compositing. Temperature and DO were measured with an air-calibrated YSI Model 57 DO meter; temperature, salinity and conductivity were measured with a YSI Model 33 salinity-conductivity-temperature meter; pH was measured with an Orion Model 407-A millivolt-meter equipped with a combination electrode, which was calibrated against standard (pH 4 and pH 7) buffers in the field. Salinity, conductivity, temperature, pH, and DO were monitored almost continuously in both ponds of site 7 with two Model 6 Hydrolab Surveyors equipped with continuous recorders.

133. All of the above parameters were measured in the same areas where the related subsamples were collected. Because burial of the sensing probes in sediment at the influent collection site was

occasionally a problem, checks were made by rapidly taking some measurements in a container of influent slurry for comparative purposes. Effluent measurements were made most often in nonturbulent water at the top of the weir or at a similar depth in the ponded water adjacent to a sluice box. Influent DO values were probably representative of maximum aeration of the influent slurry as measurements were made in the turbulent mixing pond. Occasionally, measurements were attempted directly at the end of the discharge pipe for comparison.

134. Eh and pH measurements were made in the fresh, recently dredged sediments within each confined disposal area. Probes 11 in. long with 0.2-in. lengths of exposed bright platinum wire at the probe tips were used to measure Eh in conjunction with a calomel reference electrode. The sediment pH was monitored with a combination electrode, washing and restandardizing the probe between measurements. The sites where Eh measurements were made (6 to 40 locations in each disposal area) were considered representative of the flooded regions in contact with the slurry. Although such measurements were usually limited to the periphery of the flooded areas, the Eh monitoring probes were inserted at least 3 in. into fluid sediments that were covered by water. This prevented rapid oxygen diffusion to the sediment surface. The Eh probes were allowed to equilibrate by inserting them at least 2 hours before measurement. The pH readings were obtained in sediments adjacent to the Eh probe locations.

Laboratory Procedures

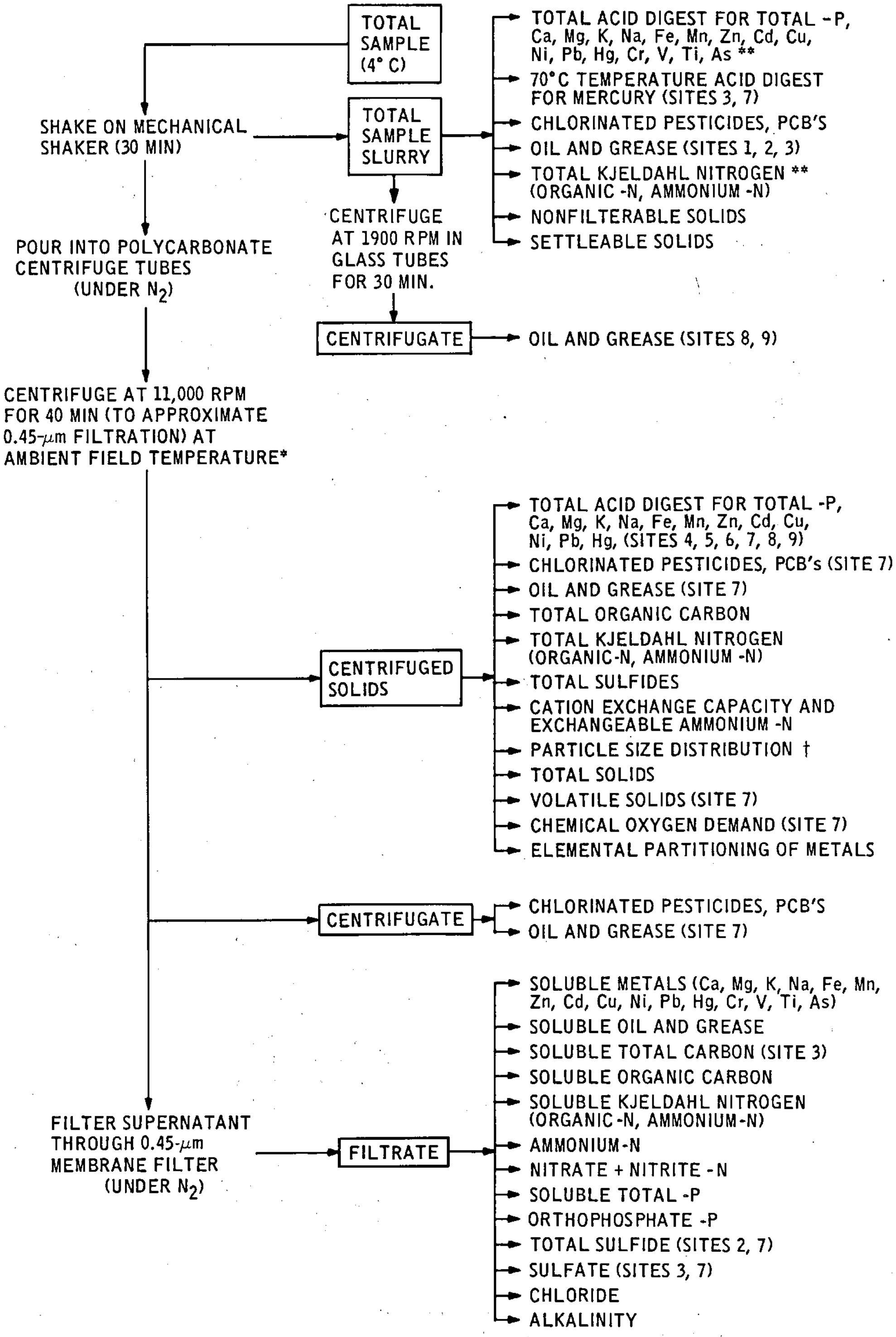
Sample preparation

135. The field-collected samples, upon arrival at the analytical laboratory, were processed as soon as possible to separate the solid from the liquid phase and to prepare the subsamples for different chemical analyses. The types of samples collected, collection dates, and preparation dates are given in Table 4. During the intervening time, all samples were stored at 4°C.

136. Sample phase separation (centrifugation and filtration), total and filterable sulfides, and geochemical phase partitioning extractions were performed in a glove bag purged continuously with

nitrogen gas to preserve the anaerobic integrity of the samples. The atmosphere was periodically checked with a DO meter to insure oxygen-free conditions. The sequential preparation scheme for influent samples is shown in Figure 12, while a slightly different scheme for effluent and background water samples is shown in Figure 13. The slightly modified preparatory procedures were necessary because of the often extreme variations in solids content between respective influent, effluent, and background water samples. Generally, if an influent sample was low in suspended solids, the total sample was subjected to total acid digestions for metals, total phosphorus, and total Kjeldahl nitrogen (TKN). Total samples from sites 1-3 were used for all analyses except total organic carbon (TOC) determinations of influents. For about 10 percent of the samples from sites 4-9, both the solid phase (centrifuged solids) and total sample were digested for the determination of some metal concentrations for comparative purposes. The solid phase of effluents having a high solids content was also used for some metal digestions. Cation exchange capacity and exchangeable ammonium were determined only in the solid phase samples. TOC was determined for influent solids with a high frequency induction furnace; filtrate and total effluent samples were analyzed with a TOC analyzer.

137. The parameters analyzed in the less than 0.45-μm (soluble phase) fractions of samples are listed in Figures 12 and 13. The soluble phase separations for chlorinated pesticide, PCB, and oil and grease determinations involved only high-speed centrifugation in stainless steel centrifuge tubes to approximate 0.45-μm filtration. From an analysis of the amount of residue collected on a 0.45-μm membrane filter following the centrifugation, it was ascertained that a velocity of 11,000 rpm, using a GSA rotor at room temperature for a period of 40 minutes, gave a close approximation for most samples. The remaining parameters, listed in Figures 12 and 13 as filtrates, were determined in liquid which was both centrifuged (in polycarbonate centrifuge tubes) and then filtered through 0.45-μm millipore membrane filters. The filters were previously washed twice with 1 M hydrochloric acid and deionized, distilled water to ensure the removal of acid-leachable

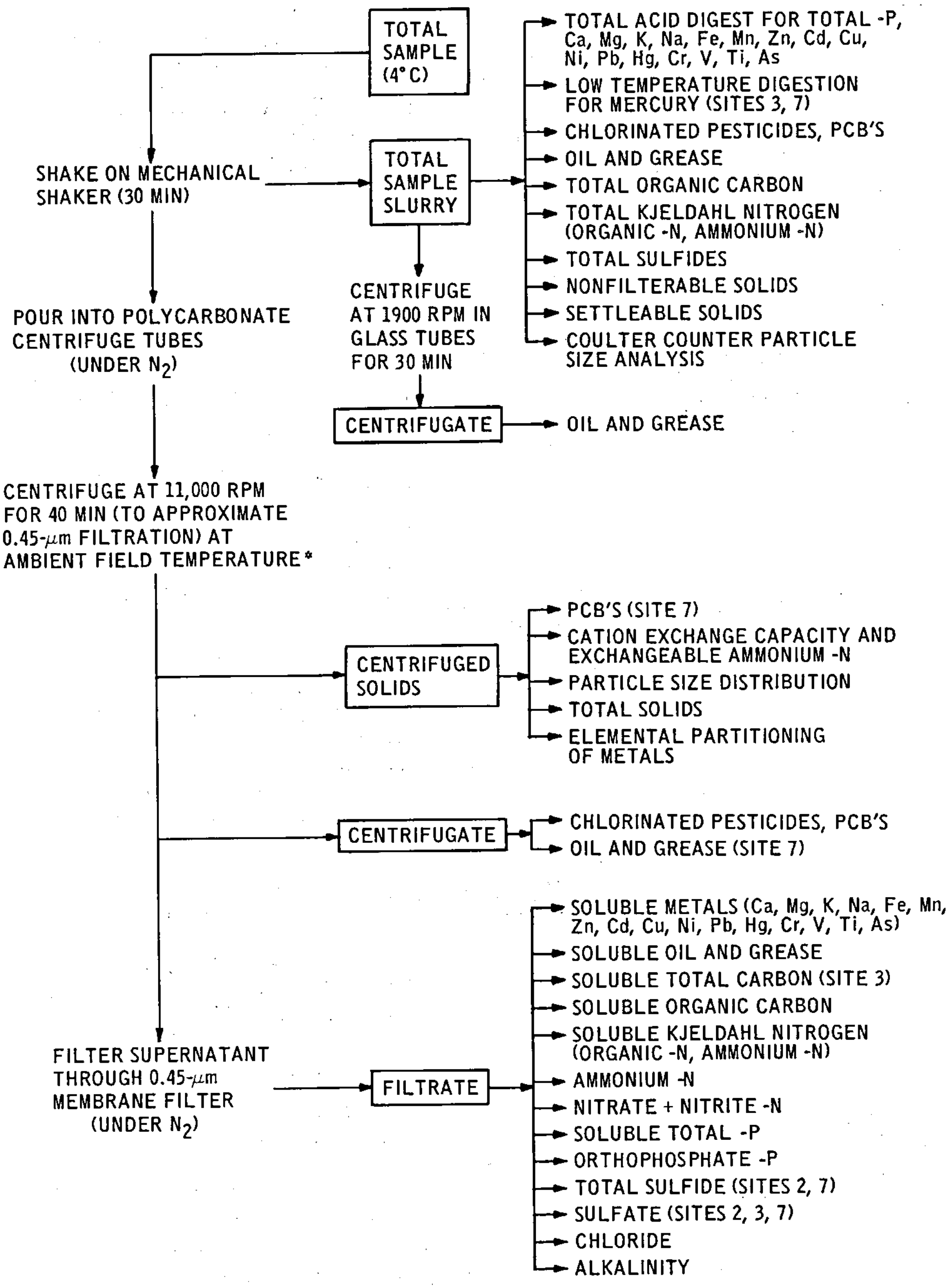


* SITE 7: 12,500 RPM FOR 20 MINUTES AT 4°C; ALL CENTRIFUGATIONS WERE MADE IN A SORVALL GSA ROTOR.

** TOTAL SAMPLE WAS DIGESTED ONLY WHEN THE SOLIDS WERE LOW.

† THE <2-μm (CLAY) FRACTION WAS FURTHER ANALYZED WITH A COULTER COUNTER.

Figure 12. Influent sample preparation



* SITE 7: 12,500 RPM FOR 20 MINUTES AT 4°C; ALL CENTRIFUGATIONS WERE MADE IN A SORVALL GSA ROTOR.

Figure 13. Effluent and background water sample preparation

chemical constituents present on or in the filters. Meticulous cleaning of all labware was routinely practiced, following precautions described elsewhere.[17,132] Centrifugation and filtration were generally performed at the field temperatures measured during sample collection in order to promote similar environmental conditions. All centrifugation and filtration steps were performed in a nitrogen gas atmosphere.

138. Following the separation of the solid and soluble phases, different aliquots were preserved for different parameters as follows:

<u>a</u>. <u>Metals.</u> Preserved with 5 ml of concentrated ultrapure (Ultrex) nitric acid per 500 ml of sample.

<u>b</u>. <u>Nutrients (TKN, NH_3-N, NO_2-N, NO_3-N, Total and orthophosphate).</u> Preserved with 40 mg of mercuric chloride per litre of sample.

<u>c</u>. <u>Total organic carbon (low solids samples).</u> Preserved with 5 drops of concentrated hydrochloric acid per 5 ml of sample.

<u>d</u>. <u>Total sulfide.</u> Preserved with 2 ml of 2N zinc acetate solution per litre of sample.

The solid phase material was placed in small plastic specimen cups under a nitrogen gas atmosphere and tightly sealed with plastic tape until digested and/or analyzed. Soluble phase samples were placed in thick-walled, tightly capped polyethylene bottles. All of the prepared samples were stored at 4°C until further processing and analysis.

Sample extraction, digestion, and analysis

139. Many measures were taken to prevent sample contamination during sample processing and analysis. As a general rule, subsamples to be used for metals analyses were processed and stored in acid-washed plastic or Teflon containers, while subsamples for chlorinated hydrocarbon analyses were processed and stored in hexane solvent-rinsed metal or glass containers to greatly reduce contaminant leaching and/or adsorption. Quality control within and between laboratories consisted of multiple digestions and analyses, standard addition, and inter-laboratory correlation of select samples. About half of the samples were subjected to these control measures.

140. The total and solid phase samples were digested in hot acid to determine total metals (Ca, Mg, K, Na, Fe, Mn, Zn, Pb, Cd, Cu, Ni, Cr, V, As, Ti) and total phosphorus. The procedures were as follows:

a. Sites 1-3. Digestion with 6 ml concentrated nitric acid, 4 ml concentrated hydrofluoric acid, and 3 ml concentrated perchloric acid per 1 g wet sediment (≈ 175°C), according to the method of Chen et al.[17]

b. Sites 4-6, 8, and 9. Digestion with 15 ml concentrated hydrofluoric acid, 10 ml concentrated nitric acid, followed by 8 ml fuming nitric acid per 1 g wet sediment (≈ 175°C), according to the method of Brannon et al.[132]

c. Site 7. Digestion with concentrated nitric acid (≈ 175°C) according to the EPA procedure (page 82).[142]

141. Mercury was digested separately as follows:

a. Site 1. Digestion of a 5-g wet weight sample with 20 ml concentrated nitric acid and 15 ml 2 percent potassium permanganate in a sealed flask (70°C for 12 hours), according to the method of Chen et al.[17]

b. Site 7. Digestion of sample with concentrated sulfuric acid and concentrated nitric acid (2:1 ratio), followed by 15 ml 5 percent potassium permanganate and 8 ml 5 percent potassium persulfate (95°C for 2 hours), according to the EPA procedure (page 124).[142]

142. All digests were performed in covered Teflon beakers, followed by filtration of diluted digests to remove any undigested material. The analytical techniques for each parameter are outlined in Table 5.

Geochemical phase partitioning analysis

143. The geochemical phase partitioning analyses (sites 4-7) were performed according to the methods outlined by Chen et al.[17] However, only the exchangeable, acetic acid extractable (carbonate), and easily reducible phases were performed for sites 4-6. The organic sulfide phase was included for site 7 samples using two different extraction methods for comparison. A brief outline of the sequential procedures followed is given below.

a. Exchangeable phase. Extraction of sediment (dry weight basis) with deaerated 1.0 M ammonium acetate at a 1:6 extractant to solids ratio under N_2 atmosphere.

b. Acetic acid extractable (carbonate) phase. Extraction of residue from above extraction with deaerated 1.0 M acetic acid 1:50 extractant to solids ratio (dry weight basis) under N_2 atmosphere.

c. Easily reducible phase. Extraction of previous residue with deaerated 0.1 M hydroxylamine hydrochloride in 0.01 M nitric acid at 1:50 extractant to solids ratio under N_2 atmosphere.

d. Organic + sulfide phase (site 7 only). Heating of each of two subsamples from the above extraction with 5 ml of 30 percent hydrogen peroxide and 3 ml of 0.02 M nitric acid (85°C for 5 hours), followed by treatment of each aliquot with:

    Twenty-five millilitres 1 M ammonia acetate in 1 M nitric acid

    Seven millilitres hydrogen peroxide in 0.01 M nitric acid

e. Remaining phases. Total digestion with hydrofluoric, nitric, and fuming nitric acid as previously described.

Wet sediments were used for all extractions and digestions, with dry weight values obtained from separate subsample aliquots. The exchangeable phase was initially corrected for soluble phase constituents according to the method of Brannon et al.[132] Each residue was washed between subsequent extractions with deaerated deionized, distilled water. Extractions were conducted by shaking the extractant with the solids for 30 minutes on a mechanical shaker, followed by centrifugation at 11,000 rpm for 40 minutes and 0.45-μm membrane filtration under nitrogen gas.

Particle size fractionation of effluent solids

144. Effluent solids from site 7 were collected in the field by processing about 500 litres of effluent water through a Sharples continuous high-speed centrifuge at a rate of about 10 to 15 ml/minute. This was necessary to trap enough solids for analysis from the low-solids-content water. The centrifugation speed seemed sufficient to trap particles smaller than 0.45 μm. However, the very fine particulate matter, especially some of the fine organic detritus, probably escaped in the centrifugate. The centrifugate turbidity did not exceed 4 JTU's.[26] The centrifuged solids were kept wet, at room temperature, until filtration and analysis by the Environmental Engineering Department at the

University of Southern California in Los Angeles.

145. The recovered effluent solids were then washed successively with distilled water through 8- and 0.45-μm millipore membrane filters, thus separating the solids into fractions (a) greater than 8 μm, (b) 0.45 to 8 μm, and (c) less than 0.45 μm in size. The solids in each fraction were then totally digested and analyzed. The digestion methods were the same as those previously listed for sites 1 to 3; the analysis methods are those listed for sites 1 to 3 in Table 5.

Standard elutriate test procedures

146. The standard elutriate test was performed by vigorously shaking (at about 100 excursions per minute on a mechanical shaker) 300 mℓ of a 1:4 sediment-water suspension at room temperature for 30 minutes, followed by settling of the slurry for 1 hour. Dredge site water was used for the dilution. After the 1.5-hour procedure, the elutriate water was filtered through a 0.45-μm membrane filter. Organic contaminants, including PCB's, were separated for analysis by high speed centrifugation at 10,000 times gravity. Chemical analyses were then performed according to the methods shown for site 7 in Table 5.

147. The Region X Laboratory of the Environmental Protection Agency performed the elutriate tests using composite sediments from five different areas in slip 1 on the Duwamish Waterway, Seattle, Wash. Sediment samples were removed to a depth of about 2 ft from slip 1. Figure 9b shows the six sections of slip 1 which were cored; six core samples from sections 1, 2, 5, and 6 were then composited, while eight core samples from sections 3 and 4 were combined. The standard elutriate tests were performed using these five composited sediment samples. Additional details of the procedures are given in a separate report.[26]

Methods of data evaluation

148. The great variability between sites, which was desirable in order to obtain a good cross section of land containment area problems, makes interpretations based solely on general trends difficult. Also, statistical data analyses can easily become misleading if not interpreted in conjunction with detailed field observations. Consequently, a discussion of the general mechanisms involved will be based on chemical and geochemical findings, but the conclusions will additionally be based on field observations and various statistical tests. The F-test will be used to show whether the variations between data sets are significantly different from variations within sets. The analysis of variance F comparisons will be made between influent, effluent, and background water data. If variations between two means within a data set appear to be significant, while having similar standard deviation values, a Student's t-test or LSD (least significant difference) will be used to determine significance for these comparisons. Significance will pertain to the 95 percent confidence level ($p \leq 0.05$) unless otherwise stated.

149. General data trends will be discussed initially, using the averages, ranges, and standard deviations computed from the data sets collected at each of the field sites, excluding Sayreville, N. J. The conclusions will include, where possible, observed interrelationships among several physical and chemical parameters. Results of the geochemical phase partitioning tests will be included in the interpretations. These data, although not totally specific, indicate which solid geochemical phases or compounds predominate in the particulate fractions of influents and effluents. General shifts in the geochemical phases during dredged material retention in confinements also allow for a detailed qualitative estimation of the factors controlling the mobilities of various nutrient and trace metals. Additionally, various trends in contaminant mobility or fixation will be discussed in connection with specific environmental features, such as dredging site salinity, sediment texture, organic content, disposal area retention time, and effect of vegetation in a containment area.

PART IV: RESULTS AND DISCUSSION

General Characteristics of Disposal Areas

150. The nine different confined land disposal areas monitored in this study included freshwater riverine, freshwater lacustrine, brackish water riverine (salt wedge estuaries), and estuarine ship channel sites. The diked areas ranged in size from less than 2 to 525 acres. Some sites were entirely ponded while others consisted mainly of thickly vegetated overland flow systems. Dredged material textures ranged from predominantly coarse river sand and gravel to black clayey estuarine muds. The contrasting factors at each sampled disposal area are listed in Table 1.

Physical and Chemical Characteristics of Influents, Effluents, and Background Water

General trends

151. The complete data sets for the nine field sites, including sample data from two distinct collection trips to a vegetated site, are presented in Appendix B, Tables B1-B19. Average values for field and analytical data from each site are given in Tables 6-8. Statistical data on each parameter, including the total number of samples, ranges, means, and standard deviations for surface background water, influent, and effluent, are presented in Table 9. The most useful comparisons would be between influents and effluents, to determine the fate of the monitored contaminants in each disposal area, and between effluents and surface background water, to roughly assess the potential impact of the effluents on water quality near the discharge site. Also, for most disposal areas there is probably a close similarity between the surface background water and bottom water composition at the dredging site; this is especially true where the dredging and disposal area effluent discharge take place in the same body of water. Generally, 77 to 98 percent by weight of disposal area influents consisted of sediment pore

and bottom water (Table 7), and at least 80 percent of this total water should be bottom water.

152. Figure 14 depicts the percent increase or decrease of a given physical parameter, chemical element, or compound during confined land disposal. The percentages were calculated by assuming the influent concentration to be 100 percent. The bar graph shows that most of the nutrients, trace metals, oil and grease, and chlorinated hydrocarbons decreased during retention in land containment areas.

153. Influents versus effluents

a. Total constituents. Comparisons of total influent and effluent digests showed prominent net decreases for all nutrients, oil and grease, PCB's, DDT, DDD, and most major elements and trace metals during land containment (Figure 14 and Tables 8 and 9). Despite the often very great variance in the solids content of the influent samples, most total contaminants showed highly significant differences between influent, effluent, and background water concentrations, using an analysis of variance F Test (Table 9). The statistical nonsignificance shown for solid phase sulfide, DDT, DDD, and possibly total mercury variability is due mainly to influent solids variability, which in reality has a minimal impact on the effluents because of attenuation of the rapidly fluctuating influents. Sodium, mercury, and DDE were the only parameters which did not seem to be greatly reduced in the total effluents. For most elements, the percent decrease in total concentration was very close to the respective decrease in total or nonfilterable solids during containment (Figure 14). However, some variances in elemental versus solids removal efficiency seemed to be greater than analytical error. Additionally, different elements showed variance in their influent-effluent solids ratios, with nitrogen, phosphorus, potassium, sodium, manganese, zinc, and mercury showing major increases in the effluent solid phase (Tables 8 and 10). It thus appears that different compounds or elements may have affinities for particles of different sizes and specific gravities. Other studies have also shown this relationship in marine sediments[74] and in ocean discharges of wastewater effluents.[133] This topic will be discussed further in later sections.

b. Soluble phase constituents. Nitrate-nitrite nitrogen was the only chemical parameter which showed a signifi-

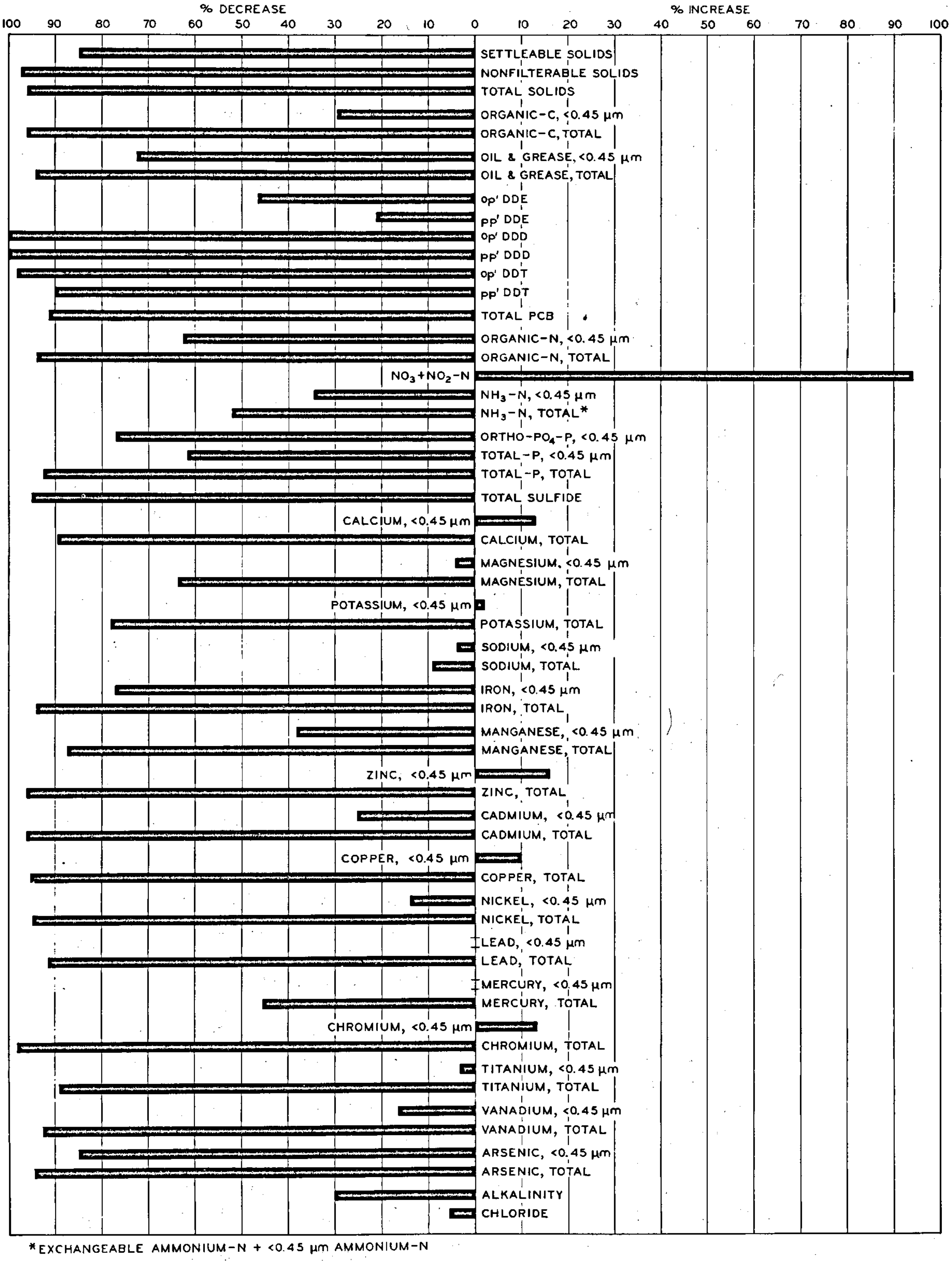


Figure 14. Percent increase or decrease of physical and chemical parameters in total and soluble phase effluent dredged material based on influent-effluent samples from eight land containment areas

cant increase during land containment of dredged material (Figure 14). The concentration increased almost twofold, from 0.18 to 0.35 mg/ℓ (the nitrite contribution being unimportant in all instances). However, this increase should not create serious impact since the discharge concentration was generally below the surface background water average of 0.46 mg/ℓ (Table 9) and well below accepted water quality standards. The greater than threefold increase in soluble chromium indicated by Table 9 is the result of high levels of chromium in effluents from the Seattle, Wash., disposal area; chromium was not analyzed in the influent samples from Seattle (Appendix B, Table B16). Excluding the Seattle data, chromium showed only a very small increase (see Figure 14). High concentrations of ammonium nitrogen and soluble manganese occurred in some influent and effluent filtrates. Soluble ammonium nitrogen in the influents averaged 20.8 mg/ℓ, with an upper range of over 70 mg/ℓ; effluent samples averaged 13.6 mg/ℓ, with maximum concentrations also around 70 mg/ℓ. Soluble manganese appeared to show a comparable decrease during residence in the land containment areas. The influent and effluent means were 2.35 and 1.45 mg/ℓ, while maximum values were 14.4 and 8.0 mg/ℓ, respectively. Some total alkalinity values (as calcium carbonate) were also very high, giving average influent and effluent values of 412 and 287 mg/ℓ in comparison to a background water average concentration of 88 mg/ℓ (Figure 14 and Tables 7 and 9). The above parameters will be discussed in more detail in their specific sections. Dissolved oxygen in effluents, based on multiple surface water measurements made inside of the disposal areas adjacent to the discharge weirs, fluctuated greatly. Ranges were from 0.6 to 12.5 mg/ℓ, with an average of 5.3 mg/ℓ. Thus, low effluent dissolved oxygen can occasionally be a problem. The few measurements which were obtained immediately at the end of the influent discharge pipe showed negligible dissolved oxygen, whereas dissolved oxygen averaged 3.8 mg/ℓ in the mixing pond immediately below the discharge pipe (Table 9).

c. Geochemical partitioning of solids. The potential mobility or toxicity of an element is highly dependent on the geochemical phases with which it is associated in the solids fraction. Table 10 and Figure 19 give the amounts and percentages of each of 14 nutrient and trace metals that were solubilized during four sequential chemical extractions of five influent and effluent solids; these were collected from four confined disposal areas (Wilmington, N. C.; Richmond, Va.; Lake Charles, La.; Seattle, Wash.). The site-specific data are given in Appendix B, Table 19. The treatments are considered to be highly specific for certain geochemical phases; namely, ammonium acetate extraction of the "exchangeable" ions, 1 M acetic acid extraction of the "carbonate phase," and 0.1 M hydroxylamine hydrochloride extraction of the "easily reducible phase." The exchangeable ions are considered to be readily available to aquatic organisms as they are mainly weakly adsorbed on the outer surfaces of fine solids. The carbonate phase may also be readily affected by changes in the environment, especially as a result of interaction with or uptake by the biota; also the carbonates of a given metal are generally more soluble than many other solid phase chemical precipitates, and thus a major shift of a metal to a carbonate complex may result in a similar increase in its soluble phase concentration. The easily reducible phase variations reflect the oxidized or reduced status of a containment area during disposal operations. An increase in this phase would suggest an overall increase in the oxidation of the dredged material slurry, and vice versa. Influent levels would give an idea of the Eh status and immediate oxygen demand of the bottom sediments being dredged. Also, a close relationship has been observed between the easily reducible phase and the release of trace metals in the standard elutriate test.[132]

d. Site-specific studies. The results of the chemical analysis of three different size fractions of effluent suspended solids from the Seattle confined disposal area will be discussed in a later section. Also, standard elutriate test data, using sediments from the Seattle dredging site, will be compared with sediment pore water and effluent concentrations from the same containment area. Thus, comparisons could be made between the geochemical phase partitioning, particle size fractionation, and standard elutriate test data from the Seattle site. Additional sections will discuss the relationship between contaminant mobility and the solids content and texture of dredged material, as well as a study of the interaction of vegetation with dredged material in land containment areas.

154. Effluents versus surface background water

a. Total constituents. Since effluent samples contained nearly 50 times the suspended solids levels of the respective surface background water samples, only the major ions (Ca, Mg, Na, K), nickel, vanadium, and some chlorinated hydrocarbons (DDD, DDE) were at comparable levels. Dissolved oxygen in the effluents averaged half the levels in surface background water, based on minimal data for the latter (Tables 8 and 9).

b. Soluble phase constituents. Figure 15 depicts overall differences in the concentrations of soluble phase and total nutrients and soluble phase heavy metals between effluents and surface background water. These data indicate that the soluble phase of confined land disposal area effluents are usually of a lower quality than the water into which they are discharged. The soluble chemical elements or compounds which showed the highest effluent to background water ratios were ammonium nitrogen, manganese, zinc, and titanium, with ratios of 50, 23, 11, and 280, respectively. However, at many sites the soluble phase concentrations of trace metals, major ions, phosphorus, and organic carbon were similar in both effluent and background water samples, while the net nitrate nitrogen levels were 49 percent higher in the background water. One interesting observation is that the very high soluble phase zinc, cadmium, copper, and nickel noted in the Southport dredged material seemed to have originated from the background water at the dredge site (Table 8).

155. Residence time and salinity effects. The bar graph in Figure 16 gives the average soluble phase concentrations of nutrients and heavy metals in effluents from each of the field sites. Figure 17 shows the average concentration changes which occurred in the soluble phase of effluents, compared to influent levels, from the nine monitored sites. The site locations in Figure 16 and 17 are arranged in the order of their observed or inferred decreasing slurry retention times, based on their effective size, influent-effluent flow rates, degree of ponding, and tortuosity of the flow path. For example, the slurry appeared to remain longest in the large Wilmington overland flow system and shortest in the small, undivided disposal area at Grand Haven. Also, the salinity regime at the dredging site is given for comparative purposes.

156. The results indicate that residence time and salinity may not be major controlling factors for nutrient and metal mobility from dredged material in confined land disposal areas. Many interacting mechanisms seem to be involved. For example, a similar slurry retention time in two different sites may result in considerably different environmental conditions. The Eh and pH changes may be most important,

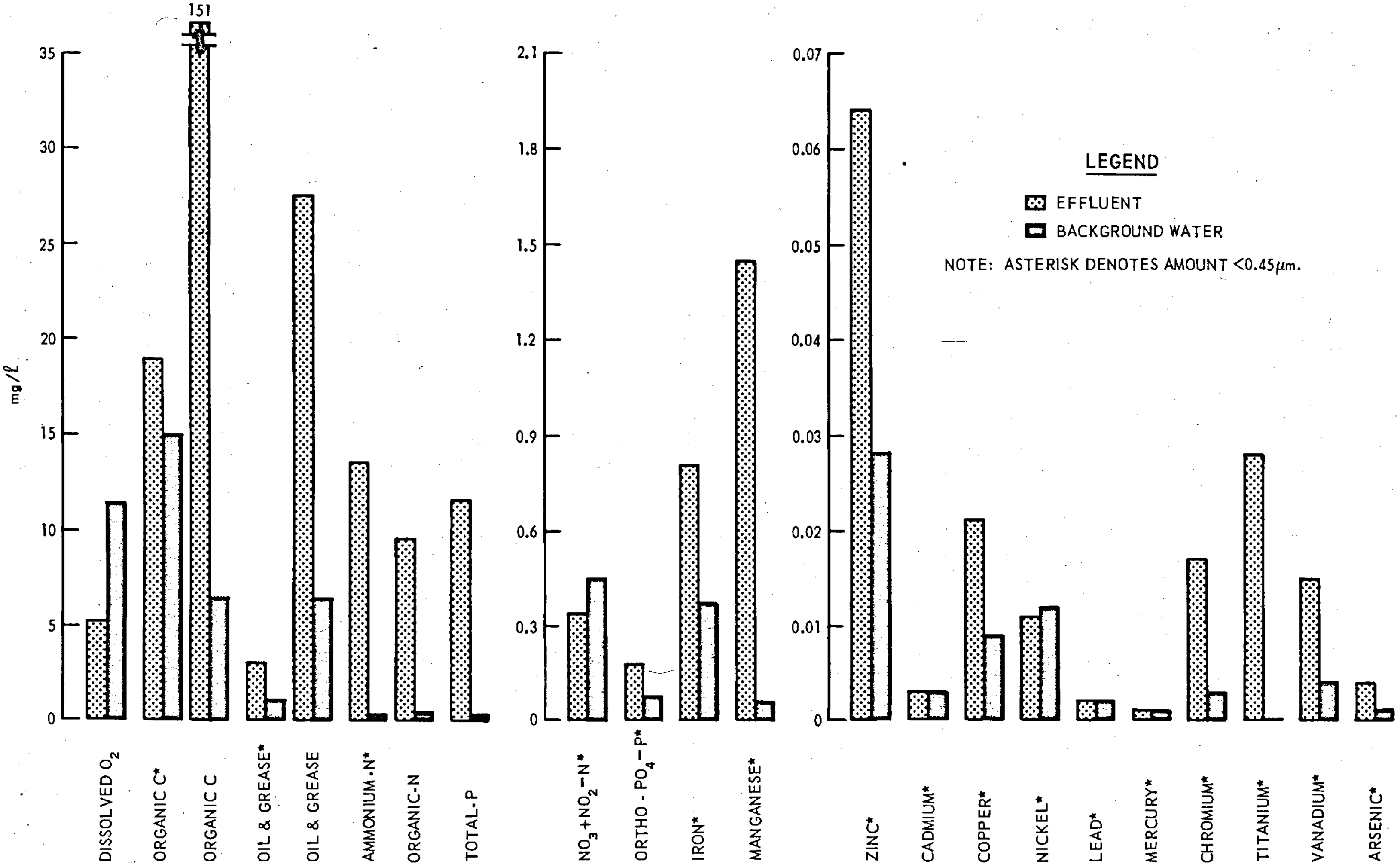
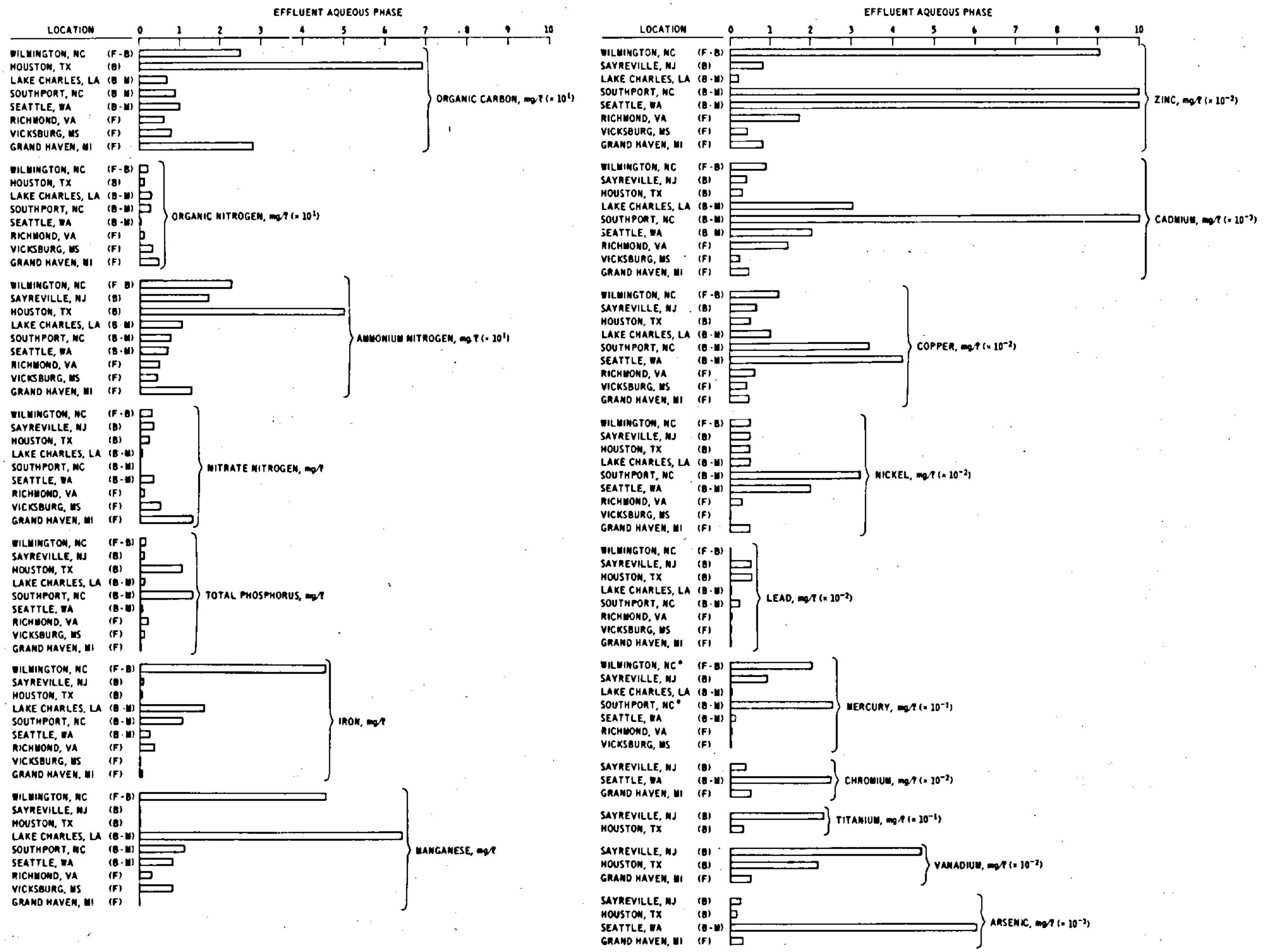


Figure 15. Average concentrations of nutrients, organic contaminants, and soluble phase heavy metals in effluents and surface background water samples representing the nine confined disposal areas

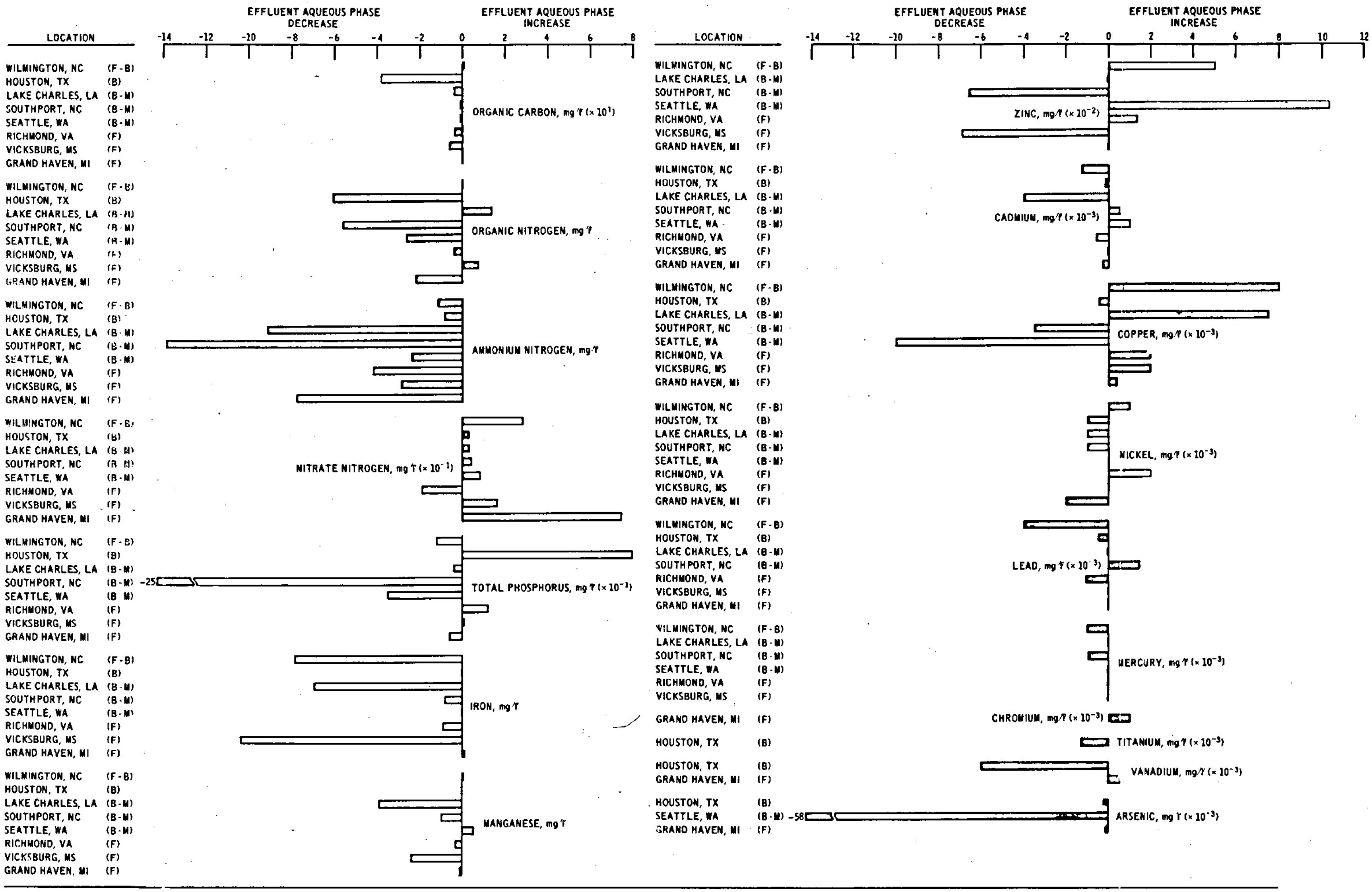


(F) = FRESH WATER: INFLUENTS WITH < 0.1% SALINITY
(F-B) = FRESH-BRACKISH: INFLUENTS WITH 0.1 TO 0.5% SALINITY
(B) = BRACKISH WATER: INFLUENTS WITH 0.5 TO 1.5% SALINITY
(B-M) = BRACKISH-MARINE: INFLUENTS WITH > 1.5% SALINITY

* ANALYSES WERE MADE WITH A ZEEMAN SPECTROPHOTOMETER, WHICH GAVE HIGHER VALUES THAN THE COLD VAPOR METHOD.

NOTE: DISPOSAL AREA LOCATIONS ARE LISTED IN THE ORDER OF THEIR DECREASING DETENTION TIMES, BASED ON SIZE, INFLUENT-EFFLUENT FLOW RATES, AND TORTUOSITY.

Figure 16. Concentrations of nutrients and heavy metals in the aqueous phase of effluents from eight confined disposal areas



(F) = FRESH WATER: INFLUENTS WITH < 0.1% SALINITY
(F-B) = FRESH-BRACKISH: INFLUENTS WITH 0.1 TO 0.5% SALINITY
(B) = BRACKISH WATER: INFLUENTS WITH 0.5 TO 1.5% SALINITY
(B-M) = BRACKISH-MARINE: INFLUENTS WITH > 1.5% SALINITY

NOTE: DISPOSAL AREA LOCATIONS ARE LISTED IN THE ORDER OF THEIR DECREASING DETENTION TIMES, BASED ON SIZE, INFLUENT-EFFLUENT FLOW RATES, AND TORTUOSITY.

Figure 17. Changes in the nutrient and heavy metal concentrations in the aqueous phase of effluents from eight confined disposal areas

and there appeared to be a better correlation between low effluent dissolved oxygen measurements and contaminant mobility. The nutrient release patterns can strongly affect the dissolved oxygen status. Generally a long residence time results in greater aeration and oxidation of the surface waters, but the proliferation of microorganisms in nutrient-rich water may promote anoxic conditions with time. Although this study fails to show conclusively any relationship between residence time, salinity, or oxidation status of dredged material and the mobility of contaminants, site specific inferences concerning the above were evident and will be discussed in later sections.

157. Generally, a direct relationship existed between increased residence time and solids removal, which would similarly be related to the removal of solid phase contaminants. However, turbulent flow conditions which occasionally existed near the effluent weir would cause the resuspension of settleable solids.

Salinity, conductivity, temperature, dissolved oxygen and pH

158. Salinity and conductivity. Marine salinity and conductivity readings of influents, effluents, and surface background water indicated that a net evaporative loss of about 15 percent occurred in the confined disposal areas (Table 6). This finding is contradicted by the chloride and soluble sodium values, which show about a 5 percent overall dilution by rain or groundwater (Table 9 and Figure 14). However, the above comparisons are not significant, using the Student's t test.

159. Temperature. Temperature increased overall by less than 2°C during the containment area monitoring (Table 9), which should not create a problem. However, temperatures were higher in late spring (Table 6). Temperature and salinity increases could become much more noticeable during summer disposal operations, although surface water near the discharge area should also increase in temperature and possibly salinity at this time.

160. Oxygen status. Dissolved oxygen levels in effluents averaged 5.3 mg/ℓ for all sites. However, site-specific fluctuations ranged as low as 0.6 mg/ℓ (Table 9). Influent dissolved oxygen measurements were

made in the turbulent mixing pool beneath the discharge pipe, which allowed for some aeration of the influent sediment slurry. Thus, there was only an apparent net increase of 1.5 mg/ℓ of dissolved oxygen during retention in the disposal areas. Most slurries sampled directly at the end of the influent discharge pipe showed no measurable dissolved oxygen. Often drastic fluctuations in dissolved oxygen were experienced with both depth and time in water adjacent to the effluent weir. Thus, circulation patterns in the disposal area and turbulent mixing at the weir are important factors in governing dissolved oxygen values in effluents. The impact of low dissolved oxygen in effluents is partly dependent on the size, rate of mixing, temperature, and stratification of the body of water receiving the effluent.

161. The data in Table 11 show an inverse relationship between total (or nonfilterable) solids and dissolved oxygen in most of the confined disposal area effluents. However, dissolved oxygen concentrations appear to be more closely related to the oxygen demand and diffusion rate in the system. High soluble nutrient concentrations can remove dissolved oxygen from water low in suspended sediments by inducing microbial proliferation, which puts a demand on the available oxygen. Several factors other than solids levels created low effluent dissolved oxygen values. The first effluent sample from Wilmington, which shows a poor relationship with the respective effluent solids content, was collected and measured for dissolved oxygen from subsurface leakage through the weir boards as the slurry was then being totally contained. During dissolved oxygen monitoring, subsurface waters in ponded areas adjacent to discharge weirs were invariably found to be lower in dissolved oxygen than were surface waters, despite observed uniformity in solids content. This is promoted by the very slow diffusion rate of oxygen through nonturbulent water. The low dissolved oxygen values for effluents from Southport, regardless of very low solids concentrations, probably resulted from accentuated biological activity, induced by the observed high soluble nitrogen and phosphorus concentrations in the effluents. The presence of reduced soluble chemical constituents (e.g., free sulfides) in the site water could

also maintain low levels of dissolved oxygen.

162. Although the two most heavily vegetated sites (Wilmington and Southport, N. C.) showed very low effluent dissolved oxygen values, the impact of the vegetation may actually be minimal. Both sites had very high solid and soluble phase nutrient concentrations in the influents (Table 8). As do most heavily vegetated sites, these confinements had extensive overland flow as a treatment mode, which results in extensive contact and mixing of the dredged slurry with anoxic sediments in the disposal areas. Therefore, extensive overland flow through unconsolidated, anoxic sediments may result in depressed effluent dissolved oxygen values. Perhaps multiple-compartment overland flow systems would prevent low dissolved oxygen values by separating solids treatment from soluble nutrient treatment. In retrospect, overland flow of low-solids dredged material over consolidated sediments should promote dissolved oxygen increases by accentuating the rate of oxygen diffusion into the slurry.

163. The above discussion emphasizes that many other factors besides high effluent suspended solids can result in low effluent dissolved oxygen. However, the presence of high effluent solids does indicate poor management of the disposal area. Also, the discharge of anoxic sediments may more seriously impact the biota, especially benthic organisms, since these solids will continue to exert an oxygen demand on the water near the discharge zone. Soluble nutrients would normally be rapidly diluted to background levels.

164. pH fluctuations. There was a significant increase in pH ($p \leq 0.01$, using the F-test) during the dredging and disposal cycle, with average surface background water, influent, and effluent values of 6.6, 7.15, and 7.5, respectively (Table 9). These fluctuations were probably controlled by the buffering of the often slightly acidic dredging site water by the near neutral influent solids, and increased effluent pH may be promoted by photosynthesis by algae in disposal area surface waters (Tables 6 and 12). Many other complex chemical reactions could have also facilitated this pH increase, though.

165. The overall increase in disposal area effluent pH should not impact water near the discharge zone. However, site specific increases in pH were above 9 in several instances. The high values were observed in effluents monitored for 1 week after influent disposal had ceased at the small barren freshwater site near Vicksburg, Miss. (Table 13). The major impact of pH values above 8 results from the conversion of ammonium to ammonia, which is very toxic to many forms of aquatic life. As shown in Table 13, the effluent discharge rate was very low at the time. High pH values in disposal area discharge water are most often the result of extensive photosynthesis by planktonic or crust-forming algae. Alkaline pH can be accentuated by high nutrient concentrations, low solids content, and long residence time of the ponded water. Vegetated areas would tend to be less impacted by these high rates of algal photosynthesis, as the plants reduce the amount of light reaching the sediment substrate or water surface.[44] However, moderately high pH values and extensive algal growth would tend to favor removal of high nitrogen levels in the site waters, through biological uptake and ammonia volatilization, especially during warm weather. Also, the increased ammonia levels could help eliminate microbial pathogens originating from the dredged material.[146]

Solids, particle size distribution, and cation exchange capacity

166. Particle size analyses. The particle size of influents ranged from 5 to 65 percent sand (mean of 20 percent), 24 to 73 percent silt (mean of 40 percent), and 11 to 65 percent clay (mean of 40 percent). Effluents contained about 60 percent clay-sized particles, but there was also a net retention of 8 percent sand in the measured samples, which was probably stirred up from the bottom of the disposal area by turbulent mixing near the weir. Sand was observed only in effluent samples from Wilmington and Lake Charles. Only samples containing high levels of suspended solids could be measured for particle size. Thus, for all of the sites there was probably a much greater proportion of clay-sized particles.

167. Coulter Counter analysis of influent clay fractions, from the mechanical particle size determinations, and total effluents were compared. The results shown in Tables 7 and 9 suggest that there was a slight shift from smaller clay-sized particle fractions in the influents toward larger diameter clay- and silt-sized particles in the effluents. In a similar manner, the total particle size distribution curves for background water samples were shifted toward even larger clay-sized particles, indicated by the greater than 50 and 80 percent values (based on the total number of particles counted) shown in Table 9. This increasing trend in particle size was highly significant ($p \leq 0.01$, using the F-test), and the most apparent cause for this shift would be a slight decrease in the overall specific gravity of the suspended particles with increased residence time. This might occur if heavier clay-sized mineral sediments preferentially settled, leaving behind a greater percent of lower specific gravity silt-sized particles. The larger particulate matter could primarily include organic detritus, although this was not substantiated by the average solid phase organic carbon data. There could also be an overall removal of a very fine particle fraction, such as might result from the slow coagulation of fine (e.g., iron or humic matter) precipitates in the water column with increasing detention time. Ferric hydroxide and humic matter precipitates are also known to form spongy, low-specific-gravity solids.[14] It was not possible to draw definite conclusions from the data, only the above inferences.

168. Cation exchange capacity. Cation exchange capacity could only be determined for six effluent solids, but the results show a significant increase ($p \leq 0.01$, using the t-test) during slurry retention in the disposal areas, with influent and effluent values of 50.9 and 82.5 meq/100 g of suspended sediment, respectively (Table 9). This change probably results from an overall decrease in the particle size and increase in the surface area of the effluent solid phase. Most effluent solids are thus capable of sorbing or desorbing considerable quantities of ions in solution. The relatively high metal concentrations in the exchangeable phase of the geochemical phase partitioning of

influent and effluent solids (Table 10 and Figure 18) show the importance of cation exchange capacity for governing the mobility of trace metals. The partitioning data will be discussed at a later time.

Chlorinated pesticides and PCB's

169. DDT and degradation products. DDT and its degradation analogs, DDE and DDD, were measured in samples from six disposal areas (Table 8). At four of these sites (Wilmington, N. C.; Richmond, Va.; Lake Charles, La.; and Vicksburg, Miss.), only op′ and pp′ DDE were detected in total influent slurries, with a lower detection limit of 0.01 mg/ℓ; pp′ DDE was highest in all cases with average values of 0.47 mg/ℓ in total influents and 0.37 mg/ℓ in total effluent samples, showing only a 21 percent removal during containment (Figure 14). At the Lake Charles dredging area, the highest op′ and pp′ DDE concentrations were present in the Calcasieu River Ship Channel surface background water. This is not surprising since continual resuspension of fine sediments was observed to result from heavy barge and ship traffic perturbations in the channel. Influents from Richmond and Vicksburg also showed high concentrations of DDE, with 51 and 98 percent removal during containment, respectively. The fine-grained sediments dredged from the dock area at Richmond on the final day of sampling showed the highest DDE levels. At the Wilmington (Eagle Island) site, an effluent containing 1.8 percent total suspended (fine-grained) solids had a higher DDE concentration than an effluent with 4 percent sand and a total solids concentration of 2.85 percent. Thus, it seems that DDE is primarily associated with a very fine clay-sized fraction at most of the sampled sites. This is in agreement with the results of other studies.[74,147]

170. DDT and DDD concentrations seemed to show common trends, as both compounds were either high in concentration (e.g., Grand Haven, Mich., and Southport, N. C.) or were at nondetectable levels. DDT and DDD concentrations were especially high in sediments dredged from the Elizabeth River, near Southport, N. C. DDD seemed to be effectively removed during land containment, with all effluent values being below detection limits. Although DDT seemed to be removed less efficiently,

it was also higher in influent samples. Thus, both DDT and DDD appear to be associated with a larger particle size fraction, which would facilitate their removal in land containment areas.

171. The direct relationship between DDT and DDD concentrations is not unusual in reduced sediments as the conversion of DDT to DDD isomers is promoted by anaerobic conditions,[69,147] while the DDE isomers form more readily under oxidizing conditions.[70] This is further substantiated by the lack of detectable DDD in oxidized surface background water samples, while the same is not true for DDE (Table 8). Thus, the DDD is probably formed in the anaerobic bottom sediments from DDT, while the DDE is probably derived mainly from surface runoff and aerial fallout from areas adjacent to the water regime. As DDT is degraded more rapidly under anaerobic conditions, through the DDD pathway,[147] land disposal of dredged sediments may tend to impede the removal of DDT analogs. However, DDD was found to be significantly degraded within 1 day in anaerobic lake water, while a similar degree of degradation was observed in 8 weeks in flooded soils.[147] Thus, further research seems necessary to properly evaluate the overall impact of land disposal on DDT breakdown. Regardless, DDE isomers seem to be by far the most common chlorinated pesticide in dredged material and also one of the most difficult to remove by land containment. This also appears to be the case for nearshore marine dredged sediments from the California coast, where pp´ DDE often accounts for 60 to 70 percent of the total chlorinated hydrocarbons.[74]

172. PCB's. PCB's were detected in influents from all six of the monitored sites (Table 8). The lower detection limit was 0.1 mg/ℓ for most site samples. However, the dredging and land disposal operation at Seattle, Wash., was initiated as a result of a localized spill of PCB (Aroclor 1242) in slip 1 of the Duwamish River, and thus great care was given to detecting PCB's at the part per trillion level. The highest recorded concentration of PCB's was in an influent sample from Grand Haven, Mich., where 21.0 mg/ℓ was extracted (Appendix B, Table B12). Aroclor 1254 accounted for 77 percent, and the remainder was Aroclor 1260. An influent sample from the dock area of Deepwater

Terminal at Richmond contained 13.7 mg/ℓ PCB's (mostly as 1242 and 1254), while an influent sample from the Seattle spill site contained 11.2 mg/ℓ, primarily as 1242 (Appendix B, Tables B14 and B16). The highest average PCB content for influents was 10.7 mg/ℓ at Grand Haven, while the lowest PCB influent concentration was 1.3 mg/ℓ at Wilmington, N. C. (Table 8). Thus, PCB's appear to be ubiquitous in sediments from various geographical and environmental areas. The influent average for all six sampled sites was 5.8 mg/ℓ (Table 9).

173. The highest concentration of PCB's in an effluent sample was 7.66 mg/ℓ. It represented a high solids sample from Grand Haven, with 84 percent of the total being Aroclor 1254 and the remainder Aroclor 1260. An effluent from Wilmington showed the second highest value of 1.44 mg/ℓ, with 76 percent being 1254 and the remainder as 1260. Only one effluent sample from Richmond contained detectable PCB's, with 1.33 mg/ℓ being recorded, again with no detectable Aroclor 1242 and with 1254 accounting for 65 percent. The average for all of the effluent samples analyzed was 0.50 mg/ℓ. The decrease of PCB's in effluents and background water was significant ($P \leq 0.01$, using the F-test) with the six sampled containment areas retaining better than 90 percent, but with effluent concentrations being ten fold higher than the overall background level of 0.058 mg/ℓ. The effluents typically contained either high or nondetectable (< 0.1-mg/ℓ) concentrations, and the high PCB values were always associated with high levels of suspended solids. However, because of the small number of effluents containing detectable levels, there seemed to be no good correlation between removal efficiency and retention time. An additional study at the Seattle site,[26] which describes in great detail the PCB findings, shows that better than 99.8 percent of PCB's can be removed after only a short residence time by using flocculents to remove the fine sediment fraction. Again, because of the great efficiency in suspended sediment removal, no relationships could be made with particle size, although it appears that PCB's may be mainly associated with larger sediment particles and are thus readily removed during land containment. A more detailed study, showing the relationship of chlorinated pesticides and PCB's with different sized

particulate fractions, is given in a separate WES Technical Report.[145]

Sulfides and disposal area sediment Eh

174. Sulfides. The sulfide content of sediments is important because most heavy metals form very poorly soluble complexes with sulfides. However, most newly formed sulfides remain stable only at negative Eh potentials, and oxidizing (positive Eh) conditions can promote a rapid dissolution of some elements bound with sulfides. This results from the microbial conversion of sulfide to sulfate. Thus, shifts in Eh can result in changes in the mobilities of complexed chemical species, especially many trace metals. Different sulfide complexes vary in their stability and rates of oxidation under increasing Eh. For example, HgS is less rapidly oxidized that CdS under similar environmental conditions,[55] but monomethyl mercury forms relatively soluble sulfides even under highly reduced conditions.[102] Iron forms one of the more soluble sulfide complexes in recent sediments, and thus the fixation and release of heavy metals and highly toxic hydrogen sulfide are usually governed by the iron concentrations in sediments. Eventually, additional sulfide is tied up in iron disulfide (pyrite) which is very poorly soluble and more resistant to oxidation at most sediment Eh and pH values. Therefore, the sulfur cycle is especially important for heavy metal recycling and organic matter interactions in diagenetically more recent sediments, including most dredged material.

175. The total sulfides in influent slurries from the nine sampled disposal areas decreased by over 95 percent during containment (Figure 14). The amount associated with the solid phase decreased by about half, from 493 mg/kg in influent solids to 208 mg/kg in effluent solids (Table 9). It is quite possible that much of this solid phase decrease was attributed to sulfide oxidation, unless the sulfides were mainly associated with the coarser sediment fraction, which is doubtful. Sulfide oxidation could result in a significant transfer of metals from poorly soluble sulfide complexes to other complexes of varying solubilities.

176. Sediment oxidation-reduction status. Eh measurements in sediments appear to closely approximate the hydrogen sulfide activity

present. This is mainly because of the rapid and reversible nature of the sulfide-elemental sulfur electron transfer reaction.[148] Thus, a general idea of the sulfide activity as well as an estimate of the overall oxidation-reduction status of freshly dredged sediments in confined disposal areas can be obtained by inserting bright platinum electrodes into unconsolidated sediments within each site. Between 6 and 40 measurements were made within each of eight disposal areas, in regions which were considered to have maximum contact with the dredged material slurry. The average Eh values are given in Table 12. Since Eh effects are influenced by pH, both measurements were made in adjacent locations. Generally, the sediments were very close to neutral pH, which is not unusual for anaerobic environments.[10] The data show that sediments in the Richmond disposal area, averaging +319 mV, were sufficiently oxidized in the upper 4 to 6 cm to allow for the precipitation of iron complexes. Also, dike sediments in pond 2 at the Seattle, Wash., site were considered to be oxidized, with an average Eh of +271 (pH 6.1). The majority of the sediments dredged at Richmond were composed of coarse sand and gravel, which would have allowed for rapid infiltration of oxygenated surface water. The dike sediments measured in pond 2 at Seattle were not dredged and consisted of a coarse-grained sandy loam. Thus, oxygenated water, which was present in this pond, could again migrate to the probe tips. Dredged sediments at Southport, Seattle, and Houston were considered to be highly reduced with average Eh potentials of -330, -124, and -108 mV, respectively. The Southport and Seattle sediments consisted of fine-grained black mud which were later found to contain the highest sulfide concentrations, often in excess of 500 mg/ℓ. The Houston sediments were very high in weathered petroleum, which could have promoted low Eh values through electron transfers mediated by reduced organic compounds. The remaining sites could best be grouped as moderately reduced, and their potentials appear to be poised mainly by moderately high sulfide concentrations (e.g., Lake Charles) or by accentuated microbial degradation, induced by high organic and nutrient levels in the dredged material and site sediments (e.g., Wilmington). The low Eh values for the Vicksburg lake sediments are difficult to explain on the basis of the associated chemical data.

177. Eh measurements cannot be considered quantitative because of various mixed and nonreversible reactions.[58] However, the Eh findings give a general idea of how the sediments in disposal areas may impact the dredged slurry. Effluent Eh or dissolved oxygen data can be misleading. They often tell very little about the length of time that the measured conditions existed. Rates of change are important, and bottom sediment Eh can greatly affect changes in the water above it. An example of this was seen at the two-compartment disposal area at Seattle. Usually the surface water leaving the first pond was very clear and low in suspended solids (Photo 15). In contrast, water in the second pond showed highly noticeable orange-colored turbidity, caused by the precipitation of soluble iron in the water (Photo 14). The Eh values substantiated the above, with sediment values for ponds 1 and 2 of -093 and +271 mV, respectively; the dredged material in pond 1 had an average Eh of -124 mV. Dissolved oxygen data for effluents (surface water) from ponds 1 and 2 averaged 5.4 and 6.5 mg/ℓ (Table 6; Appendix B, Table B7), respectively, which would not have shown the full impact of the dredged material on the ponded water. Apparently the turbulent mixing within pond 1 promoted the continual dissolution of iron precipitates which may have formed in the aerobic surface water.

Organic carbon and oil and grease

178. Organic carbon. Organic carbon averaged about 2.5 percent in influent solids, (range of 0.1 to 5.3 percent) with no significant decline in the solid phase of effluents. However, on a volume basis, organic carbon showed about a 96 percent decrease from a net influent value of 3880 mg/ℓ to a net effluent concentration of 151 mg/ℓ (Table 9 and Figure 14). Soluble phase organic carbon did not show a significant decrease during land containment, with rather narrow ranges and standard deviations. Data for each site indicate that in most instances the soluble carbon remains quite constant in influent and effluent filtrates (Table 9 and Figure 16). The major exception was the Houston disposal area, where effluents showed an overall decrease of about 35 percent from high influent values. The high concentrations of petroleum

residues in the dredged material probably accounted for this decrease, with most of the loss possibly attributable to the evaporation of volatile organic compounds from the surface of the water; a Kerosene-base, odor-controlling chemical was also periodically added to the slurry in the influent pipe at the Houston site, which would readily evaporate at the mild water temperatures. However, it appears that most of the organic carbon in dredged material is highly resistant to rapid degradation or volatilization, including most highly weathered petroleum residues which are often associated with dredged sediments.

179. Oil and grease. Oil and grease was very high in dredged material from both the Houston (Ship Channel) and Seattle (Duwamish Waterway) sites. Total concentrations in the Houston and Seattle influents averaged about 600 mg/ℓ, while concentrations in the effluents decreased by 88 and 85 percent, respectively. However, values of 73.5 mg/ℓ and 89 mg/ℓ for these effluents warrant concern. The average oil and grease value for effluents from all of the sampled disposal areas was 27.5 mg/ℓ (Table 9 and Figure 14). The oil and grease fraction can include other organic compounds besides petroleum residues, such as various indigenous oils, fats, waxes, and some humic matter.[141,142] However, most of the effluent oil and grease is probably derived from emulsified petroleum residues, which frequently form a fluid, stratified layer in the deeper water of disposal area settling ponds. About 85 percent of the total oil and grease recovered from Seattle effluent samples resulted from one sample, which was collected when the final settling pond was pumped to a low level; high total oil and grease values for Houston effluents were also obtained when effluent discharge over the weir was not properly regulated, allowing for the upwelling of the emulsified oily stratum from near the bottom of the settling pond. Oily dredged material appears to require a longer residence time for clarification due to the formation of low-density emulsion layers which are often difficult to destabilize.

180. Pronounced surface oil slicks were not observed at the dredging sites, including Houston and Seattle. This may have resulted from continual wind disruption and filtering by vegetation and debris in the

disposal area. However, the accumulation of thick organic surface scums can be a problem at some sites, although their composition frequently includes other organic materials not directly derived from petroleum.[43]

181. Based on data collected in this study, oil and grease remaining after high-speed centrifugation of effluents (to approximate 0.45-μm filtration) are not a problem from a quantitative viewpoint (Figure 17). Based on data from six sites, soluble phase oil and grease rarely exceeded 5 mg/ℓ in effluents (Tables 8 and 9). However, many low-molecular-weight, more soluble organic compounds derived from petroleum are highly toxic to organisms.[79] A detailed study of the composition of influent and effluent oil and grease fractions is included in a related DMRP study.[145]

Nitrogen

182. Organic and ammonium nitrogen. The average total Kjeldahl nitrogen (organic plus ammonium -N) concentration in the disposal area influents was 189 mg/ℓ or about 1930 mg/kg (0.19 percent) in dry weight dredged sediments. As shown in Table 9 and Figure 14, organic nitrogen compounds contributed about 89 percent, averaging 168 mg/ℓ (including exchangeable ammonium). The total (filterable plus exchangeable) ammonium nitrogen averaged 45.6 mg/ℓ, with ranges of 7.3 to 86 mg/ℓ in sampled influents. Approximately half of the total ammonium nitrogen in the influents was recovered from sodium acetate extracts of the solids, indicating that this nitrogen could be desorbed from the solid phase by exchange reactions with other ions. Filterable ammonium nitrogen in influents averaged 20.8 mg/ℓ. The total ammonium is possibly available to benthic organisms but only the filterable ammonium represents what would be rapidly released from perturbed bottom sediments. Considering the average sediment pore water dilution factor due to dredging to be about 6.3, based on an influent solids value of 11.7 percent (Table 9) and 60 percent of bottom sediments being pore water,[26] ammonium nitrogen concentrations in the pore water of the bottom sediments could average about 130 mg/ℓ; however, additional ammonium may be released from solids as a result of the dredging operation.

183. Total combined nitrogen decreased by about 87.5 percent in the disposal area effluents, with an average concentration of 23.7 mg/ℓ (Table 9); the background water contained 1.1 mg/ℓ total combined nitrogen (Figure 15). These variances were highly significant ($p \leq 0.01$) using the F-test.

184. The effluents also showed a significant decrease (using the t-test comparison) in total ammonium nitrogen of 57 percent, with an overall concentration of 19.6 mg/ℓ (Table 9 and Figure 14). However, the exchangeable ammonium associated with the solids increased from 110 to 196 mg/kg dry weight (Table 9). The exchangeable ammonium contributed only a small fraction of the total effluent ammonium nitrogen concentration except when high effluent solids were prevalent. The increase in exchangeable phase ammonium is the result of an increase in the specific surface area of the suspended effluent particles, as shown by a respectively larger cation exchange capacity. Filterable effluent ammonium nitrogen averaged 13.6 mg/ℓ. Total organic nitrogen contributed only about a third of the nitrogen in effluents from the confined disposal areas.

185. Nitrate + nitrite nitrogen. Nitrate + nitrite nitrogen were not found to be a problem in the monitored disposal area effluents as background water samples typically showed higher concentrations. Additionally, the highest monitored effluent value of 1.8 mg/ℓ (Table 9) should not create a serious impact on water quality near the discharge point. The noticeable increase in effluent nitrate from the Grand Haven disposal area in Michigan was derived from river water concentrations at the dredging site, as indicated by a comparable background water value (Table 8); nitrification would have been inhibited at the near-freezing water temperatures experienced during sampling (Table 6). However, nitrate accumulation had been observed in an earlier study,[34] monitored in the Penn 7 site on the Maumee River in Ohio during August to November; an effluent retention time of about 2 months appeared evident from the data.

186. Retention time. Table 14 shows the relationship between the retention time of dredged material in several disposal areas and the

decrease in ammonium and nitrate nitrogen. For the Lake Charles site, the initial daily influent should closely represent the respective daily effluent because of the very short observed residence time of only a couple of hours. The effluents collected on the final two sampling days should more closely correlate with the average influent value for the 3 days because of a residence time of perhaps several days. Although the overall data (Tables 8 and 9) show that the ammonium nitrogen decreased during land containment, the more specific comparisons, shown in Table 14 and Figures 15 and 16, fail to show a good relationship between residence time and ammonium removal. These data suggest that much of the observed ammonium nitrogen removal occurs very rapidly, which suggests that sorption reactions with the disposal area sediments might be an important removal mechanism in many containment areas.

187. Several disposal areas, especially at Houston, Grand Haven, and Wilmington, experienced high influent ammonium levels with very little removal during disposal area retention (Table 8). For Houston and Grand Haven the retention times were short, being usually less than 12 hours. Sediment sorption of ammonium in the Grand Haven site could have been impeded by its small size and lack of slurry contact with the sediment, lack of sorptive capacity of the sandy sediments, and a net release of ammonium from the high-organic-content dredged material. The mostly sandy, highly petroliferous sediments at Houston may have also inhibited sorption reactions. The masking of sorption sites by petroleum hydrocarbons has been documented.[78] The very large overland flow treatment area at Wilmington was sampled in the winter, when most of the thick vegetation in the disposal area was dead or dormant. The poor ammonium removal may thus be related to the decomposition of plant material, which could create seasonally high pulses of nitrogen in effluents. Actively growing vegetation in the Southport disposal area seemed to provide for very efficient nitrogen removal initially, although this efficiency decreased somewhat as much of the vegetation was buried beneath dredged material and ponded water; the nitrogen contribution from organic matter decay could not be assessed. The data from the Southport site suggest that actively growing vegetation can

facilitate a high removal efficiency of ammonium nitrogen from vegetated land containment areas, as this disposal area showed the best removal compared to the other monitored sites. The vegetation interaction study will be discussed in greater detail in a later section. Temperature effects on nitrogen removal could not be critically evaluated as mild temperatures prevailed at most sites during the sampling periods, except for the near-freezing water conditions at Grand Haven (Table 6).

188. High concentrations of ammonium nitrogen in disposal area effluents seem to be a major problem for the land containment of dredged material. The high ammonium levels in influents are not surprising since ammonium is also found in the interstitial water of some bottom sediments in very high concentration.[14] Figure 18 depicts the nitrogen cycle which is operative in most bottom sediments. Ammonium nitrogen is stable and tends to accumulate in reduced sediments, and nitrate, which forms only in aerobic environments, is unstable and does not accumulate. Rapid diffusion of ammonium out of sediments is impeded by exchange reactions on the surfaces of negatively charged solids and by the presence of a thin oxidized zone at the sediment-water interface. Ammonium that does diffuse from the sediments is rapidly utilized by microorganisms near the bottom or, if the pH of the water becomes alkaline, volatilization of un-ionized ammonia may also lead to a depletion of nitrogen. Dredging results in a rapid release of ammonium to the solution phase. Rapid conversion of ammonium to nitrate should be favored in land containment areas, since oxidizing conditions generally prevail. However, the nitrate nitrogen data shown in Table 14 and Figures 16 and 17 fail to show a direct relationship between residence time and effluent nitrate concentrations. Perhaps nitrate loss by denitrification at anaerobic zones in the sediment-water slurry may counterbalance the rate of nitrate formation. Also, a low rate of nitrification may result from a low population of nitrifying bacteria. Bacteriological analyses were not conducted in this study.

189. Summary. The findings suggest that residence time affects different disposal areas dissimilarly, with the net nitrogen removal being dependent on the overall rates of uptake and release, both of

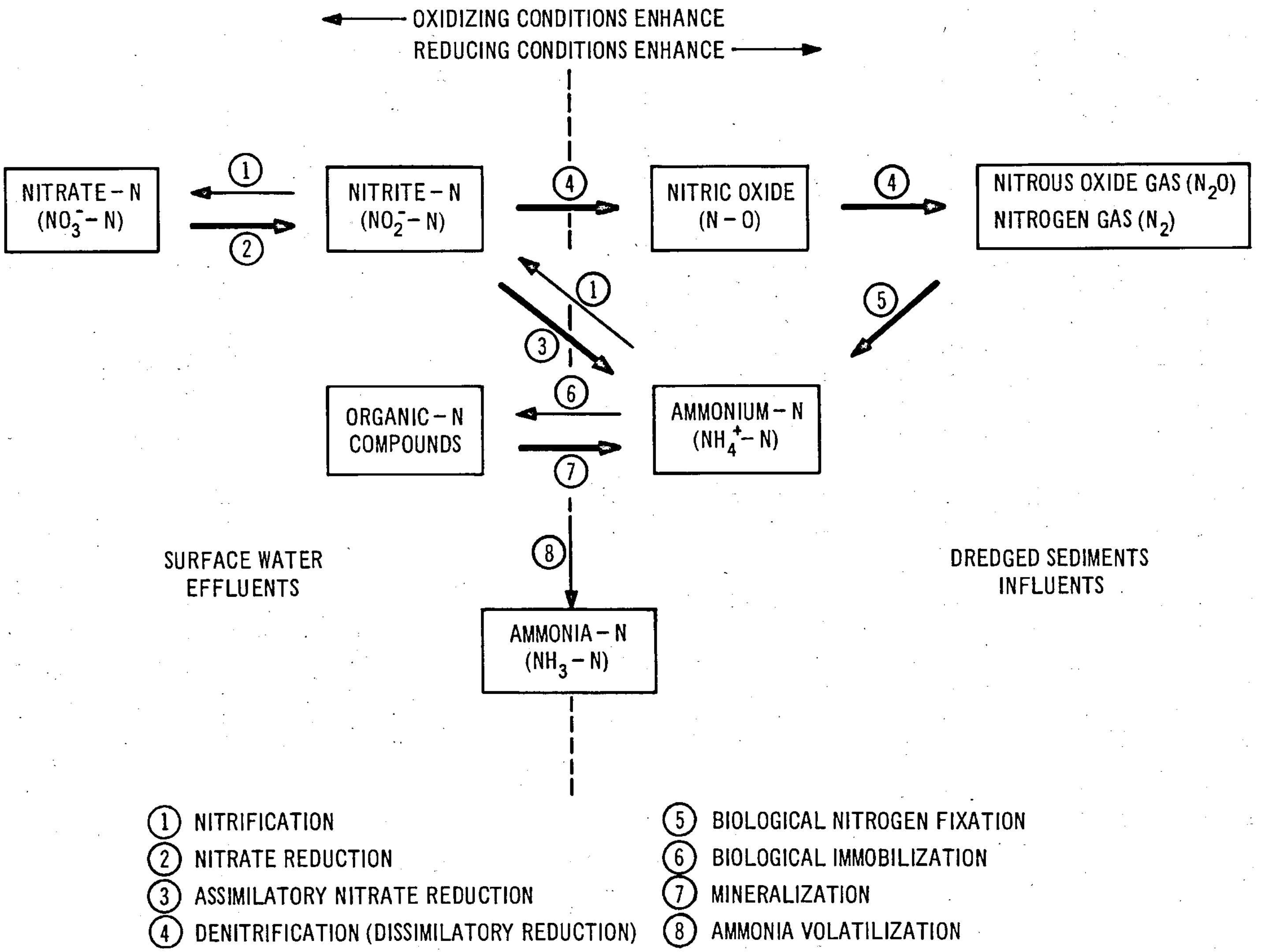


Figure 18. Nitrogen transformations which could occur in dredged sediments and confined disposal area slurries, with bold arrows showing the main pathways favored in most dredged sediments, promoting high concentrations of ammonium; ammonia volatilization is only important at pH values above 8.5

which seem to occur simultaneously. A nitrogen mass balance was not attempted at any of the sites to verify the sources or sinks for the nitrogen. Actively growing vegetation in disposal areas appears to contribute to the removal of nitrogen from dredged material slurries. However, the removal of dead annual vegetation and surface litter during the dormancy period may be warranted. This would remove the immobilized nitrogen in the plant tissues and prevent the possibility of high nitrogen pulses in effluents discharged during winter disposal and precipitation runoff. This management scheme would be most practical if the vegetation consists primarily of rapidly growing seasonally dormant or annual grasses and shrubs, especially plants such as common reed grass (Pragmites communis) which readily regenerate from the roots. Most of the disposal areas contained seasonal vegetation. Upland forest vegetation is not recommended for the disposal of brackish water dredged material because of resultant salt damage to the plants. A long retention time, lack of vegetation, and high ammonium concentrations in disposal areas all contribute to a high effluent pH as discussed earlier. Considerable ammonia loss through volatilization could occur at pH values above 8.5, but the high toxicity of ammonia may be detrimental to organisms near the effluent discharge area.

Phosphorus

190. Total phosphorus. Total phosphorus varied greatly in influent dredged material, with dry weight solids ranging from 639 to 4400 mg/kg and a mean value of 1850 mg/kg. Total influent concentrations ranged from 12.8 to 496 mg/ℓ, with an average concentration of 155 mg/ℓ (Table 9).

191. Effluents contained an average total phosphorus concentration of 11.7 mg/ℓ, with ranges of 0.1 to 82.1 mg/ℓ (Table 9). Thus, the sampled disposal areas showed a net decrease of 92.5 percent, with most sites showing total phosphorus removal efficiencies of greater than 95 percent (Table 8 and Figure 14). The disposal area at Houston, Tex., showed the poorest total phosphorus removal, with an efficiency of only 74.5 percent.

192. Soluble phosphorus. Total phosphorus in the less than 0.45-μm filtrates very closely duplicated the respective orthophosphate phosphorus concentrations in most influent and effluent samples (Table 8); thus, filterable (soluble) phosphate is mainly in the orthophosphate form. Influent filterable phosphate averaged around 0.8 mg/ℓ with ranges from trace quantities to 9.5 mg/ℓ for a Southport, N. C., sample (Table 9; Appendix B, Table B18b).

193. The Clinton site in Houston also showed the greatest release of soluble phosphorus, with a four fold increase during the short residence time, to an average effluent value of 1.05 mg/ℓ (see Appendix B, Table B11). This contrasts to the average decreases in total soluble phosphorus and orthophosphate phosphorus for all of the sites of 62 and 77 percent, respectively (Figure 14).

194. Most of the effluent and background water samples contained around 0.1 mg/ℓ soluble phase phosphorus (Figures 15 and 17; Table 8). Thus, it seems that the disposal site waters had approached an equilibrium condition with the dredged sediments in most instances. There was no obvious relationship between alkalinity or iron concentrations and soluble phosphorus levels. However, soluble phosphorus concentrations are most often controlled by iron and calcium carbonate solids.[14, 90] In areas which are continually subjected to sediment perturbations, such as in ship channels, high phosphorus concentrations in background water are not unexpected, especially for highly reduced sediments. The dredging of black, highly organic, fine-grained estuarine sediments (e.g., Southport and Seattle dredging sites) seemed to release the highest levels of soluble phosphorus, as shown by high influent concentrations. A possible correlation was noted in another study[16] between the mobility of phosphorus and the oil and grease content of sediments. It may be more than coincidental that the Houston site was noted for very high oil and grease levels and also for the greatest release of soluble phosphate during disposal area retention of the dredged material, resulting in high effluent levels of greater that 1 mg/ℓ (Tables 8 and B11). This relationship should warrant further study.

Calcium, magnesium, potassium, sodium, chloride, and alkalinity

195. Soluble forms. Most dredged material did not show noticeable increases in soluble calcium, magnesium, potassium, and sodium during land containment, and overall variations between influent, effluent, and surface background water were not significant (Table 9). However, certain sites showed notable increases in soluble calcium, with the large vegetated overland flow treatment system at Wilmington showing greater than a two fold increase. The Wilmington site also showed the highest alkalinity increase during treatment (Table 7 and 8; Appendix B, Table B13). Surprisingly, the slurry pH in this site increased from 6.6 for influents to 7.4 for effluents. The low influent pH probably resulted from acidic conditions in the bottom sediments, which would have promoted the dissolution of calcium carbonate; this acidity may have been aided by the decomposition of the high levels of organic matter present in the sediments. The slightly alkaline effluent pH was most likely promoted by algal photosynthesis in the nutrient-rich surface water in the site. However, the slightly acidic disposal area sediments, with an average pH of 6.75 (Table 12), may have induced continual calcium carbonate dissolution, counteracting any precipitation occurring in the water flowing over the sediments. This is indicated by the geochemical phase partitioning data for the site (Table 10; Figure 19), where carbonate phase calcium remained at similar levels in both influents and effluents. The vegetated Southport disposal area also showed small increases in soluble calcium. Although alkalinity was very high in the influents, effluents showed noticeable decreases. The reason for this decrease is not certain, but calcium or magnesium carbonate formation appears not to be the most important mechanism.

196. Soluble magnesium also increased in effluents from the Wilmington site. Grand Haven was the only other site to show a noticeable change in soluble magnesium, with a ten fold decrease in effluents. The reason for this decline cannot be determined from the data. Soluble potassium and sodium showed no major overall or site-

specific changes in influents and effluents, although surface background water from salt wedge estuaries (e.g., Houston, Wilmington, and Lake Charles) showed decreases associated with the lower salinities. At the Grand Haven site, effluent dilution by precipitation was reflected by decreases in soluble sodium and potassium (Table 8).

197. Chloride concentrations showed a closely parallel relationship with soluble sodium in influents, effluents, and background water (Tables 8 and 9). The sodium and chloride concentrations indicated that a net evaporative water loss and resultant salinity increase were not prevalent during the winter and spring sampling at the nine different disposal areas. However, this trend may not persist during warmer and dryer summer months. The net soluble calcium, magnesium, potassium, and sodium concentrations were relatively close to their abundance in seawater, with calcium and potassium showing slight increases and magnesium and sodium showing slight decreases. This trend reflects the greater number of brackish water sites which were sampled.

198. Total concentrations. Sodium concentrations in total influent and effluent samples were very similar to values in their respective filtrates. This indicates that the minerals present in the sediments from most of the sites were not rich in sodium. Calcium, magnesium, and potassium were mainly associated with the solid phase of influents, but, as a result of solids removal, they were mainly associated with effluent filtrates. As will be discussed in later sections, potassium seems to be predominantly associated with a crystalline particulate fraction, with little geochemical phase change during slurry confinement (Figure 19). It is possible that potassium-rich suspended clays may be a major contributor to solid phase potassium. Geochemical phase partitioning data (Table 10) fail to indicate that exchangeable potassium changes appreciably in confined dredged material.

199. Alkalinity. Average alkalinity for nine sites showed a poor relationship with average calcium and magnesium concentrations. However, significant decreases in alkalinity ($p \leq 0.01$, using the F-test) existed between the influents, effluents, and background water samples (Table 9). This trend is in agreement with the elemental partitioning data for

sediments from four of these sites, which show an increase in the carbonate phase of effluent solids for many trace metals, but with a slight decrease in carbonate phase calcium (Table 10 and Figure 19). Because of the complexity of chemical reactions and the great number of chemical species contributing to alkalinity, only site specific inferences can be drawn. These will be discussed in a later section. However, biological activity probably has a major impact on alkalinity and carbonate phase changes in sediments from confined disposal areas. At Wilmington, the alkalinity and carbonate phase increases may result from intense microbial degradation of the dead plant material in the thickly vegetated site. At the Lake Charles site, decreases of alkalinity in different effluent water samples nicely corresponded to the abundance of planktonic algae observed in the samples; photosynthesis tends to induce the precipitation of carbonates.

Iron and manganese

200. The mobilities of iron and manganese are strongly influenced by pH and oxidation-reduction changes in the environment, with greater mobility generally resulting under more acidic and reducing environmental conditions. Although total iron levels in sediments are often around 4 to 5 percent, usually less than a ten-thousandth of this is released to the solution phase. Reasons for this are the rapid reaction of ferric iron with water under aerobic conditions, forming very insoluble iron hydroxide and hydrous oxide compounds. Under reduced conditions, iron tends to form poorly soluble sulfide complexes;[14] also, anaerobic sediments tend to be buffered at near neutral pH.[10] Thus, high concentrations of soluble iron occur mainly where acidic waters come in contact with sediments or under reduced conditions in the absence of high sulfide levels. Manganese does not have as great of an affinity for sulfides as iron, and it will also not form the highly insoluble hydroxide complexes as rapidly as iron under oxidizing conditions.[14] Thus, manganese has a greater tendency to be released into the water phase of dredged material and will remain in the soluble form for a longer period of time than will iron.

201. Total constituents. Total acid digests of the solid phase of influent dredged material gave average values for iron and manganese of 4.2 and 0.06 percent, respectively (Table 9). These concentrations are similar to averages found in the earth's crust.[149] The average concentrations of iron and manganese in total influents were 3400 and 63 mg/ℓ, respectively. These values reflect the nearly ten fold dilution of dredged sediments with bottom water at the dredge site.

202. Total effluent iron values for all of the sites averaged 193 mg/ℓ, giving an overall removal efficiency of greater than 94 percent. The major mechanism for this removal seemed to be the settling of solids, including some of the iron hydroxide precipitates which formed in the aerobic surface water within the land containment areas.

203. Total manganese showed an overall decrease of 87.5 percent in effluents, with an average value of 7.9 mg/ℓ.

204. Soluble phase constituents. Because of the low solubility of iron sulfide and hydrous oxide complexes, filterable iron in influents averaged only about 3.5 mg/ℓ (Table 9), or about 0.1 percent of the total iron concentration. However, a Wilmington influent sample contained 15.9 mg/ℓ soluble iron, while one from Lake Charles gave a value of 14.6 mg/ℓ.

205. Soluble influent manganese showed a close relationship with the soluble iron values. The average value for soluble phase manganese in the influents was 2.35 mg/ℓ, which represented 3.7 percent of the total manganese present (Table 9). The highest net soluble manganese release was observed in influents from the same sites which showed high iron mobility, namely, Wilmington (4.4 mg/ℓ), Lake Charles (10.4 mg/ℓ), and Vicksburg (3.2 mg/ℓ). Values of around 2.5 mg/ℓ were also observed in Southport influents (Table 8).

206. Soluble iron showed better than a 75 percent decrease in effluents with an average value of about 0.8 mg/ℓ and a range from 0.01 to 10.1 mg/ℓ (Table 9; Figures 16 and 17). It is also possible that some of this filterable iron actually consists of very fine iron hydroxide precipitates, which have been found to pass through 0.45-μm filters.[150] This was noted to occur at the Seattle site, where the

effluents consisted mainly of various sized iron precipitates.* The background water samples showed only a slight decrease in the soluble phase iron over the effluent levels, which further verifies the rapid rate of iron precipitation and coagulation in confined disposal areas. The variance between the influent, effluent, and background water values was found to be highly significant ($p \leq 0.01$) by the F-test (Table 9).

207. Soluble manganese showed a significant reduction of about 38 percent during disposal area treatment, with an average effluent concentration of 1.45 mg/ℓ and range from 0.002 to 7.95 mg/ℓ (Table 9 and Figure 14).

208. Geochemical partitioning. Geochemical phase partitioning data (Table 10 and Figure 19) suggest that iron is not commonly associated with the exchangeable phase of dredged solids and the majority resides with the more resistant phases (e.g., easily reducible, moderately reducible, organic-sulfide, and residual phases**). Because of the high levels of iron in the dredged sediments, a small shift to one of the more resistant chemical phases could very readily result in the observed decreases of soluble iron during containment. The partitioning data show only a very small increase in the easily reducible phase, which includes fresh iron hydroxide precipitates. This clearly shows the insignificance of soluble phase iron in the geochemical cycle.

209. Geochemical phase partitioning data for manganese (Table 10 and Figure 19) indicate that the exchangeable and carbonate phases of dredged material solids tie up significant amounts, with each phase accounting for about 20 percent of the total manganese. The greater potential mobility of elements in these phases may be reflected by the greater relative mobility of manganese compared to iron. The major phase shifts for manganese in effluents are a 5 percent decrease in

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* Personal communication, Dr. J. Blazevich, EPA Region X Laboratory, Seattle, Wash., April 1976.

** The moderately reducible phase refers to metals which are primarily associated with crystalline iron oxide coatings on the solids; the residual phase includes the metals remaining in the solid phase after the previous extraction, and represents a total dissolution of the particulate minerals by hot acid treatment.

exchangeable manganese and a 10 percent increase in easily reducible manganese. Perhaps much of the exchangeable phase manganese is slowly precipitated on the surfaces of the associated solids or trapped within iron precipitates as they continue to grow and age during confinement.

210. Mechanisms governing iron mobility. The release of iron seemed to be site specific, and high levels did not always relate to unusual influent pH or Eh conditions. For example, Wilmington, Lake Charles, and Vicksburg influents showed the highest average soluble iron concentrations, with net values of 12.4, 8.6, and 10.5 mg/ℓ, respectively (Table 8). However, influent dissolved oxygen and sediment Eh values in these disposal areas did not indicate any obvious similarities in these conditions. Perhaps changes in Eh were of major importance, but these changes could not be properly assessed. High soluble iron levels at the Wilmington dredging site could have resulted from interaction of mildly acidic background water with the bottom sediments; the surface background water had a pH of 5.5. The presence of high alkalinity and low sulfides in the influent samples suggests that moderately soluble iron carbonate (siderite) may have been the solubility-controlling solid in the Wilmington sediments. Acid water conditions as low as pH 4.2 have also been recorded periodically in the Calcasieu River, near the Lake Charles site,* although surface background water samples collected during dredging gave a pH value of 6.3 (Table 6). It is inferred that iron release may result from several combinations of factors which are difficult to correlate with the limited data available, but carbonate and organic complexation are considered to be important. All of the sites showing a high release of iron also showed high influent alkalinity values (Table 7). There seemed to be little relationship between total organic carbon or total iron in the sediments and the release of soluble phase iron (Table 8).

211. Mechanisms governing manganese mobility. The correlations with other parameters are equally as difficult with manganese as they were for iron. The high manganese solubility in the Wilmington disposal

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* Written communication, A. J. Heikamp, U. S. Army Engineer District, New Orleans, February 1976.

area also seems to be regulated by carbonate phase complexes, with 37.5 percent of total manganese present in the acetic acid extractable (carbonate) phase (Table 10). The solubility of manganese at many of the sampled disposal areas could also be regulated by soluble organic complexes, which are well known to alter the theoretical solubilities of heavy metals.[9,14] A qualitative determination of the organic compounds present in the disposal areas was not performed. The data suggest that the major mechanism for the removal of soluble manganese in confined disposal areas is precipitation of manganese hydroxides and oxides and coprecipitation with oxidized iron precipitates, as indicated by the increase in the easily reducible phase. Perhaps some of the manganese in effluent filtrates is also associated with very small filterable solids (see Figure 20). The poorer removal efficiency for soluble manganese is probably related to its slower rate of oxidation and precipitation under oxidizing conditions. Generally, significant oxidation of manganese will not occur in the presence of ferrous iron.[14] Thus, manganese tends to remain in the solution phase of disposal area slurries, even when the effluents have a color and dissolved oxygen content suggesting highly oxidizing conditions.

Zinc and copper

212. Total constituents. Total zinc decreased by 96.5 percent during containment (Table 9 and Figure 14). Total zinc in influents averaged 27.5 mg/ℓ (323 mg/kg in dry solids) with a range of 0.6 to 206 mg/ℓ, while effluents averaged 1.2 mg/ℓ with a range from 0.026 to 5.49 mg/ℓ. Total copper showed a net decrease of 95 percent in an average effluent, with influents averaging 6.09 mg/ℓ (52.2 mg/kg in dry solids) and effluents averaging 0.30 mg/ℓ (range of 0.02 to 1.59 mg/ℓ).

213. Soluble phase constituents. Soluble zinc and copper showed some site-specific increases during the land containment of dredged material. Average soluble zinc concentrations increased from 0.055 to 0.064 mg/ℓ, whereas copper increased from 0.019 to 0.021 during land containment (Table 9). Only the average zinc value is considered to be outside the range of analytical error, although increases for specific sites seemed to be important for both zinc and copper (Figure 17).

Soluble zinc and copper were also more than twice as high in effluents than they were in the surface background waters (Table 9 and Figure 15). However, these variations were not significantly different (using the F-test) because of wide variances in the effluent ranges (Table 9).

214. Soluble zinc did not present a serious problem at the disposal areas included in this study, as most effluent concentrations were less than 0.1 mg/ℓ and were comparable to background levels (Table 8; Figures 15 and 16). Although the copper levels in effluent filtrates were usually above background levels, the concentrations were generally much less than 0.05 mg/ℓ (Figure 16).

215. Geochemical partitioning. The average geochemical phase partitioning data for four sites (Table 10 and Figure 19) show that over 10 percent of the zinc in influent solids was associated with carbonate complexes. Also, there was a major shift of the solid phase zinc into the carbonate phase during confined land disposal, with effluent solids showing a net increase to 33 percent. This phase shift was significant, using the F-test, as there were no net major shifts in any of the other phases analyzed. Copper did not seem to be bound appreciably in carbonate complexes of influent solids. However, there was a major shift into the carbonate phase during slurry retention in the disposal areas, increasing from 5 to 12 percent. About 11 percent of both the total zinc and copper was found in the easily reducible phase extracts of influents. However, both elements showed only small increases in this phase during containment, considering both the dry weight and percent values given in Table 10.

216. Most of the zinc and copper in influent solids (75 to 80 percent) was associated with the acid digest fraction (Table 10 and Figure 19), which includes the moderately reducible, organic-sulfide, and residual phases of other related studies.[17,132] The geochemical phase partitioning of sediment, influent, and effluent solids from the Seattle site includes the organic-sulfide phase (Appendix B, Table B19). These data show that over 55 percent of the total zinc and 75 percent of the total copper were in the organic-sulfide phase of influent solids, although the respective amount associated with organic matter or sulfides

can only be inferred. This study also showed major decreases of zinc and copper in the organic-sulfide phase of effluent solids. Thus, the slight decreases of zinc and copper in the final acid digest extractions of effluent solids (four sites) may be due to a decrease in the organic-sulfide geochemical phase. This assumption is of course based on limited data from a single site. However, the oxidation of the dredged material during confinement should result in decreases in both sulfides and solid phase organic compounds.[151] High acid digest values (65 to 75 percent) for zinc and copper in partitioned influent solids from Richmond were also noted (Appendix B, Table B19). Since total sulfides were low at this site, averaging 60 mg/kg, it appears that organic complexation of zinc and copper is an important mechanism for binding these metals in dredged material. Zinc silicates may also contribute to the total acid digest fraction, especially under oxidizing conditions.[151]

217. The geochemical phase partitioning data in Figure 19 indicate that zinc mobility in land containment areas is primarily controlled by moderately soluble solid phase carbonate complexes, and that zinc is not readily complexed with oxidized iron-manganese precipitates. This is also strongly suggested by geochemical phase partitioning data of effluent solids from the Seattle site, where the major solid phase was composed of oxidized iron precipitate. Zinc increased in the carbonate phase from 3.3 to 61.5 percent, while the easily reducible phase showed only a minor increase from 6.8 to 11 percent.

218. The geochemical phase partitioning data (Figure 19) suggest that copper may be regulated by several geochemical phases associated with dredged material solids. As previously discussed, both the carbonate and easily reducible phases seemed to regulate copper mobility. Geochemical phase partitioning of the iron precipitates collected at the Seattle site also showed an increase in the carbonate phase from 0.3 to 17.7 percent and a shift in the easily reducible solids from 0.5 to 20 percent during disposal area detention (Appendix B, Table B19).

219. Particle size fractionation. Figure 20 shows the partitioning of various metals in three different particle size fractions of effluent solids from the Seattle site. The solids were separated by

filtration into the greater than 8-, 0.45- to 8-, and less than 0.45-μm-sized particles. These data show that most of the solid phase zinc is associated with large (> 8-μm) suspended particles. Low-density material, such as organic matter, could be the major component of this larger particle size fraction.

220. The particulate fractionation data of effluent solids from Seattle (Figure 20) show a large proportion of solid phase copper in the less than 0.45-μm digests. As the soluble phase was not included and since the geochemical phase partitioning data show little exchangeable phase copper, a large amount of solid phase copper seemed to be associated with fine solids in mainly nonexchangeable form. Because of the observed association of copper with fine particles, it may be more difficult to remove than zinc from dredged material slurries during land containment.

221. Mechanisms governing zinc mobility. The effect of retention time on zinc mobility could not be conclusively shown. The large disposal area at Wilmington showed a net increase in soluble zinc, from 0.04 to 0.09 mg/ℓ, along with an increase in alkalinity (Tables 7 and 8). Residence time variability at Lake Charles induced no noticeable change, while increased residence time at the Seattle site, in conjunction with a decrease in alkalinity, seemed to promote the release of zinc, although much of this may have originated from the dike sediments (see Table 16).

222. Since zinc is an important micronutrient for both plants and microorganisms, its mobility may be governed to a great extent by biological uptake and release. The high soluble zinc in one of the effluent samples from the Wilmington site could have been promoted by the decomposition of the abundant dead vegetation in the disposal area, whereas actively growing vegetation in the Southport containment area may have resulted in the initially large decrease in soluble zinc. At the Lake Charles site, algae in the water may have counteracted the residence time effect by removing the zinc as a nutrient or by promoting zinc precipitation in the carbonate phase, induced by algal photosynthesis.

223. It appears that soluble zinc removal may result from biological uptake and precipitation as carbonate complexes, while the degradation of organic matter and the solubilization of carbonate complexes, especially in acidic environments, may promote zinc release. The pH of the sediments in the disposal areas seemed to correlate better than effluent pH with zinc release; most of the disposal area effluents showed slight increases in pH (Tables 6 and 12), which should promote the immobilization of zinc as carbonate precipitates. Very fine precipitates which form in the effluent water could pass through a 0.45-μm membrane filter. In general, increased detention time appeared to promote high soluble zinc concentrations, but only on a site-specific basis, as shown by Figures 16 and 17. High zinc levels could be expected mainly in areas with high levels of readily decomposable organic material.

224. Mechanisms governing copper mobility. Data presented in Table 8 and Figure 17 show a close similarity between influent and effluent filterable copper at most sites. Exceptions to this are samples from the Wilmington and Lake Charles areas, which showed moderate increases in soluble phase copper. The net increase in the easily reducible phase copper in effluent solids suggests that copper becomes associated with iron-manganese precipitates under oxidizing conditions. A related study,[151] involving the geochemical phase partitioning of marine sediments, also shows an increase in reducible phase copper with increased oxidation of resuspended solids. The literature suggests that reducible copper is mainly associated with hydrous oxides of manganese.[18]

225. Perhaps the greater release of soluble copper at Wilmington may be related to the greater interaction of the slurry with moderately reduced, slightly acidic sediments in the disposal area, which would promote the dissolution of copper from organic-sulfide complexes and inhibit the formation of easily reducible copper complexes. Similar to Wilmington, soluble iron and manganese were also high in effluents from the Lake Charles site. Thus, complexation of copper with the easily reducible solid phase may have been inhibited at both of these

sites. The sediments dredged at the Lake Charles site consisted of a mixture of highly reduced dark grey mud with masses of fine sand and silt showing evidence of iron oxidation. This mixed Eh environment may have helped to promote a rapid release of copper in the disposal area. This study cannot substantiate organic complexation as being important. However, the association of copper with organic matter is well documented.[10,151] Copper appeared to be relatively stable in most of the dredged material solids during the usually short retention periods. The findings of this study suggest that high copper mobility should rarely occur in land containment areas.

Cadmium and lead

226. Total constituents. Cadmium and lead seemed to show geochemical properties analogous to those of zinc and copper. However, in contrast to zinc and copper, which are micronutrients, cadmium and lead are not essential for the growth of living organisms. Average total cadmium in influents from nine sites was 1.39 mg/ℓ (range: 0.002 to 7.17 mg/ℓ), while average effluents contained 0.055 mg/ℓ (range: < 0.002 to 0.37 mg/ℓ). The removal efficiency was thus about 96 percent. The solid phase showed a general decrease during land confinement (Tables 8 and 9) although the total for the geochemical phase partitioning data showed a slight increase in the solid phase of effluents from four sites (Table 10). The range in solid phase cadmium in influents was from 0.05 to 45.3 mg/kg (Table 9).

227. Total lead averaged 16.2 mg/ℓ (range: 0.24 to 86.5 mg/ℓ) in influents and 1.07 mg/ℓ (range: 0.001 to 7.6 mg/ℓ) in effluent samples, resulting in a net removal of over 93 percent in land containment areas. Lead also showed an overall decline in effluent solids from all sites but a small increase in the partitioned effluent solids. Solid phase lead ranged from 1 mg/kg in an effluent sample to 327 mg/kg in an influent sample.

228. Soluble phase constituents. Soluble phase cadmium showed no significant overall change in concentration in dredged material slurries during confined disposal area retention, averaging 0.004 to 0.003 mg/ℓ in influents and effluents, respectively. The Wilmington site showed

the greatest change in soluble cadmium, with effluents showing about a three fold decrease (Table 9). Moderately reducing conditions, high carbonate alkalinity, and high organic content of the sediments are several environmental factors possibly contributing to this decrease. Average data for the individual disposal areas (Table 8) show little change in soluble cadmium during slurry detention. Surface background water usually showed very similar soluble cadmium concentrations, with a high value of 0.01 mg/ℓ present in the Atlantic Intracoastal Waterway at Southport.

229. The soluble lead data (Table 9 and Figures 14, 15, and 17) showed no change in either the influent, effluent, or background water concentrations, with only trace quantities being detected at most sites. The average was 0.002 mg/ℓ with an influent sample from Wilmington showing the maximum value of 0.012 mg/ℓ.

230. Geochemical partitioning. The geochemical phase partitioning of influent and effluent solids (Table 10 and Figure 19) shows that cadmium increased in the carbonate phase during land containment from an influent average of 21.4 percent to an effluent average of 56.7 percent. Also, cadmium showed a noticeable increase in the easily reducible phase from 9.2 to 11.8 percent. The source for these increases was a decrease in the acid digest fraction, from 49.3 to 13.5 percent; geochemical phase partitioning data from the Seattle site (Appendix B, Table B19) infer that the organic-sulfide phase was most important in this decrease and also that most of the cadmium in the influent solids was associated with the organic-sulfide complexes. In contrast, effluent solids often showed more cadmium in association with the carbonate and easily reducible phases than in the organic-sulfide phase. Cadmium also increased noticeably in the exchangeable phase during dredged material containment, with effluent solids binding about 20 percent of the cadmium in exchangeable form. Similar findings have been noted for cadmium in dredged material under oxidizing conditions, especially at slightly acidic pH.[18] However, data from related research[132] failed to show significant cadmium in either the exchangeable or easily reducible phases of reduced sediments.

231. Geochemical phase partitioning data (Table 10 and Figure 19) show that lead increased appreciably in the carbonate phase of effluent solids; influent solids contained only 0.2 percent, while effluent solids increased to 11.4 percent. Easily reducible phase lead was comparable but relatively high in both influent and effluent solids, averaging about 18 percent of the total. Exchangeable lead was a smaller, although important component, with no notable change during slurry retention. Under oxidizing conditions, the organic-sulfide phase appeared to decline, as would be expected. Partitioning data from Seattle (Appendix B, Table B19) suggest that lead also increases slightly in the moderately reducible-residual phase of effluent solids.

232. Mechanisms governing cadmium mobility. The partitioning data from this study indicate that the retention of dredged material under oxidizing conditions should shift cadmium from less available phases (e.g., organic-sulfide) to more readily bioavailable phases, including the easily reducible, carbonate, and exchangeable phases. The exchangeable phase is considered to be readily available to living organisms, while carbonate and easily reducible complexes can be rapidly modified under changing pH and Eh conditions. The partitioning data for Seattle show only a negligible increase in exchangeable phase cadmium. As these solids were mainly iron precipitates, it appears that iron hydrous oxides are not involved appreciably in the ion exchange reactions at slightly alkaline pH.

233. Although oxidizing conditions in the land containment areas seemed to shift cadmium to potentially more available or mobile solid phases, the cadmium appeared to remain with the effluent solids. Thus, total effluent analysis may provide the best estimate of the cadmium impact by discharges from confined disposal areas. This study indicates that total cadmium may become more available to living organisms as a result of the land containment of dredged material. Further research is needed to assess the fate of the cadmium associated with the effluent solids after it reaches the receiving water.

234. Mechanisms governing lead mobility. The reason for the low lead solubility may relate to the low solubility of most inorganic lead

complexes. Although the binding of soluble organic compounds with lead can greatly enhance its mobility, a related study[18] indicated that the stability of lead-organic complexes may be reduced under oxidizing conditions. In other research with dredged sediments,[17] the greatest lead release from marine sediments occurred under oxidizing conditions. Nevertheless, the results of this research as well as the studies mentioned above indicate that lead should not create a serious problem during most dredging and disposal operations. However, similar to cadmium, lead associated with effluent solids may be in forms which are possibly more available to the biota.

Nickel, vanadium, chromium, and titanium

235. Nickel and vanadium are discussed together since they are the most abundant metals in petroleum, being especially concentrated in the asphaltic fraction.[82] Nickel and chromium, according to the literature[151] and the geochemical phase partitioning data from this study (Figure 19), display slow phase changes when subjected to changing environmental conditions. Also, chromium and titanium generally showed little change in their soluble phase concentrations during land containment of dredged material (Figures 14 and 17).

236. Total constituents. Total nickel decreased by 94.5 percent in effluents from nine disposal areas, with an average influent concentration of 5.8 mg/ℓ (range: 0.21 to 18.2 mg/ℓ) and an effluent value of 0.32 mg/ℓ (range: < 0.01 to 1.7 mg/ℓ). The solids showed no significant change during containment, with a net value of 47 mg/kg (Table 9).

237. Data for vanadium are based on only three sites; namely, Sayreville, Houston, and Grand Haven (see Table 8). These show a 93 percent removal of total vanadium, which averaged 3.5 mg/ℓ in influents.

238. Chromium data were also somewhat limited, with only samples from Sayreville, Grand Haven, and Seattle being analyzed (Table 8). The only total influent samples measured were from Grand Haven, which showed a net total chromium value of 63.8 mg/ℓ. Influent samples were not analyzed for chromium at the Seattle site. However, the dredged sediment contained a dry weight value of 52 mg/kg, which indicates that average total influents contained about 3.5 mg/ℓ chromium, based on

average influent total solids. The average total effluent concentration for three sites was 0.21 mg/ℓ, with the highest level being 0.58 mg/ℓ in a high solids effluent from Grand Haven (Table 9). The Sayreville effluent sample contained 0.34 mg/ℓ of total chromium.

239. Total titanium in influent samples was measured only for the Houston site. The treatment efficiency for total titanium in the Houston disposal area was 97 percent (Figure 16 and Table 8).

240. Soluble phase constituents. Soluble phase nickel in influents and effluents averaged 0.014 and 0.012 mg/ℓ, respectively, with most sites showing comparable release (Tables 8 and 9). Nickel sulfides have been considered to be the solubility-controlling solid in highly reduced marine sediments.[17] Nickel sulfides are also among the most soluble of the trace metal sulfides.[55,102] This may account for the high soluble nickel (0.02 to 0.035 mg/ℓ) in background water and dredged slurries at the Seattle and Southport sites, since the highly reduced sediments at these locations contained the highest levels of total sulfides. However, there is no conclusive evidence to substantiate this observation.

241. Soluble vanadium decreased by about 17 percent, with influent and effluent averages of 0.018 and 0.015 mg/ℓ, respectively (Table 9). There was no good relationship between oil and grease values and either nickel or vanadium concentrations in total or filtered samples. For example, the Houston site showed the highest oil and grease values, mainly derived from petroleum, while only average nickel and vanadium concentrations were noted. Thus, on a quantitative basis, petroleum does not seem to be an important source for these trace metals.

242. Soluble chromium in two comparable influents and effluents averaged 0.0036 and 0.0044 mg/ℓ, respectively. From these data, it appears that chromium does not readily undergo transformation in dredged material slurries during land containment. The individual data points also indicate this. The highest soluble phase chromium values were measured at the Seattle disposal area, with average and maximum effluent concentrations of 0.025 and 0.033 mg/ℓ, respectively (Figure 16 and Appendix B, Table B16).

243. Titanium is often dispersed, through atmospheric transport, as a very fine titanium oxide dust. This was particularly evident at the Sayreville dredging and disposal areas, which are adjacent to a titanium oxide plant. The high titanium concentrations in influent, effluent, and background water filtrates from Sayreville undoubtedly result from passage of these highly insoluble particulates through the 0.45-μm membrane filters (see Table 8). Houston was the only other site where titanium was measured in slurry samples. Filterable titanium at Houston averaged about 0.03 mg/ℓ in both influents and effluents, although the background water (collected about 4 miles from the dredging site) contained only 0.0001 mg/ℓ soluble phase titanium. Filterable (soluble) titanium showed no noticeable decrease in concentration during residency in both the Sayreville and Houston disposal areas, with average influent and effluent concentrations of 0.122 and 0.132 mg/ℓ, respectively.

244. Geochemical partitioning. Geochemical phase partitioning data showed approximately 5 percent of the solid phase nickel in the easily reducible and exchangeable phase extracts, respectively, with no important change in these phases during retention of the slurry in the disposal areas. Nickel declined appreciably in easily reducible phase extracts of effluent solids from Wilmington and Richmond. However, the hydroxylamine extractant is not totally selective for iron-manganese complexes. About 85 percent of the nickel remained in the final acid digest. The geochemical phase partitioning data from Seattle (Appendix B, Table B19) shows about 60 percent of the nickel in the moderately reducible-residual phases, and related research[132] has shown nickel to be dominant in the residual phase, representing mainly the fraction in the crystalline structure of mineral particles. About 25 percent of the nickel was found in the organic-sulfide phase of influent solids from the Seattle site, with effluent solids showing a small decrease to 15 percent.

245. Geochemical phase partitioning data for chromium were obtained only for the Seattle site. Based on the results, about 95 percent of the solid phase chromium was equally divided between the

organic-sulfide and moderately reducible-residual phases, with no observable difference between the dry weight concentrations in the influent and effluent suspended particulates. Although there was a small decline in easily reducible phase chromium in effluent solids, this may not relate directly with its decrease in reducible iron-manganese precipitates, as was previously mentioned.

246. There were no partitioning data for vanadium or titanium.

247. Particle size fractionation. The particle size fractionation of effluent solids from Seattle (Figure 20) indicated that both nickel and vanadium are quite evenly dispersed among the different particulate fractions, but with a greater portion in the less than 0.45-μm fraction in comparison to the bulk of the iron and manganese precipitate, which dominated the solid phase.

248. The particulate fractionation of effluent solids from Seattle (Figure 20) shows that chromium was associated mostly with the less than 0.45-μm-sized particulate fraction. The increased surface area of the smaller particles should promote increased chromium adsorption at exchange sites on the colloid surfaces. However, there was only a very small quantity of exchangeable chromium associated with these same effluent solids. Therefore, much of the particulate chromium probably exists as a very fine chemical precipitate. The decline in the easily reducible phase chromium suggests that coprecipitation with iron-manganese precipitates may not be important. The particle size fractions were not analyzed for titanium.

249. Mechanisms governing mobility. In summary, soluble phase nickel, vanadium, chromium, and titanium showed negligible change during land containment of dredged material (Figures 14 and 17). High soluble nickel values, which were noted in influent and effluent samples from the Seattle and Southport disposal areas, were also observed in the respective background water samples (Table 8). It thus seems that these high concentrations originated from the bottom water at the dredging site and were not appreciably changed by short-term slurry retention in land confinements. The high background levels were thought to occur mainly through long-term equilibration of dredging site water with nickel

sulfides in the highly reduced sediments.

250. The overall results of this study strongly indicate that solid phase chromium should not be readily mobilized during the retention of dredged slurries in disposal areas. Other research[17,55] also substantiates this conclusion, noting little chromium release from dispersed sediments under oxidizing conditions. The slow oxidation of reduced chromium hydroxide ($Cr(OH)_3$) precipitates has been suggested as a possible reason for its slow mobilization.[55] The high chromium levels at Seattle are unusual since reduced chromium (Cr III) is generally highly insoluble at pH values above 5.5, unless complexed with soluble organic compounds; chromium has also not been noted to oxidize to more soluble chromium (VI) forms under short-term oxidizing conditions.[152] One possible explanation, suggested by the particulate fractionation data (Figure 20), is that much of the 0.45-μm filterable chromium actually exists in very small particulate matter, probably as hydroxides of chromium III. Quantitatively, soluble organic carbon showed no direct relationship with soluble phase chromium.

251. High levels of filterable titanium are thought to mainly represent very fine aerosol particles, as most forms of titanium are very insoluble. Vanadium was also found to show little change during retention in confined disposal areas, but further data needs to be collected concerning this poorly studied element. Data concerning the complexing of vanadium, nickel, and other trace metals with oil and grease extracts from confined disposal areas are given in another WES Technical Report.[145]

Mercury and arsenic

252. Mercury and arsenic show a combination of both similar and contrasting properties. Both elements can be microbially methylated, forming highly toxic products.[153] Mercury can become highly volatile if reduced to its elemental state (Hg^o) and moderately volatile if transformed to dimethyl mercury; arsenic can be volatilized if reduced to arsine (AsH_3), or di- and trimethyl arsine.[10,153] Although methylated mercury and arsenic derivatives can form under both aerobic and anaerobic conditions, they persist for a longer period of time in

aerobic environments.[153,154] The synthesis of methyl mercury compounds is inhibited by the presence of sulfides, and thus aerobic conditions may be more conducive to methylation reactions in estuarine and marine environments.[14] The adsorption of mercury by solids also seems to be inhibited by increased oxidation and decreased pH in dispersed sediments.[18] Mercury forms some of the most poorly soluble sulfide complexes, while arsenic forms very soluble sulfides. However, polysulfides of both mercury and arsenic, which tend to form at high free hydrogen sulfide concentrations, are somewhat more soluble.[10,102] The solubility of mercury may thus tend to increase under conditions of greater oxidation and acidic pH as these favor the decomposition of sulfides and organic matter. However, highly reduced sediments could release some soluble mercury if conditions are right for polysulfide formation (e.g., low iron levels in the sediments). Arsenic appears to be most readily released from sediments under reduced conditions. Reasons for this could be the observed close association of arsenic with iron oxide and iron phosphate precipitates[155] (easily and moderately reducible phases), and the soluble nature of the arsenic sulfide complexes. Reduced arsenic compounds also appear to be most toxic.[10]

253. Total constituents. The average total mercury in influent, effluent, and background water samples from all of the monitored sites was 0.044, 0.024, and 0.001 mg/ℓ, respectively. Thus, land containment resulted in only a 45.5 percent decrease of total mercury. Although accurate solid phase calculations are subject to considerable error when dealing with low concentrations, it appeared that there was an increase of mercury in effluent solids, from 0.46 to 0.79 mg/kg (Table 9); this increase probably reflects the anomalous removal efficiency.

254. Total arsenic decreased by about 90 percent, to an average effluent value of 0.16 mg/ℓ (Table 9 and Figure 14). The highest values for total arsenic came from Grand Haven, with maximum influent and effluent concentration of 6.0 and 0.41 mg/ℓ, respectively. The lowest values came from Seattle effluent samples, averaging 0.009 mg/ℓ; these samples were also the lowest in total solids.

255. Soluble phase constituents. Soluble mercury, which was measured at seven disposal areas, generally showed only trace quantities (< 0.0002 mg/ℓ) using the cold vapor analytical technique, while the values obtained using the Zeeman spectrophotometer were considerably higher (see Figure 16). This variance is thought to be due to the breakdown of very fine (< 0.45-μm) filterable particles in the high temperature of the Zeeman instrument. The samples from a given disposal area were usually all run by either the Zeeman or cold vapor methods. The results show that mercury was at very low and comparable levels in influent, effluent, and background water filtrates, with average values for each of 0.001 mg/ℓ (Table 9). The Wilmington site showed the highest filterable mercury concentrations, with influent, effluent, and background water samples giving values of 0.008, 0.006, and 0.004 mg/ℓ, respectively, using the Zeeman instrument. The Sayreville site gave the highest values using the cold vapor technique, with influents showing 0.0009 mg/ℓ while effluent and background water samples each contained 0.0007 mg/ℓ (Table 8). The overall results strongly indicate that mercury does not experience increased mobility from solids in dredged material as a result of land containment. At the Wilmington and Southport sites, small decreases were observed in soluble phase mercury.

256. Soluble phase arsenic decreased by about 90 percent in effluent samples from the four monitored confined disposal areas; namely Sayreville, Houston, Grand Haven, and Seattle (Table 9 and Figure 14). The Seattle site showed the highest soluble phase arsenic, with influent and effluent filtrates averaging 0.064 and 0.006 mg/ℓ, respectively. Since values for effluents from the first pond at Seattle averaged 0.012 mg/ℓ, it appears that increased residence time under oxidizing conditions may aid in the removal of soluble phase arsenic. The other three sites showed average influent and effluent arsenic values of about 0.0004 and 0.0002 mg/ℓ, respectively (Table 8), which indicate that arsenic removal efficiency may be dependent on concentration.

257. Geochemical partitioning. The geochemical phase partitioning data, as shown in Figure 19, indicate that most of the mercury

resided in the less readily available phases; namely, the organic-sulfide, moderately reducible, and residual phases. These phases are represented by the final acid digest fraction. There was a small decrease in the acid digest phase, from 99.4 to 94.7 percent, during the land containment of dredged material. With this decrease, Figure 19 shows a small increase in the easily reducible phase mercury in effluent solids; however, this shift was only observed in the effluent solids from Wilmington (Appendix B, Table B13), where about a 17 percent increase was noted. The increase in exchangeable mercury was also site specific, and the average value represents an increase in one effluent solids extraction from Lake Charles in conjunction with a small increase in an influent solids sample from Wilmington.

258. Partitioning data for arsenic include only influent and effluent solids from Seattle. The results (Table 10 and Figure 19) show that 98.7 percent of the influent and 85.4 percent of the effluent solid phase arsenic are associated with the less soluble phases (e.g., organic-sulfide, moderately reducible, residual). The data in Appendix B, Table B19 indicate that about a third of the arsenic in the solids was associated with the organic-sulfide fraction, while the remainder resided mainly with the moderately reducible and residual phases. It is interesting to note that the second extraction scheme for the organic-sulfide fraction elicited a much greater recovery of arsenic. The methods used are given in Part III under the geochemical phase partitioning analysis section. Since the first method appears to be a harsher treatment, this finding is difficult to explain, unless the arsenic was volatilized during the first extraction scheme.

259. The exchangeable phase extracts showed the only other measurable levels of arsenic, with 1.3 percent in the influent sample increasing to 14.6 percent in effluent solids. Surprisingly, the easily reducible phase showed no measurable arsenic, despite evidence that arsenic has an affinity for the oxidized precipitates of iron and manganese;[10,155] these precipitates constituted the major part of the effluent solids at the Seattle site. However, the preferential sorption of arsenic by amorphous iron and aluminum hydroxides is also well

documented,[10] which might account for the exchangeable phase increase. Iron-manganese oxide complexes are supposed to be predominantly positively charged at pH 7.7, which was the average effluent pH value at the Seattle site.[156] Since arsenic is mainly in anionic form (usually as AsO_3^{-3} or AsO_2^-), the positively charged iron colloids should be important in exchange reactions.

260. Particle size fractionation. The particulate fractionation data for Seattle (Figure 20) indicate that most of the arsenic was associated with the larger (> 8-μm) particulate fraction. Since exchangeable arsenic would tend to predominate in the finer fraction, which would have a greater unit surface area, the larger particulates could represent mainly organic or moderately reducible phase arsenic; the residual fraction was thought to be of minor importance in the Seattle effluents because of an extensive residence time and use of a flocculating agent.[26] Mercury was not determined for the different particle size fractions.

261. Mechanisms governing mobility. The data indicate that mercury is very stable in most dredged material and that mobilization into the aqueous phase or into more available geochemical phases does not appear to readily occur during the short residence times incurred in confined disposal areas. The strong binding of mercury with sulfides and the low oxidation rate of mercuric sulfide[10,55] seem to be plausible mechanisms for the above findings.

262. The poor removal efficiency of total mercury (Figure 14) and the small increase in solid phase mercury in effluents suggest that mercury is mainly associated with the finer particulate fractions. Thus, total mercury removal from dredged material during land containment may be only moderately successful. Although the solid phase mercury does not appear to be in readily available chemical forms, ingestion of the small particulates by organisms could result in the release of this mercury. Thus, if total mercury seems to be a problem, extra effort should be directed toward the removal of the fine particulate matter.

263. The arsenic data suggest that arsenic can be mobilized in appreciable quantities during the dredging of highly contaminated sediments. However, the solid-phase chemical form that the arsenic is in also seems to be important, as total arsenic appeared to be equivalent or higher in sediments from Grand Haven. The Grand Haven sediments were very sandy while the Seattle sediments consisted of highly reduced black mud, high in total sulfides. The data thus agree with previous findings. Reduced environments favor the mobilization of arsenic, while oxidizing environments favor arsenic immobilization. As most land containment areas promote oxidizing conditions, this disposal mode should be most favorable for the disposal of dredged material containing high levels of arsenic.

Geochemical Phase Partitioning of Influent and Effluent Solids

General information

264. Data presented in Table 10, Figure 19, and Appendix B, Table B19 show the results of the geochemical partitioning of 14 different elements associated with the solid phase of five influent and effluent samples from four different confined disposal areas. Locations of disposal areas were: Wilmington (Eagle Island), N. C.; Richmond (Deepwater Terminal), Va.; Lake Charles (Calcasieu River, disposal area No. 22), La. (2 samples); and Seattle (slip 1, Duwamish Waterway), Wash.

265. Geochemical partitioning phases were as follows: exchangeable carbonate, easily reducible, organic-sulfide (Seattle), and total acid digest.

266. Each of the above geochemical phases was removed by subjecting the same sediment samples to different, generally harsher chemical treatments. A description of each chemical extractant and the extraction procedures is given in Part III. The exchangeable phase includes mainly the fraction of each metal which is physically adsorbed at the charged surfaces of solid particles; the carbonate phase selects primarily the metal carbonate complexes; metals associated with manganese

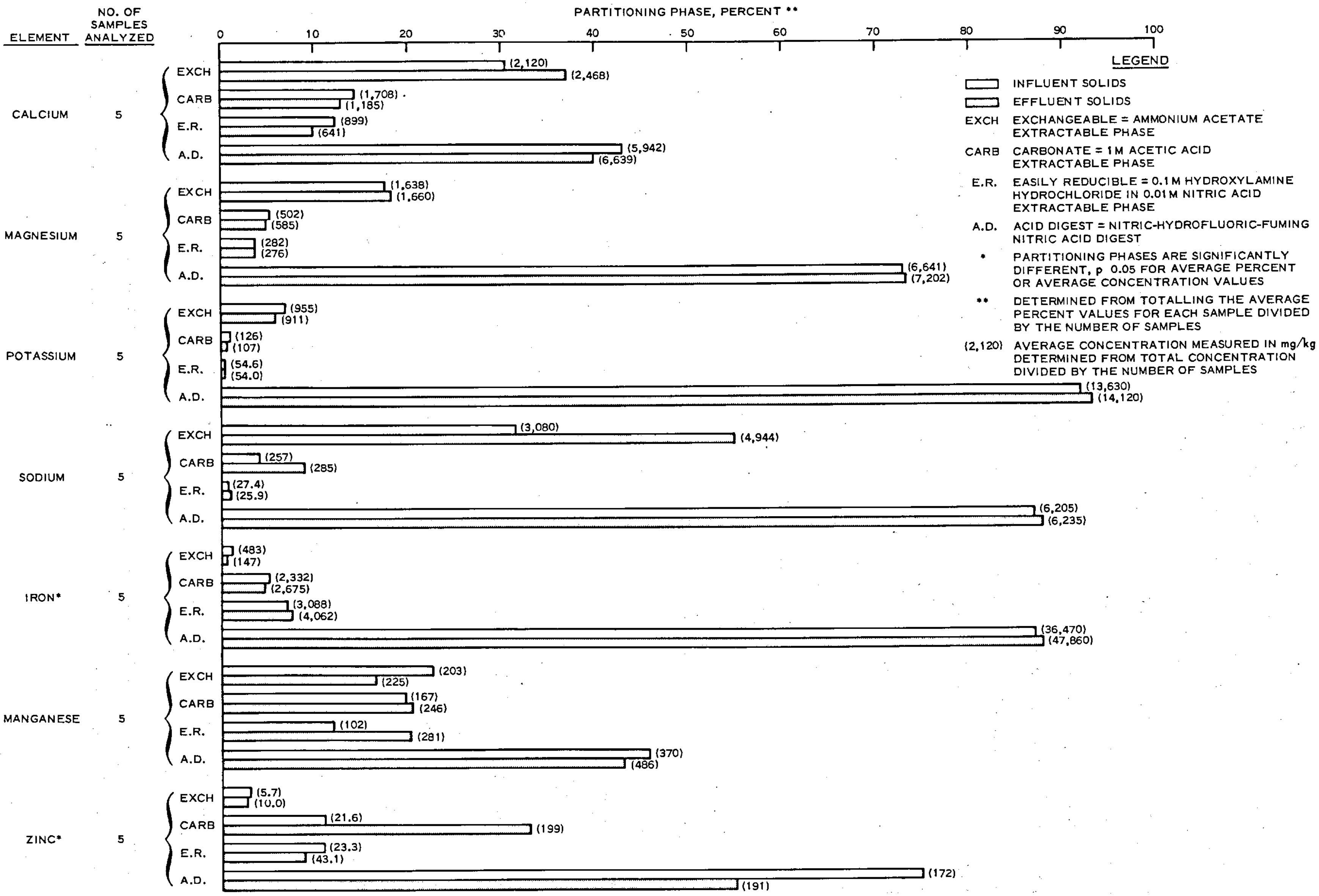

Figure 19. Geochemical phase partitioning of metals in influent and effluent solids from four confined land disposal areas (sheet 1 of 2)

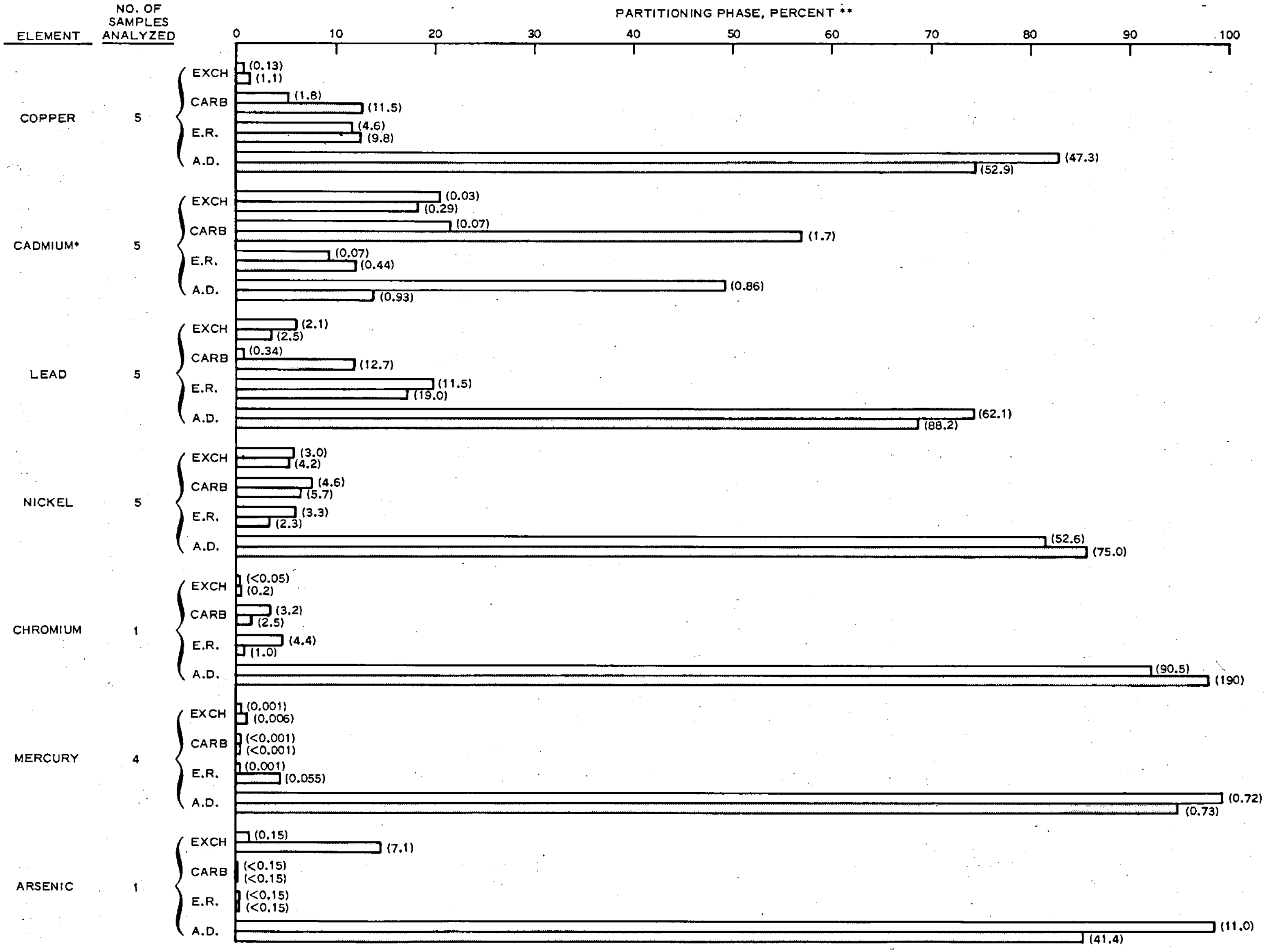

Figure 19 (sheet 2 of 2)

and amorphous iron hydroxide and hydrous oxide precipitates are released into the easily reducible phase extracts. Sediments, influent solids, and effluent particulates from the Seattle site were also chemically treated to mobilize metals associated with most of the solids-bound organic matter and sulfide complexes. However, the extractants used in removing the above physically- and chemically-bound metals are not completely selective for a given phase. For example, the carbonate phase extraction could include some amorphous iron and manganese precipitates.[17] The final treatment was a hot acid digest, which would remove mainly the metals which are only very slowly mobilized in the natural environment. This would include the highly insoluble sulfides and much of the tightly bound organic matter (except for Seattle samples), crystalline iron oxide coatings on mineral particles, and elements bound in the crystalline structures of various primary and clay mineral particles.

267. Eleven elements were measured in influent and effluent solids from the five sites; namely, calcium, magnesium, potassium, sodium, iron, manganese, zinc, copper, cadmium, lead, and nickel. Mercury was not measured in solids from Seattle, while cadmium and arsenic were measured only in Seattle samples.

268. Contrasting environments were included in this study. The Richmond site was in a freshwater regime; Wilmington showed only slightly brackish conditions (3 $^{o}/oo$ salinity); Lake Charles and Seattle showed influent salinities of about 20 and 30 $^{o}/oo$, respectively. Seattle sediments consisted of highly reduced black sulfide mud; Lake Charles showed mixed sediments of varying texture and Eh status; Richmond and Wilmington solids samples represented reduced sediments from quiescent harbor areas, containing only moderate sulfide levels (30 to 60 mg/kg dry weight). The Wilmington site contained thick dead or dormant vegetation, mainly grasses; the Richmond disposal area had a moderate cover of forest vegetation; the Lake Charles site contained abundant planktonic algae in the surface ponded water. The Seattle disposal site contained no observable vegetation or algae; however, effluent turbidity in the final settling pond was observed to be

predominantly composed of oxidized iron precipitate. The disposal areas ranged in size from the 1.9-acre double ponded disposal area at Seattle to the 400-acre overland flow system at Wilmington. Additional site characteristics are listed in Table 1.

Total metals

269. Total acid digests and summations of the geochemical phase partitioning data show that there was an overall increase of most elements in effluent solids. This was true for all of the metals analyzed in the partitioned solids (Figure 19), with cadmium and arsenic showing greater than three fold increases while total zinc and chromium doubled in concentration. Iron, manganese, lead, copper, and nickel all increase about 1-½ times in the effluent solids. Elements showing only a small accumulation in the effluent particles include calcium, magnesium, potassium, and mercury. Small increases of metals in effluent solids seem plausible since settling of the heavier particles should favor the removal of silica and aluminum oxides, which tend to dilute the heavy metal precipitates that may be forming in the solution phase. Also, the adsorptive capacity of the solids should increase with decreasing particle size and increasing organic matter content; organic detritus would tend to remain in suspension because of its low density. However, the sources for the increases appeared to be highly variable for each element.

Exchangeable phase

270. The exchangeable phase concentrations of metals are generally thought to be governed by dynamic equilibria with their soluble phase concentrations.[132] However, the percent adsorption is governed by other factors, including complexation, ion charge, hydrated ion radius, and pH.[11] About a third of the solids-bound calcium and sodium were removed during exchangeable phase extractions. Calcium is known to compete well for exchange sites, while the high soluble sodium concentrations in most of the samples promoted the sorption of sodium to the particulate fraction. Manganese, magnesium, and cadmium in the influent and effluent solids showed average exchangeable phase levels of greater than 10 percent. In the partitioned samples, calcium,

sodium, copper, and arsenic also showed measurable increases in the exchangeable phase of effluent solids. The increase in exchangeable calcium in effluent solids is partly explained by the net increase in soluble phase calcium in the same effluents. Only iron and potassium showed decreases in the exchangeable phase of effluent solids. The low iron value could be promoted by the rapid chemical precipitation of sorbed ions; potassium, which seems to be associated with a fine-particulate fraction at the Seattle site (Figure 20), may also be undergoing rapid chemical binding of some unknown nature. The remaining elements showed no significant change in the exchangeable phase during detention of the suspended particulates.

Carbonate phase

271. The majority of the elements showed increases in their carbonate phase concentrations as a result of confined disposal, with higher levels of zinc, cadmium, manganese, lead, copper, and sodium being eluted from the effluent solids. Influent solid phase cadmium and manganese each showed high values of 20 percent, while carbonate phase cadmium, zinc, and manganese contributed to the effluent solids by 57, 33, and 20 percent, respectively. Although lead and copper were not very high in the carbonate phase of influent solids, their values increased considerably in effluent particulates. Calcium and chromium showed small decreases. Much of the soluble phase calcium increase observed in the effluent samples probably originated from dissolution of calcium carbonate in the slightly acidic sediments of some disposal areas (Tables 9 and 12). Iron, magnesium, potassium, mercury, and arsenic showed low concentrations and little change in the carbonate phase of the solids.

Easily reducible phase

272. Iron, manganese, cadmium, and copper increased in the easily reducible phase of effluent solids; effluent particulates showed net decreases in calcium, nickel, and chromium, while the remaining elements showed no definite change or uncertain, site-specific trends. The increase in solid phase iron and manganese during containment is the result of a general decrease in their soluble phase concentrations

under oxidizing conditions; the easily reducible phase results confirm that iron and manganese hydroxide formation is important in the removal of these metals in land containment areas. These hydroxide precipitates, in turn, act to scavenge other metal species. However, the exchangeable and easily reducible phase data from Seattle suggest that these precipitates may have had only minimal importance in sorbing other metals from the slightly alkaline pH water at this site. Some metals may predominate as neutral or anionic complexes, especially in the marine environment,[151] and the charge present on a soluble complex or precipitate is highly dependent on pH and Eh conditions.[156] For most elements, the easily reducible phase contained less than 10 percent of the total solid phase concentrations. Calcium, manganese, zinc, lead, and copper were present in the 10 to 20 percent range, with manganese increasing in effluent solids to over 20 percent. As previously mentioned, only the very fine amorphous iron precipitates are included in this mildly reducing extraction, while manganese hydroxides and hydrous oxides are specifically attacked.

Organic-sulfide phase and acid digests

273. The fraction of each element remaining in the extracted solids, namely, after removal of the exchangeable, carbonate, and easily reducible phases (also the organic-sulfide phase for Seattle samples), was obtained after complete dissolution of the solids by hot (nitric, hydrofluoric, fuming nitric) acid digestion. Iron, nickel, and chromium increased in the acid digests. The additional organic-sulfide extraction data from the Seattle site suggest that these increases resulted from several factors; namely, a stable organic-sulfide phase for iron and chromium, notable increases in the moderately reducible and residual fractions for iron, chromium, and nickel, and overall increases in total metals of the effluent solids. The iron increase in the moderately reducible fraction would be expected as this phase represents mainly aged (crystalline) iron precipitates. It is of interest to note that there were major decreases for most metals in the organic-sulfide phase of the effluent solids from Seattle, with only iron, chromium, sodium, and arsenic showing no important change. The

small change in organic-sulfide phase iron may indicate the presence of significant quantities of slowly oxidizable iron pyrites in sediments at the Seattle dredging location. Zinc, cadmium, mercury, and sodium showed decreases in the acid digests of effluent solids. The zinc decrease appeared to be caused by moderate declines in both the organic-sulfide and moderately reducible-residual fractions. Cadmium showed a major decrease in the organic-sulfide phase. The small decrease in sodium was noted to result from negligible decreases in the organic-sulfide and moderately reducible-residual fractions. In contrast, potassium showed a moderate increase to over 90 percent in the moderately reducible-residual fraction, indicating that potassium is mainly associated with very small diameter clay particles; sodium clays seemed to be of minor importance at most of the monitored sites.

Summary

274. Findings and conclusions from the partitioning data can be summed up as follows. The general decreasing trend for the organic-sulfide phase in effluent solids suggests that appreciable oxidation of metal sulfides and/or organic complexes occurs in disposal areas during slurry detention. The reductions in metal concentrations associated with sulfides and organic complexes generally result in minor increases of most metals in the moderately reducible phase. Most metals, except calcium, nickel, and chromium, showed small increases in the easily reducible phase; manganese exhibited the only major increase, resulting from the selectivity of the extractant for oxidized manganese precipitates. The easily reducible phase increases suggest that most trace metals can chemically bind with manganese (and amorphous iron) oxide precipitates and that metal scavenging in land containment areas by these precipitates can help to deplete the soluble phase concentrations. However, by far the majority of most metals tended to precipitate as carbonate complexes. This was especially true for cadmium, zinc, lead, and copper; manganese was important in the carbonate fraction but only a minor increase was noted as a result of slurry retention. The exchangeable phase, considered to represent the most available solid phase fraction, also was important for a few elements. Exchangeable

phase calcium and magnesium were high in influent solids, and major increases in these elements were noted in the effluent extracts. Copper and arsenic also showed important increases in the exchangeable phase of effluent solids. At this time, the relationship of arsenic with the exchangeable phase is unclear. Iron and potassium appear to decrease in the exchangeable phase, possibly as a consequence of very rapid chemical bonding induced by oxidizing conditions in the disposal areas. Amorphous oxidized precipitates of iron and manganese seem to play a minor role in exchange reactions for most metals in slightly alkaline disposal area surface waters.

Site-Specific Studies

Particle size fractionation of effluent solids - Seattle site

275. Site description. The land containment area at Seattle, Wash., was equally divided into two ponds. Although the total area was only 1.9 acres, with a total capacity of about 48,000 yd^3, the low influent flow rate, combined with the addition of a flocculent, resulted in very good retention of solids within the first pond. Visible turbidity was noticed at the weir discharge into the final pond on only a few occasions, during continuous influent discharge. Usually the pond 1 effluent showed very low turbidity.* However, water within pond 2 contained a fine orangish colloidal suspension which appeared to be caused by the precipitation of mainly iron hydroxides and hydrous oxides. The larger particles were gradually settling to the bottom of the pond, but continual formation of new particles was apparently taking place. The source for the iron was probably mainly from the dredged material, which contained about 5 percent total iron, combined mostly with sulfides; soluble phase iron entering pond 2 averaged 0.2 mg/ℓ. However, additional iron may have been released from the sandy loam sediments comprising the dikes. Iron hydroxide and hydrous oxide precipitates

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* Personal communication, J. M. Blazevich, EPA Region X Laboratory, Seattle, Wash., April 1977.

are known to be very efficient scavengers of soluble trace metals.[14,91] Therefore, an attempt was made to recover the suspended particulates for size fractionation and elemental analysis.

276. General findings. The results of the effluent particulate fractionation, which are given in Figure 20, showed the following distribution of elements (percent values relate to the total concentration):

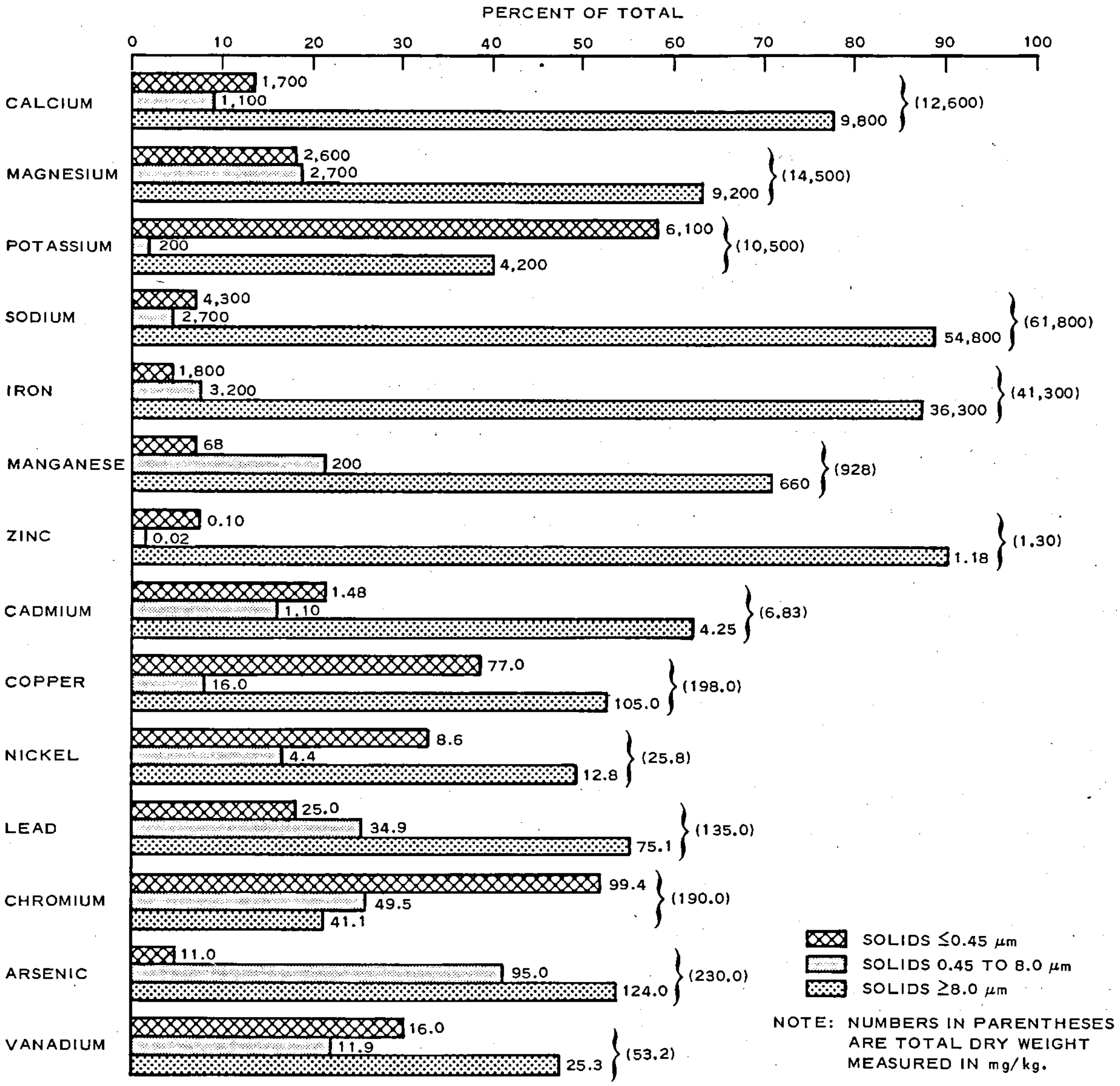
$>$ 8 µm: Zn (91%) > Na (89%) > Fe (88%) > Ca (78%) > Mn (71%) > Mg (63%) > Cd (62%) > Pb (56%) > As (54%) > Cu (53%) > Ni (50%) > V (48%) > K (40%) > Cr (22%)

0.45 to 8 µm: As (41%) > Cr (26%) > Pb (25%) > V (22%) ~ Mn (22%) > Mg (19%) > Ni (17%) > Cd (16%) > Ca (9%) > Cu (8%) ~ Fe (8%) > Na (4%) > K (2%) ~ Zn (2%)

< 0.45 µm: K (58%) > Cr (52%) > Cu (39%) > Ni (33%) > V (30%) > Cd (22%) > Pb (20%) > Mg (19%) > Ca (14%) > Zn (8%) > Mn (7%) ~ Na (7%) > As (5%) > Fe (4%)

277. The above distributions show that zinc, iron, manganese, sodium, and calcium are associated mostly with the large (> 8-µm) particulate fraction, while chromium, potassium, nickel, copper, and vanadium are concentrated in the smaller (< 8-µm) filtrates. Figure 20 shows that the majority of the elements were lowest in the 0.45- to 8-µm fraction. This indicates that for these elements, namely, calcium, potassium, sodium, zinc, cadmium, copper, nickel, and vanadium, two entirely different phases may have been involved in metal binding.

278. Geochemical partitioning correlation. Since a geochemical phase partitioning analysis was performed on an aliquot of these same centrifuged solids, comparisons with the particulate fractionation data were attempted. One previous assumption was that the exchangeable phase should correlate with the finer (< 0.45-µm) filtrate. However, no correlation was obvious upon careful examination. Another assumption was that the larger particulate fraction (> 8 µm) might be composed mostly of lower density organic detritus. Unfortunately, no good correlation seemed to exist between this fraction and the organic-sulfide phase of the geochemical phase partitioning analysis. In



NOTE: NUMBERS IN PARENTHESES ARE TOTAL DRY WEIGHT MEASURED IN mg/kg.

Figure 20. Chemical analytical data for total acid digests of three particle size fractions of effluent suspended solids collected from the confined disposal area (pond 2) at Seattle, Wash., by continuous high-speed centrifugation

fact, none of the geochemical phases could be matched with the orderly array of elements in each effluent solids size fraction. These findings tend to show that the different geochemical phases, as differentiated by the extractants used, were not confined to any one size fraction. Additional particulate fractionation data for effluent solids from confined land disposal areas can be found in another DMRP report.[145]

279. Although overall trends were not observed among any large group of elements, the particulate fractionation study, strengthened by the geochemical phase partitioning data, did indicate some trends for specific elements. For example, the fine fraction of chromium seemed to represent a very fine precipitate, since the partitioning data showed very low levels of chromium in the exchangeable, carbonate, and easily reducible fractions and a small increase in the moderately reducible-residual fractions (Figure 19). Potassium appeared to be associated with very fine, mainly less than 0.45-μm crystalline particles. Although exchangeable potassium may contribute to the total potassium associated with the fine fraction, over 90 percent was bound in the moderately reducible-residual phase. Zinc occurred almost totally in the fraction larger than 8 μm. The partitioning data showed a substantial increase in carbonate phase zinc, while the larger fraction should have contained higher levels of low-density detritus. An association may thus exist between the organic particulate fraction and carbonate complexes of zinc. However, the exact relationship is not apparent.

Relation of dredged material texture and solids content to contaminant release - Richmond site

280. Sample descriptions. The influence of the dredged material texture and solids content (slurry dilution) on contaminant mobility could only be evaluated appropriately at the freshwater riverine dredging site near Richmond, Va. None of the other monitored sites showed sufficient variability in both of these parameters. The following list shows the relationship between texture and nonfilterable solids in each of the composite samples collected at Richmond. The remaining data for this site are given in Appendix B, Table B14.

Sample No.	Texture, %			Nonfilterable Solids, percent
	< 2 μm	2 to 50 mm	> 50 μm	
1	1	2	97	1.70
2	4	17	79	1.06
3	13	27	60	0.58
4	26	49	25	3.30

281. The first sample, showing 97 percent sand-sized particles, also appeared to have a respectively high organic content, with organic carbon and organic nitrogen concentrations in the dry weight solids of 3.7 and 0.36 percent, and a high cation exchange capacity of 50. The second sample showed higher organic carbon, organic nitrogen, and cation exchange capacity than the third sample, although both of these samples were lower than the first sample in all three parameters. The fourth sample came from a similar section of the river but in a quiescent area adjacent to the Deepwater Terminal docks; this sediment contained 75 percent silt and clay and also showed a high organic content, with organic carbon and organic nitrogen values of 3.4 and 0.23 percent, respectively, and a cation exchange capacity of 37. The nonfilterable solids were highest for the fine-grained dredged material; the other samples showed dry weight solids increasing with increasing sand content.

282. The fine-grained sample (No. 4) appeared to be the most reduced, containing high levels of both organic matter and sulfides (91.6 mg/kg). The next most reduced sediment seemed to be the most sandy sediment (Sample 1), based on its comparatively high sulfide (73.8 mg/kg) and organic contents. Samples 2 and 3, which were intermediate in texture, appeared to be the most oxidized.

283. Texture-solids relationship. The data presented in Appendix B, Table B14, show that the changes in the total contaminant levels, for all the parameters measured in the influent slurries, were directly related to changes in their respective solids contents. The soluble phase concentrations failed to show this relationship with the exception of iron and manganese. These elements should have been highest in the sediment pore water. There also appeared to be a good correlation

between total organic carbon and filterable carbon and between total organic nitrogen and both filterable organic nitrogen and ammonium nitrogen. The cation exchange capacity seemed to correlate mainly with the organic content, as indicated by influent sample 1. The sediment at collection site 4 contained a high silt and clay content, yet these sediments contained a lower cation exchange capacity than did the sandy sediments at site 1. Thus, in agreement with similar findings,[85] organic matter was the main contributor to the cation exchange capacity.

284. The mobility of the different metal contaminants at the Richmond site showed considerable variability with changes in influent texture and solids content (slurry dilution). Both iron and manganese appeared to be most readily released from coarse-grained sediments, especially those containing high levels of organic matter (high cation exchange capacity). This release may also be partly related to the larger volume of pore water in the coarser grained sediments. Manganese was not readily mobilized from the reducing environment of the fine-grained sediments, even though the associated solids contained a comparatively high level of manganese. Zinc seemed to be mobilized most readily from the fine-grained sediment, while filterable cadmium appeared to be scavenged by the fine-grained sediments, resulting in a significant depletion over background water levels. Lead mobility seemed to be directly related to the organic content or cation exchange capacity, while the remaining elements showed mixed results. However, because most trace metal concentrations were quite similar in the different filtrates at the Richmond site, further similar research seems warranted.

285. Summary. The foregoing discussion indicates that texture may be important in governing the concentrations of some contaminants in the sediment pore water, mainly by dictating other physicochemical conditions (e.g., Eh and pH). However, the organic content, which could be inferred by cation exchange capacity or organic carbon determinations of bottom sediments, seems to be much more important than texture. The greater ion mobility in coarse-grained sediments with a high organic matter content could possibly be related to the entrapment

of easily degradable organic compounds in the rapidly shifting sands. The clay sediments, indicative of a low-energy environment, may contain less easily biodegradable organic matter. The biodegradation of the organic matter could then result in localized areas of high acidity in the more poorly buffered sandy environment, and also low Eh. The high acidity and low Eh may then aid in mobilizing many metals, particularly iron and manganese. The rapid movement of aerated water into the sands may also accentuate the rate of organic matter breakdown, thus contributing to high levels of ammonium nitrogen in the pore water. The rapidly fluctuating Eh conditions should also favor the mobility of many metal contaminants.[17,18]

Disposal area vegetation-dredged material interactions - Southport site

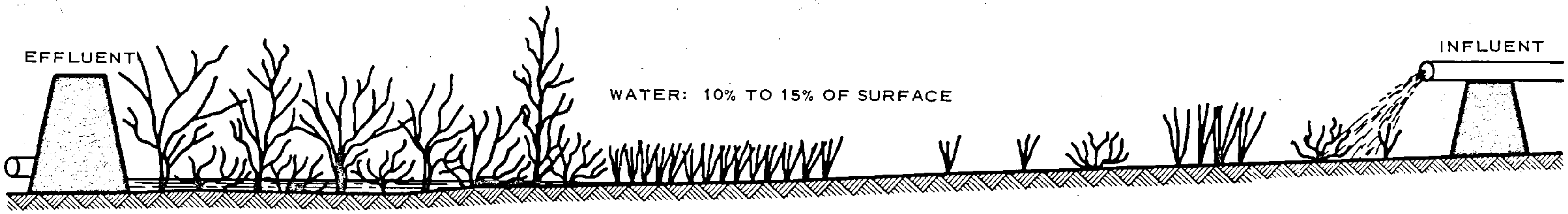
286. Site description. The 48-acre confined disposal area on Oak Island, just south of Southport, N. C., was monitored on two separate occasions during a dredging and disposal operation. The site was about 80 percent covered with three distinct vegetation communities; namely, saltmarsh grasses in the southern third, brackish and upland grasses and brush in the northern and eastern sections, and thick upland forest vegetation in the western third of the containment area (Photos 17 and 18). A detailed description of the disposal area and its vegetation communities is given in Part III and in Figures 5 and 11.

287. Three influent and effluent slurry and water samples were collected on each of two trips to the disposal area; the first was initiated one day after dredging began and the second was 2 weeks later, toward the end of the dredging operation. The purpose for this dual sampling was to assess the impact of thick vegetation in a confined disposal area on effluent water quality during a disposal operation. The initial assumption was that by using the same disposal area, several variables could be reduced in magnitude. These variables included: the effective disposal area size, influent pumping rate and solids content, and the physicochemical composition of the dredged material slurry. The major factor anticipated to change was the surface area of vegetation in contact with the dredged material slurry. During the

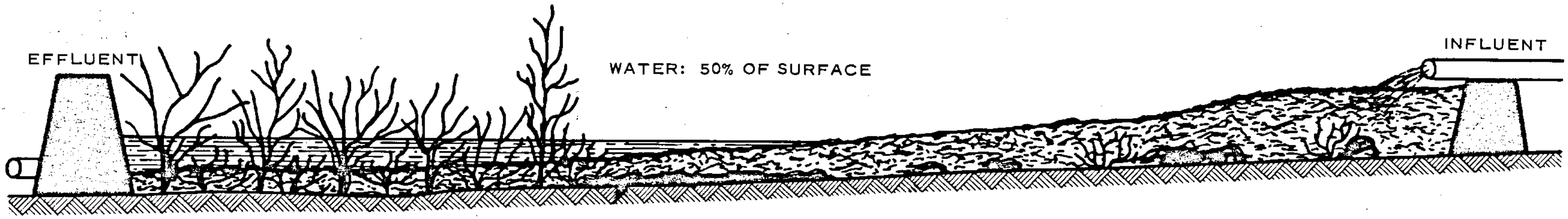
initial sampling trip, most of the vegetation was in contact with the slurry, whereas most of the grasses and short vegetation were buried at the time of the second sampling. The total surface area of vegetation was estimated to be significantly reduced after 2 weeks of disposal. However, an unforeseen complication developed; namely, the amount of ponding in the disposal area during each sample collection trip. Initially, ponding covered only 10 to 15 percent of the lower end of the disposal area (Photo 18). During the second trip, about 50 percent of the disposal area was covered by quiescent waters (Photo 19). This increase in ponding may have served to counteract the suspected filtering effect of the thick vegetation. The situation is depicted in Figure 21, where initially there is minimal ponding and maximum interaction of the slurry with the vegetation and finally there is extensive ponding in conjunction with burial of the shorter vegetation. Figure 22 shows the influent and effluent total and soluble phase (< 0.45-μm) concentrations of nutrients and metals in samples collected on the initial and final trips to the Oak Island site. The rates of chemical change during disposal area retention are indicated by the slopes of the lines drawn between the influent and effluent mean values, assuming that retention times were comparable during the two sample collection trips. Ranges are shown by the vertical lines. The rough data for samples from the first and second collection trips are given in Appendix B, Tables B18a and B18b.

288. The extensive subsampling of influents and effluents should have made good correlations possible between influents and effluents for daily and average samples from each trip. However, even if there is a poor relationship between respective influent and effluent samples, the rates of change (slopes of the lines) should indicate how one element is changing with respect to the other elements at the times of collection.

289. Solids removal. Influent solids fluctuated greatly during the sampling period, ranging from 3.0 to 32.2 percent by weight. However, the average solids in samples collected on each trip were similar, namely 17.1 and 18.5 percent (Table 8). One should assume that great



a. INITIAL DISPOSAL AREA SAMPLING TRIP



b. FINAL DISPOSAL AREA SAMPLING TRIP

Figure 21. Vegetation-dredged material slurry interaction in the Oak Island disposal area, Southport, N. C.

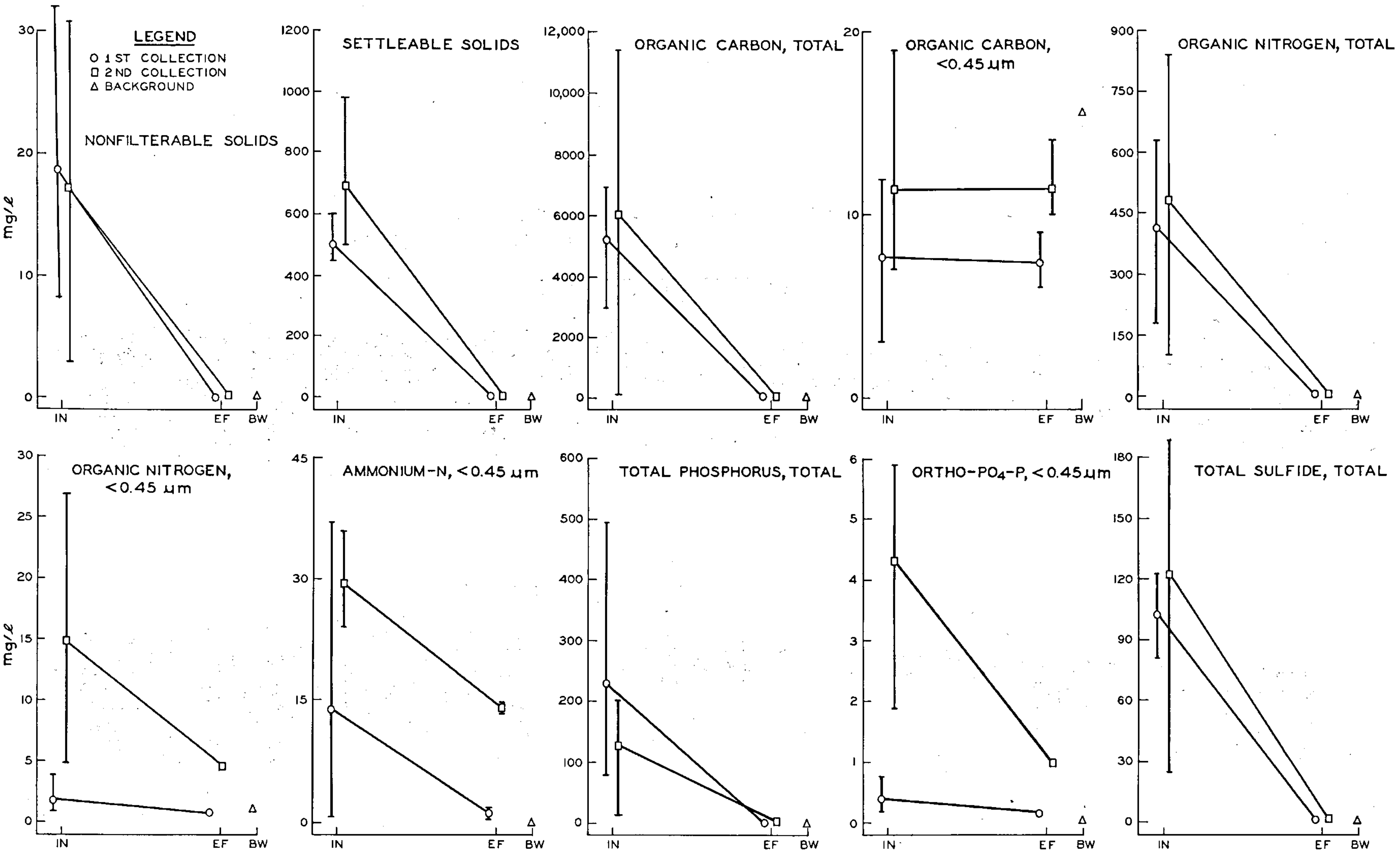


Figure 22. Comparative data for total and soluble phase nutrient and metal concentrations in influent and effluent samples, collected on the initial and final trips to the vegetated Oak Island disposal area, Southport, N. C. Rates of chemical change during residency are shown by the slopes of the lines drawn between the mean values; ranges are shown by the vertical lines (sheet 1 of 4)

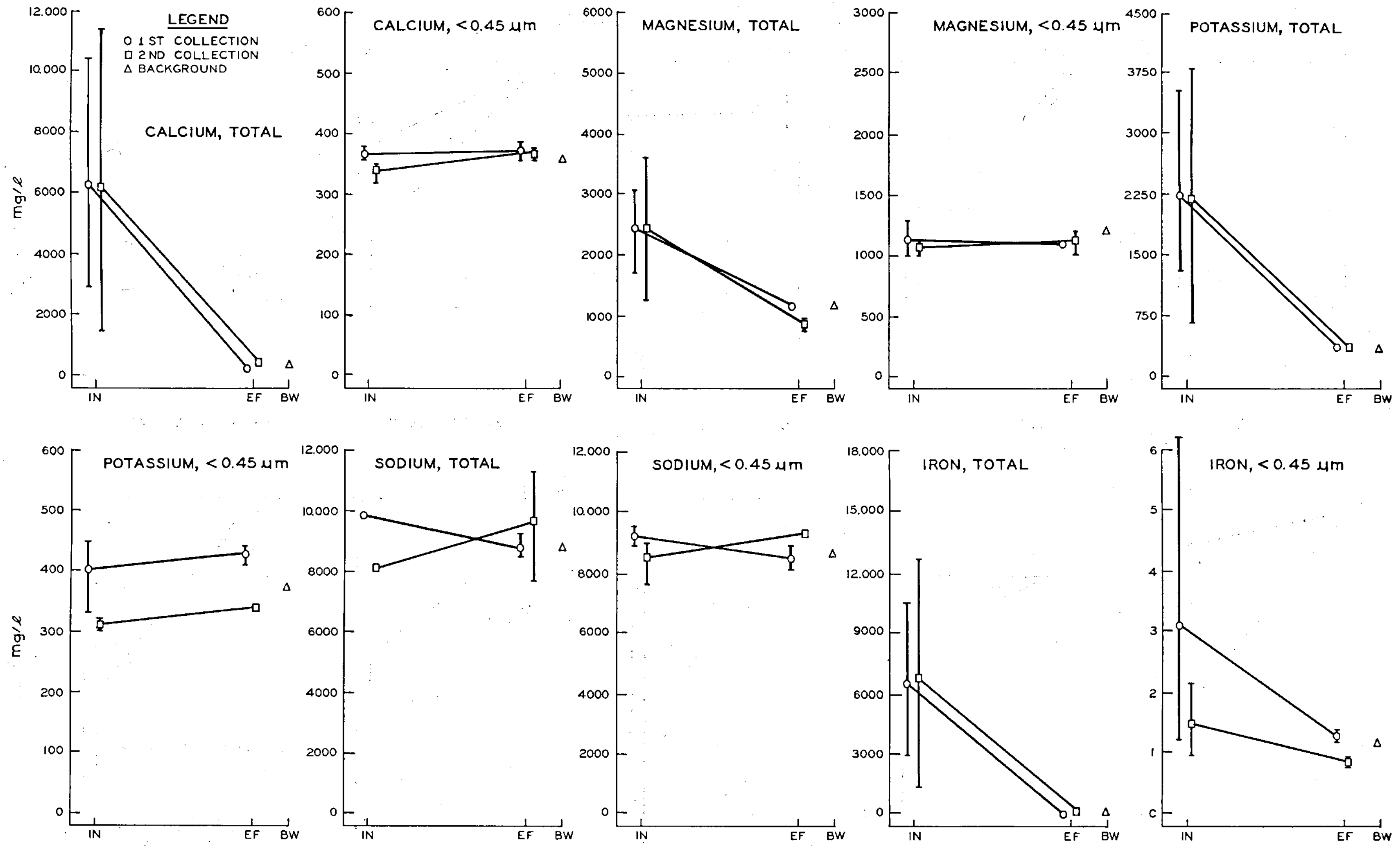


Figure 22 (sheet 2 of 4)

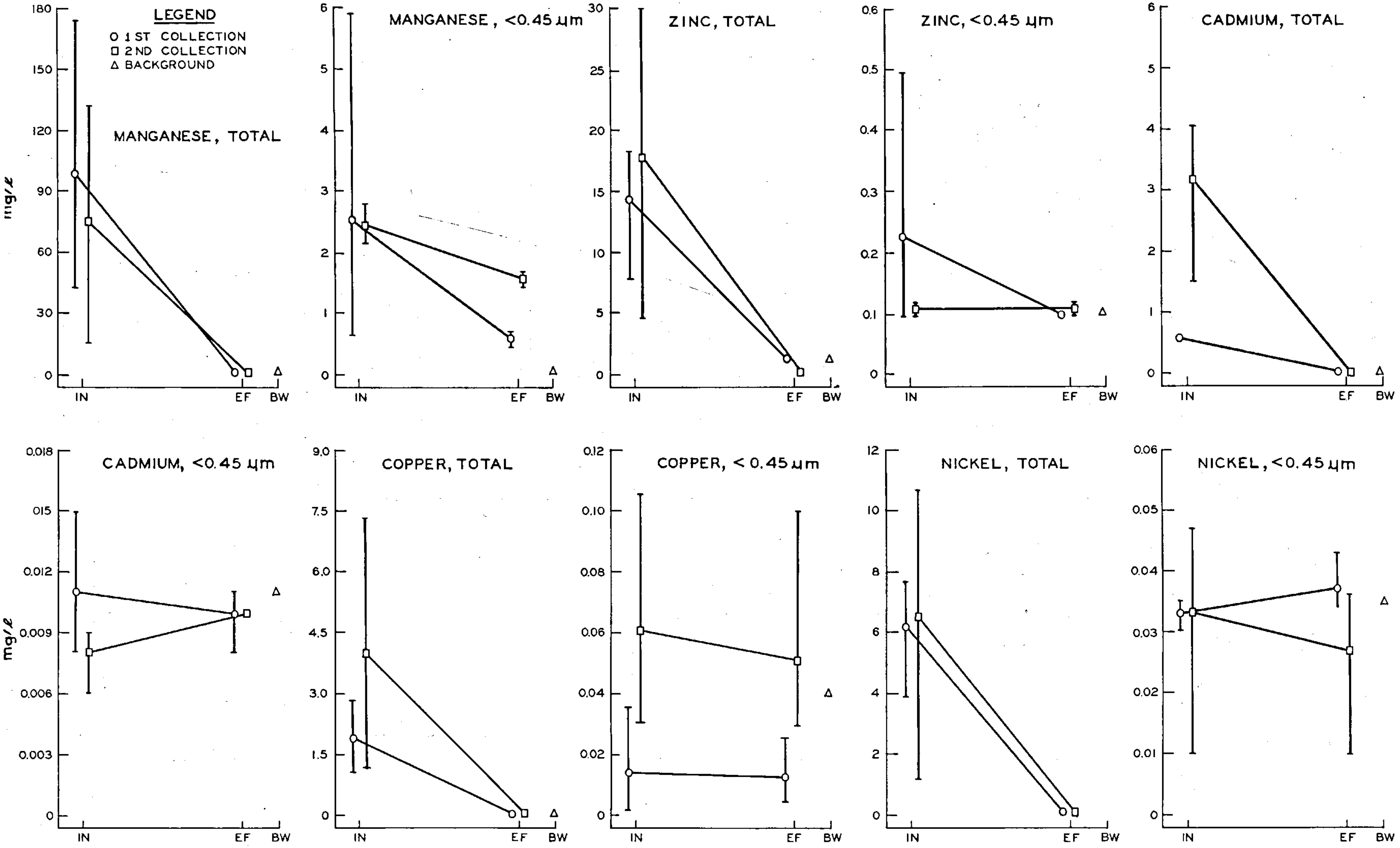


Figure 22 (sheet 3 of 4)

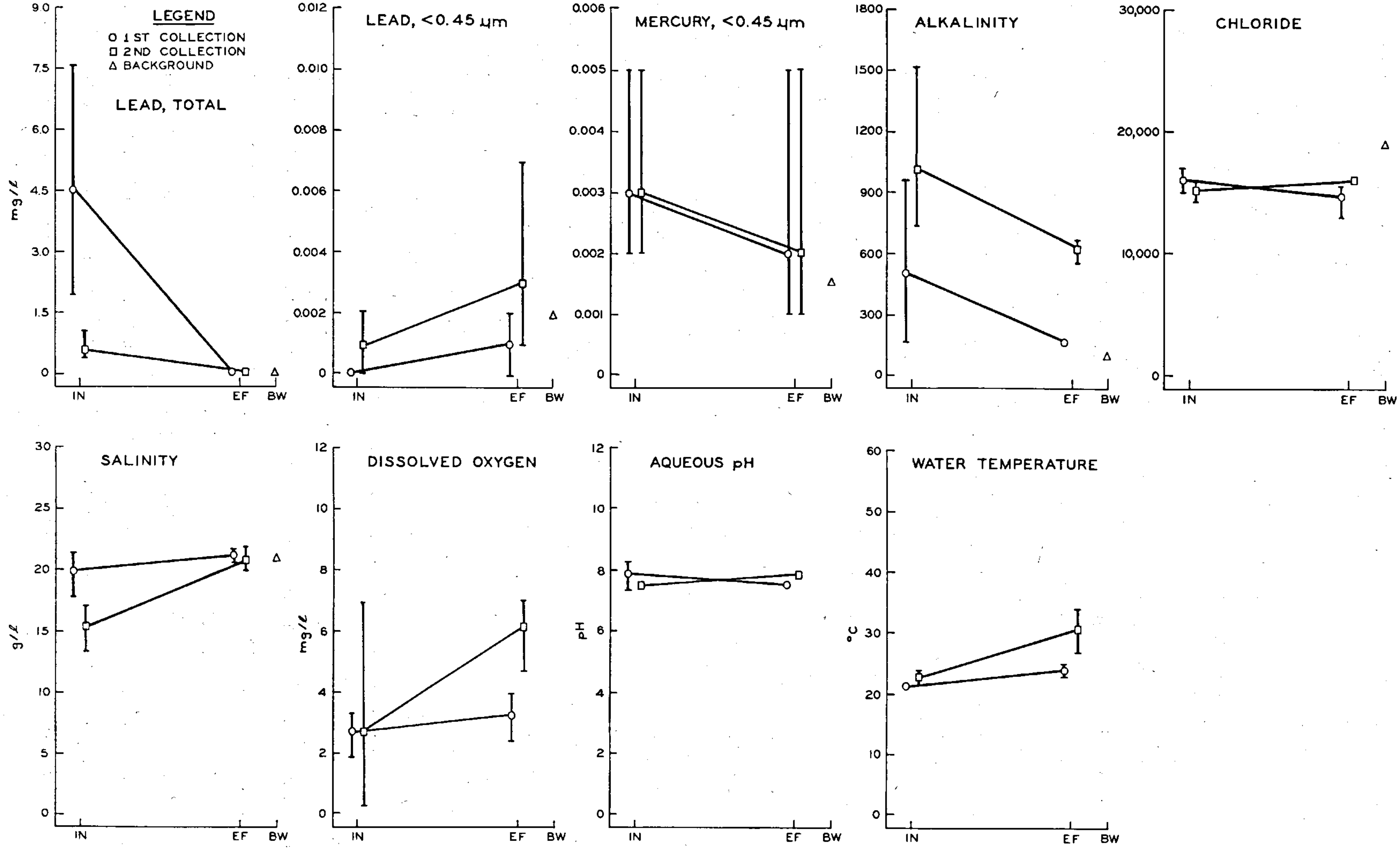


Figure 22 (sheet 4 of 4)

variability in influent solids will continually occur but that these fluctuations will not be highly noticeable in effluents. The character of a given influent pulse should be significantly masked during its movement across a large disposal area. It must also be emphasized that the collected influent samples may not correspond well with the collected effluents. The main purpose for the influent sampling was to obtain a rough estimation of what was entering the disposal area. Greater emphasis should be given to influent filtrates than to total influents as the former should and did show less time-dependent variability.

290. Effluent samples from the Oak Island site showed the lowest turbidity and suspended solids content of any of the other sampled sites, except for the Seattle operation where a chemical flocculent was employed. Samples collected on the first trip varied from 0.008 to 0.025 percent by weight, and showed almost no visible color or turbidity (see Photo 19). In comparison, the clear background surface water contained 0.009 percent solids by weight. The final set of effluent samples showed a very slight increase in visible color and turbidity (variability being nonsignificant by the F-test) with suspended solids values ranging from 0.048 to 0.366 percent by weight in comparison to a background level of 0.027 percent (Figure 22). The discharge water during this final collection showed a noticeable brown coloration but it was quite clear in comparison to most of the disposal area effluents encountered. It should be emphasized that effluent comparisons to surface background water, collected near the effluent discharge, may not necessarily have a close relationship to the quality of bottom water at the dredge site.

291. Nutrient removal. Vegetation should most readily remove the nitrogen and phosphorus compounds in dredged material slurries as these are required in the greatest quantity by actively growing plants. Since the site was monitored during May, most of the vegetation was considered to be in an active state of growth. Decaying vegetation and associated microflora could also elicit a net release of these nutrients, as well as organic carbon compounds. The main source of nitrogen in influent samples was from organic compounds, but most of the soluble

phase nitrogen was in ammonium form. Ammonium was the dominant form of nitrogen in effluents, and essentially all of the organic nitrogen was associated with the filtrates of samples collected on both trips to the site. Also, most of the phosphorus associated with the effluent samples was in the soluble orthophosphate form.

292. Only the soluble organic carbon and orthophosphate phosphorus showed significant variability between similar type samples collected on the two trips to the disposal area (using the F-test). The carbon appeared to show generally no change in concentration across the site (Figure 22), and values were always at or below surface background water levels. The organic compounds involved seem to have been mainly poorly degradable molecules (e.g., fulvic acid) derived primarily from the dredging site water, as the soluble phase influent and effluent values fluctuated in unison with background water levels. A significant difference was noted for orthophosphate phosphorus in both sets of influent and effluent filtrate samples, using the Student's t distribution. A decrease in the soluble phosphorus concentration was noted in both sets of effluent samples, but the lesser slope for the initial samples was probably promoted by the low influent orthophosphate concentration, which approached background levels (Figure 22). Although the actual influence of the vegetation on phosphorus removal cannot be verified by the data, it appears that soluble phosphorus compounds can be reduced to near background levels in confined disposal areas containing actively growing vegetation, and that partial burial of thick vegetation during disposal operations does not promote a net release of phosphorus.

293. Soluble organic and ammonium nitrogen in influents from the initial and final collections were not found to be significantly different (using the F-test), probably because of the great variability in replicate samples. However, effluent comparisons for ammonium nitrogen were shown to be significantly different, using the Student's t distribution. Figure 22 suggests that the ammonium removal rate was equivalent during both the initial and final collections, as the parallel slopes indicate. This would assume that influent values

were representative of the respective effluent concentrations and that average residence times were comparable. However, previous studies[44] showed that ammonium removal is nonlinear at low concentrations, probably because of a substrate-limiting reaction rate. Additionally, a slightly higher average pH value of 7.9, increased temperature, and apparently longer residence time of the effluent water during the second collection trip should have favored increased ammonia volatilization or microbial assimilation at this time. Considering the above, it appears that the disposal area was initially more proficient for ammonium removal, indicating that the vegetation may have played an important role in this removal.

294. Metal fluctuations in total effluent samples. The elemental composition of total effluents shows the close relationship between total solids and contaminant removal; namely, total effluents and less than 0.45-μm effluent filtrates were similar because of the low solids content of the disposal area discharge water. Also, total effluents collected on the initial trip were most often lower in contaminant levels than were total effluents collected on the final trip, when slightly higher solids were observed. However, there were no significant differences (using the F-test) between the two sample sets, which was partly attributed to the high influent solids variability, as shown in Figure 22. The only elements which were higher in total effluents from the first sample collection are magnesium, zinc, nickel, and lead. Fluctuations in the magnesium and zinc could be primarily controlled by biological processes, induced by the vegetation and microorganisms. However, the true interaction is unclear as vegetation growth and decomposition were probably occurring simultaneously during the entire disposal operation. The trend for lead may be due to sporadically high lead values in influent solids during the initial sampling; this trend is not similarly depicted by concentrations in either influent or effluent filtrates, which indicates that the variation may be due to analytical error. The higher nickel in total effluents from the first collection trip can be attributed to the observed nickel increase in the respective effluent filtrates.

295. Metal fluctuations in effluent filtrates. The dredging site was in a brackish water regime, with resultant high levels of sodium, potassium, calcium, and magnesium. Values for the above major cations agree closely with the chloride values, indicating that the bottom water at the dredge was about 80 percent of marine salinity. Water of about 25 ppt salinity is probably detrimental to most of the upland vegetation in the disposal area, and this assumption is verified by a high percentage of dead trees on the site. There appeared to be a general change in the vegetation structure in the disposal area toward salt-tolerant species. Evapotranspiration appeared to account for less than a 10 percent decrease in the influent water volume, based on comparisons of influent and effluent chloride concentrations in samples from the second collection. The effluent samples from the initial trip showed about a 10 percent decrease in chloride, which may have been partly caused by slurry dilution with the rain or groundwater observed in the borrow ditches previous to disposal. If the vegetation was removing and excreting large quantities of salt from the water, the evapotranspirational loss could be underestimated.

296. There were some interesting changes in the trace metal concentrations in the less than 0.45-μm filtrates. During the initial collection, when very little of the vegetation was buried and ponding was minimal, only nickel and lead increased as the slurry flowed across the disposal area. During the final collection, when the vegetation surface area was greatly decreased and ponding covered about half of the confinement, zinc, cadmium, and lead increased in effluent filtrates. Decreases in sodium, iron, manganese, zinc, cadmium, and copper were noted initially, whereas iron, manganese, copper, and nickel decreased in the final set of samples. Iron, manganese, and zinc appeared to decrease at a more rapid rate initially, while copper and nickel showed the greatest declines in the final samples. The greater initial decreases could be due to dilution of the first set of effluents by the ponded borrow ditch water and/or by increased evapotranspiration of the second set of effluents. A water temperature increase of about 5°C, measured for the effluents on the final trip, could account for an

increased evaporative loss and also for greater dissolution of contaminants from the solids (Table 6 and Figure 22). Other factors to consider would be the release or fixation induced by the increased residence time associated with the greater ponding, and the release of organic chelates from the vegetation. It would be difficult to comment further on these mechanisms with the collected data and the many variables present. Although organic carbon concentrations of effluents and background water were similar, the types of organics present, which are of most importance, were not determined.

297. The overall impact of vegetation. The Oak Island disposal area proved to be extremely efficient in the removal of solids from dredged material before effluent discharge. This efficiency was noted on both sampling trips although the initial effluents were visually and experimentally cleaner. However, vegetation contact with the dredged material persisted during the entire disposal operation, and thus one was dealing mainly with the degree of contact. Considering that slurry residence time probably continually increased during the disposal operation as a result of increased ponding, the thick vegetation on the site seemed to have a remarkable capacity for the removal of suspended solids, ammonium nitrogen, and soluble phosphorus, if at high concentration in influents. Efficient solids removal by vegetation has also been observed in other field operations.[43] The release of total contaminants is closely related to solids removal as most contaminants are associated with the solid phase. When dealing with very fine particulates, such as the suspended solids in disposal area effluents, perhaps a better estimation of environmental impact should be obtained by including the total contaminants associated with these solids as they may be available to living organisms. This study indicates that thick vegetation is equally as efficient as ponding in reducing the suspended solids content of dredged material slurries. Further conclusions about vegetation interactions should result from additional research with better controlled field plots, using different vegetation communities during different seasons.

Assessment of the standard elutriate test for land containment areas - Seattle site

298. General. The standard elutriate test was originally developed as one means of assessing the potential environmental impact of the open-water disposal of dredged material. However, the applicability of this test to predict the mobility of various contaminants from confined land disposal areas has not been emphasized. A detailed description of the procedures and evaluation of the standard elutriate test can be found in the literature[157-159] and in Part III of this report. The initial methodology for the test involved the collection of bottom sediments at the dredging site and mixing these sediments in a 4 to 1 volumetric ratio with unfiltered water from the disposal site.[158] Later, the test was modified to specify dredging site water.[159]

299. Test evaluation. The data from the standard elutriate tests for the Seattle site are given in Appendix B, Table B16. For comparative purposes, the standard elutriate test filtrates were compared with the sediment pore water from the dredging site. In order to determine the chemical changes which occurred during the shaking and settling periods of the elutriate test, the sediment pore water contaminant concentrations were divided by four to simulate the same dilution used in the elutriate tests. Table 15 gives the averaged data for the sediment, sediment pore water, diluted sediment pore water, and the standard elutriate contaminant concentrations. The shortcomings of the standard elutriate test were evaluated by comparing its data with chemical concentrations in actual effluents from pond 1 (short residence time) and from pond 2 (longer residence time) of the disposal area. Data from the diluted sediment pore-water calculations served as a control for chemical changes induced by actual dredging and disposal operations; namely, agitation, aeration, actual dilution effects, salinity and temperature fluctuations.

300. Based on the analytical data comparisons given in Table 15, chemical concentrations in the cored sediments closely approximated those in the influent dredged solids. Dry weight solids comparisons, for different chemical species in the slip 1 sediments versus the

disposal area influents, showed a poor correlation only for oil and grease, PCB's, exchangeable ammonium nitrogen, and mercury. The other parameters gave comparable averages. Extensive coring of sediments at the dredging site is mandatory for this predictive test. As stated above, the elutriate test calls for a fixed 1 to 4 dilution ratio of the sediments with dredging site water; the average sediment and influent solids of 44.3 and 6.7 percent, respectively, indicate that the actual sediment dilution may have been greater than 1 to 6 at the Seattle disposal area. Sediment pore water may be a major source for contaminants released during dredging. Since there is often a close relationship between the quantities of sediment and the sediment pore water dredged, a variable dilution factor seems warranted. A ratio should be used which more closely duplicates a site-specific dredging operation. The solids removal rate could be approximated once the dredging contractor has been chosen.

301. Summary. A summary of the standard elutriate and diluted pore water test findings, plus concluding remarks, are included in Table 16. Independent conclusions and remarks are given in a separate report.[26] Based on the results of these tests, the standard elutriate appeared to be slightly superior to the diluted pore water for predicting filtered effluent water quality from both ponds at the Seattle site. Additionally, the standard elutriate test seemed to predict effluent water quality from pond 1 slightly better than for pond 2. However, the overall results of the standard elutriate and diluted pore-water tests seemed to indicate inadequacies in their present state for predicting the soluble phase concentrations of contaminants in the confined disposal area effluents at Seattle.

302. According to Table 16, only 8 and 7 parameters of the diluted pore water calculations showed variability of less than ± 3 times the pond 1 and pond 2 effluent filtrates, respectively. In comparison, the standard elutriate had 9 and 10 parameters with variability of less than ± 3 times the pond 1 and pond 2 effluent filtrate values, respectively. Because most of the values were very small, in the parts per billion range, the variability was often quite nonuniform. However,

some parameters seemed to be quite stable to change. For instance, if the effluent concentrations of a given chemical species were closely predicted by both tests, then its effluent concentrations were also often similar in both pond 1 and pond 2 effluent filtrates. Organic carbon, nitrate nitrogen, mercury, chromium, and arsenic fit the above category. Previous findings suggest that soluble phase organic carbon, mercury, and chromium are very stable parameters, and respond very slowly to changing environmental conditions. The nitrate concentrations originate from the water column and in the surficial aerobic sediments, which suggest that this close correlation may be coincidental. The data for arsenic are difficult to explain, but they probably result from sample variability; a longer detention time should tend to reduce the arsenic concentration to levels approximating pond 2 effluent filtrates.

303. The previously discussed findings strongly suggest that the settling time for the standard elutriate test should be variable and site specific, using a settling interval which more closely approximates the dredged slurry residence time in the containment area under consideration. The results of this study do not indicate whether an exact duplication of the slurry detention time would provide the most meaningful results. More specifically, the small-scale test may not precisely duplicate the field conditions. There were strong indications that a longer settling time for the Seattle elutriate tests would have resulted in better comparisons between the elutriate test and the disposal area effluent filtrate chemical concentrations, especially for pond 2.

304. The Seattle data for soluble phase iron and phosphorus are good examples of the effect of increased retention time on contaminant mobility (Table 15). The diluted sediment pore water iron concentrations were considerably higher than the effluent levels from either pond 1 or 2, whereas the standard elutriate iron levels were more comparable to pond 1 and pond 2 effluent filtrates. Iron rapidly forms highly insoluble precipitates under the oxidizing conditions present in both disposal area ponds and in the standard elutriate. In contrast, phosphates are removed more slowly than iron, usually by sorption with calcium and iron precipitates. This slower reaction is demonstrated

by similarly high soluble phosphorus levels in the diluted sediment pore water and standard elutriate filtrates, and the filtrates from pond 1. These three filtrates were either in a reduced environment or were subjected to an oxidizing environment for only a short period of time. The depletion of soluble phosphorus in pond 2 filtrates, resulting in poor correlation with the standard elutriate test data, probably resulted from soluble phosphate scavenging by the iron precipitate observed in this pond, which was promoted by the long retention time in an oxidizing environment.

305. Problems. Certain additional variables may be involved in the differences observed between the elutriate test and the disposal area effluent filtrate characterizations. At the Seattle site, there was considerable contact between the dredged slurry and a porous sandy loam dike material, probably to a much greater extent than at most confined disposal areas. The data (Table 15 and 16; and Appendix B, Table B16) indicate that there might have been some additional release of contaminants, especially of iron, manganese, and zinc, from the dike sediments. This could be promoted by the rapid reduction of the high organic, porous dike sediments as a result of submergence. The reducing conditions would tend to release metals bound with oxidized iron precipitates and some of the organic complexes. Thus, the interacting soil or older dredged material must also be considered, although the major impact should occur during the initial flooding, before fresh sediments have accumulated. The slurry residence time at the Seattle site was also much longer than what would be indicated by its small size. The poor correlation between the standard elutriate test and effluent soluble phase PCB concentrations probably resulted from the nonhomogeneous concentration of PCB's in the slip 1 sediments. The tendency for many organic compounds to concentrate at the water surface or be sorbed at solid surfaces should also be kept in mind while conducting these tests.

306. Similar environmental conditions will probably never be duplicated in different disposal areas. Thus, it may be very difficult to develop a quantitative elutriate test. However, in most cases

oxidizing conditions will develop in land containment areas initiating a directional trend in geochemical recycling of chemical constituents. Therefore, the test should at least provide a qualitative estimate of what contaminants would tend to be released over time. Several time-variable tests may prove to be best for determining these changes.

PART V: MAJOR FINDINGS AND CONCLUSIONS AND RECOMMENDATIONS

Major Findings and Conclusions

Influent and effluent characterizations

307. Removal efficiences. The major findings and conclusions for total and soluble phase chemical constituents are as follows.

a. Total. The removal efficiencies for most total metal concentrations (Fe, Zn, Cd, Cu, Ni, As, V, Pb), closely approximated the removal of total or nonfilterable (suspended) solids, each with removal efficiencies of 96 and 97 percent, respectively. The metals with removal efficiencies of ≤ 90 percent include calcium (90 percent), titanium ($\overline{8}9$ percent), manganese (88 percent), potassium (78 percent), magnesium (64 percent), mercury (46 percent), and sodium (9 percent).

Most total nutrient concentrations (total organic carbon, organic N, Total P) showed removal efficiencies approximating the total solids removal. Total ammonium nitrogen removal was only 57 percent, with average and maximum effluent concentrations of 19.6 and 80.3 mg/ℓ, respectively.

The chlorinated hydrocarbon materials generally showed good removal efficiencies. Only DDT, DDD, DDE, and PCB analogs were detected in total influent samples at concentrations greater than 0.1 mg/ℓ; DDE was not removed very efficiently during land containment, but comparable levels were usually also observed in surface background water samples.

Oil and grease can be effectively removed during land disposal. High petroleum concentrations were noted to inhibit the settling of solids, resulting in the formation of low-density emulsion layers at shallow depths in ponded areas. Poor management could result in the discharge of the oily substrata, which could also release entrapped solids and contaminants; surface films also have the capacity to concentrate certain contaminants although this study did not attempt to substantiate the above.

There was an overall decrease in particle size in the dredged slurry during land containment, although turbulent mixing near the effluent weir resulted in the resuspension and discharge of sand-sized particle fractions during containment area detention.

There was a 60 percent increase in the cation exchange capacity of effluent solids; this increase was reflected by general increases of exchangeable ammonium and metals associated with the effluent solids. Low solids effluents should be negligibly affected by the increase in cation exchange capacity.

b. Soluble. Chemical constituents which showed soluble phase decreases of less than 50 percent in effluents include manganese (38 percent), ammonium N (35 percent), alkalinity (30 percent), organic C (30 percent), cadmium (25 percent), vanadium (17 percent), nickel (14 percent), chloride (6 percent), magnesium (4 percent), sodium (4 percent), titanium (3 percent), lead (0 percent), and mercury (0 percent); parameters which showed increases in effluents are: potassium (2 percent), copper (11 percent), calcium (13 percent), chromium (13 percent), zinc (16 percent), salinity (17 percent), dissolved oxygen (40 percent), and nitrate + nitrite -N (94 percent).

A significant proportion of soluble phase Ti, Hg, Pb, and DDE appeared to be associated with very small colloidal sized particles. At the Seattle site, important fractions of potassium, chromium, copper, nickel, and vanadium were associated with solids which could pass through a 0.45-μm membrane filter, although the removal of total metals at this site was excellent. Much of the soluble phase iron and chromium (Seattle site) was thought to be associated with filterable precipitates.

Poor removal of total ammonium N and manganese seemed to relate to their high soluble phase concentrations in both influents and effluents. Although both parameters showed above average soluble phase removal efficiencies, filterable ammonium N and manganese each showed high average effluent concentrations of 13.6 and 1.45 mg/ℓ, respectively. High soluble phase concentrations of calcium, magnesium, potassium, and sodium, with only small changes during retention, are a reflection of the sampling at six brackish water dredging sites. Three freshwater sites were included in this study, which also occasionally showed moderately high soluble phase calcium, magnesium, and potassium concentrations.

Soluble phase phosphorus or orthophosphate P was usually not a problem during the land disposal of dredged material, except at two of the sites.

308. Mechanisms for constituent mobility. The major findings and conclusions regarding the mechanisms for constituent mobility are as follows.

- a. Geochemical partitioning. Geochemical partitioning data for solid phase material suggest that during containment many metals are mobilized from organic and sulfide complexes, with subsequent increases in more readily available phases. Calcium, sodium, copper, and arsenic (Seattle) showed measurable increases in the exchangeable phase. Carbonate phase increases were observed especially for zinc, cadmium, manganese, lead, copper, and sodium. Iron manganese, cadmium, and copper increased in the easily reducible phase; the only major increase was for manganese.
- b. Alkalinity. Although a direct relationship between carbonate phase shifts and influent-effluent alkalinity values was not consistently noted, high alkalinity undoubtedly played an important role. Alkalinity showed an overall decrease during containment, although the trends were site specific. Major shifts in alkalinity during dredged slurry containment seemed to be promoted by biological activity; the highest influent alkalinity values were noted when the dredged sediments contained a high total organic carbon concentration.
- c. Control of pH. There was a small increase in pH during the dredging and land disposal cycle. Algal photosynthesis appeared to be an important source for the increase, with pH values in excess of 9 observed in effluents from a nonvegetated disposal area with a long retention time. High nutrient levels can increase growth in site waters, resulting in alkaline pH; nitrogen loss through ammonia volatilization and increased ammonia toxicity to microbial pathogens in the dredged material or to the biota in the receiving waters are possible results.

  High turbidity and vegetation in the disposal areas should prevent excessively high pH. Ammonia inhibition of algal photosynthesis will also tend to curb excessively high pH but the resultant rapid die-off of algae could alter the chemical reactions occurring in the disposal areas.
- d. Vegetation. Thick vegetation in land disposal areas appeared to promote very low solids effluents. Actively growing vegetation was shown to elicit notable removal of soluble phase ammonium and orthophosphate; the presence of dead or dormant vegetation

in disposal areas could induce a poor quality effluent, augmented by increased microbial degradation, and release of ammonium and organic detritus.

e. Oxygen status. Dissolved oxygen averaged 5.3 mg/ℓ in effluents, and generally effluents containing higher solids contents were lower in dissolved oxygen. Some very clear effluents which contained high concentrations of soluble nutrients were also low in dissolved oxygen. Subsurface effluent discharge had lower dissolved oxygen than surface discharge.

Low dissolved oxygen values were observed in vegetated overland flow treatment areas. This trend may be prompted by the turbulent mixing and greater contact of reduced sediments in overland flow systems. Influents entering the vegetated overland flow areas were also unusually high in nutrients, which may have prompted low levels of dissolved oxygen by accentuating microbial growth.

f. Residence time and salinity. There was no observed relationship between residence time or salinity changes and disposal area effluent water quality. Other variables seemed to have masked their influence in most cases.

g. Texture. Generally, coarse-grained sediments containing a high organic content seemed to release increased levels of heavy metals, especially iron and manganese. Low apparent release of heavy metals from fine-grained sediment may be partly due to adsorption of the released metals by the fine particulate matter and the lower volume of sediment pore water in many fine-grained sediments. The influence of texture on other environmental variables (e.g., pH, Eh) seemed most important.

Effluent and surface background water characterizations

309. The major findings and conclusions of the effluent and surface background water characterizations are as follows.

a. Background water impact. Comparisons between constituent levels in effluent and surface background water (near effluent mixing zone) showed total lead and manganese to be 125 and 74 times higher in the effluents. High lead concentrations were very site specific with usual levels comparable to other low-level heavy metals. The remaining parameters were at or below the average nonfilterable solids level, which was 47 times higher in effluents than in background water. Soluble constituents which were greater than 1.5 times higher in effluents than in the background water samples include: manganese (25x), chromium (6x), arsenic (4x), vanadium (4x), organic nitrogen (3x), oil and grease (3x), alkalinity (3x), zinc (2.5x), copper (2.5x), iron (2.5x), soluble total phosphorus (2x), orthophosphate phosphorus (2x), and chloride (2x).

b. Predictive tests. The present standard elutriate test is not adequate for quantitatively predicting effluent water quality from confined land disposal areas. However, bottom sediment elutriation tests could be made a useful predictive tool if modified to represent more closely the conditions in land disposal areas. Generally, the standard elutriate test data were better than 1:4 diluted sediment pore water data for predicting soluble phase contaminant concentrations in effluents.

Recommendations

310. The following recommendations are proposed as a result of the findings of this study:

a. Since the majority of the chemical constituents (metals, organic matter, chlorinated hydrocarbons) are associated with the solid phase, adequate residence time for the removal of most of the solids is necessary. If a major contaminant appears to be bound with very fine particulates (e.g., mercury, DDE) or tends to shift to more available phases of the solids (e.g., cadmium), then other measures (e.g., use of flocculants) are warranted. However, an excessively long detention period, especially if nutrients are in high concentration, could lead to increased mobility of some contaminant species.

b. Actively growing, thick stands of vegetation should be used in confined disposal areas to improve the removal of solids and soluble nutrients (e.g., ammonium and orthophosphate). Vegetation with a large surface area should be best, which includes most natural vegetation communities in disposal areas. The placement of saline dredged material on salt-tolerant vegetation is recommended, although salt-tolerant plants will tend to naturally take over a disposal area subjected to high salinity.

c. The removal of above-ground dead vegetation during the dormant winter period would prevent the release of nutrients resulting from the decomposition of this material at a later time. The decomposition processes could promote the mobilization of contaminants. Many grasses and other vegetation could be harvested for animal feed in the fall months, provided that contaminant uptake is not a problem.

d. It is recommended that total effluent samples be analyzed for chemical constituents, excluding any short-term settleable solids. The association of contaminants with less than 0.45-µm solids and the poor replicability often observed for different analytical instruments and techniques when measuring the soluble phase are important reasons. Also, the fate of very fine particulate-bound contaminants after ingestion by living organisms is poorly understood at this time. The shift of metals from poorly soluble sulfide complexes to more available carbonate and exchangeable phases also suggests that effluent solids-bound metals may be more available to organisms, resulting from their release after discharge to the receiving water or upon ingestion. It is not recommended that bulk analysis of bottom sediments or influents be used to determine the pollution potential of a given dredged material.

e. The present standard elutriate test, which was originally designed for open-water disposal, appeared to be a qualitative indicator of soluble phase effluent water quality. A modified test, involving elutriation of bottom sediments with dredging site water, should give better predictive results. Modifications which are recommended include the use of a variable sediment-water mixing ratio to more closely approximate the average slurry solids content expected during the planned dredging operation; a longer mixing and/or settling time to more accurately duplicate the oxidizing conditions and solids settling time in the land containment area; and use of unfiltered water, obtained from the settling phase of the test, for chemical analyses, bioassessment testing, or other predictive methodologies.

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Table 1

Descriptions of the Dredging and Confined Land Disposal Areas and Operations

						Description of Disposal Area				Description of Treatment				
Site No.	General Location	Location of Dredging Site	Location of Disposal Area	Predominant Dredged Sediment	Effluent Discharge Site	Size of Diked Area acres	Size of Treatment Area, acres	Shape	Remarks	Treatment Modes	Effective Length of Treatment Area, yd	Overland Flow Distance, yd	Size of Ponded Area acres	Description of Vegetation
1	Sayreville, N. J.	South channel of Raritan River, mile 5 (brackish water)	National Lead Industries Disposal Area No. 4, adjacent to river	Dark grey silt	Raritan River (surface brackish water)	44	≈ 40	Roughly circular	Divided into 3 equal rectangular compartments; sluice box discharge	Overland flow Ponding Vegetation interaction Internal dikes	1400	≈ 700	≈ 35	70 percent cover of half of site by common reed grass (*Phragmites communis*)
2	Houston, Tex.	Houston Ship Channel at sta 1040+00 and in ship turning basin at sta 1080+00 (brackish water)	East half of Clinton disposal area, about 1 mile inland from channel	Fine reddish sand and silt, often heavily impregnated with weathered crude oil	Hunting Bayou-Houston Ship Channel, 2 miles east of dredging site (surface fresh to brackish water	280	≈ 225	Rectangular	Large sluice box discharge	Overland flow Ponding	1300	≈ 200	≈ 200	Sparse
3	Grand Haven, Mich.	Grand River, main channel at sta 120+00 (freshwater)	Verplank's Coal & Dock Co., disposal area, Ferrysburg, Mich., adjacent to river	Fine sand with some oily fines	Grand River (surface freshwater)	6	≈ 6	Roughly rectangular	Sluice box discharge	Ponding	250	--	≈ 6	None
4	Wilmington, N. C.	Anchorage Basin, Cape Fear River (fresh to brackish water)	Eagle Island disposal area, between Cape Fear and Brunswick Waterways	Dark grey silt and clay	Brunswick Waterway (surface fresh to brackish water)	525	≈ 400	Circular	Discharge by 1 sluice box and 2 D-shaped weirs	Overland flow (Minor ponding) Vegetation interaction	2000	≈ 1800	≈ 75	Approximately 80 percent cover by dead or dormant grasses and brush; *Phragmites communis* predominates
5	Richmond, Va.	James River, main channel and dock area at Deepwater Terminal (freshwater)	Disposal area on east bank of James River	Coarse sand and gravel, some light brown silt; dark grey silt on last day	James River (surface face freshwater)	70	≈ 35	Long and narrow	Divided into 3 equal square compartments; sluice box discharge	Overland flow Shallow ponding Cross-dikes	1100	≈ 700	≈ 20	Approximately 20 percent low density cover by forest and dormant undergrowth
6	Lake Charles, La.	Calcasieu River, main channel near northwest end of Lake Calcasieu (brackish water)	Disposal area No. 22, 10 river miles south of Lake Charles, on dredged material islands between the ship channel and lake	Dark grey to reddish-brown mixed silt, clay, and fine sand; some oily sediments	Calcasieu River (surface fresh to brackish)	185	≈ 150	Roughly rectangular	Pipe discharge,* and discharge over large rectangular weir**	Overland flow Ponding	300* 1300**	≈ 300	≈ 125	Sparse; less than 10 percent cover by large bushes and dead grasses
7	Seattle, Wash.	Duwamish Waterway, Slip No. 1 (brackish to marine)	Old wastewater treatment plant sludge lagoon area north of dredging site, 200 ft from waterway	Black silt-clay	Duwamish Waterway (surface brackish water)	1.9	1.9	Rectangular	Divided into 2 equal rectangular compartments, each 150 by 280 by 15.5 ft; effluent pumped from second compartment	Ponding Cross-dike	200	--	1.9	None
8	Vicksburg, Miss.	Brown Lake, WES, dredging concentrated at upper end of 23.5 acre lake, to a water depth of 10 to 14 ft	Adjacent to upper end of lake	Light grey silt with light brown crust	Durden Creek-Brown Lake (surface freshwater)	5	5	Rectangular	Divided into 2 equal compartments; sluice box discharge	Ponding Natural cross-dike	440	--	5	None
9	Southport, N. C.	Elizabeth River, in open channel at confluence with the Cape Fear estuary and near the Coast Guard Boat Harbor (brackish to marine)	East end of Oak Island, with dikes adjacent to Intracoastal Waterway	Black silt-clay (both sampling trips)	Atlantic Intracoastal Waterway (surface brackish water)	48	≈ 45	Elongated	D-shaped weir discharge	Overland flow Ponding Vegetation interaction	1500	≈ 700† ≈ 400††	≈ 5† ≈ 20††	15 acres of thick stand of trees and bushes in lower section; 15 acres of rushes and tall grass in scattered areas; 10 acres in southern section covered with virgin salt marsh vegetation, dominated by tall grass (*Spartina alterniflora*)

* Day 1.

** Days 2 and 3.

† Collection trip 1 (6-7 May).

†† Collection trip 2 (17-20 May).

Table 2

Descriptions of Compartments in Sayreville, N. J., Disposal Area

Compartment No.	Location Within Disposal Area	Percent of Total Area	Overland Flow Distance yd	Area Ponded acres	Percent of Area Covered by Vegetation	Percent of Area Covered by Stems*
1	Westernmost	19	700	--	77	7.0
2	Middle	31.5	--	15	65	18.9
3	Easternmost	49.5	--	20	0.05	8.2

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* Measured only for those areas having vegetation; based on random samples.

Table 3

Influent and Effluent Daily Pumping Rates and Samples Collected

	Compartment 1			Compartment 2	
	Influent		Effluent	Effluent	
Date of Sampling	Volume* gal	No. of Samples Collected	No. of Samples Collected	Volume gal	No. of Samples Collected
4 Mar 76	442,500				
5 Mar 76	371,000				
6 Mar 76	450,000				
7 Mar 76	--				
8 Mar 76	450,000				
9 Mar 76	480,000				
10 Mar 76	472,000				
11 Mar 76	450,000				
12 Mar 76	450,000				
13 Mar 76	450,000			100,000	
14 Mar 76	--			93,000	
15 Mar 76	472,500			65,000	
16 Mar 76	450,000	1		223,000	1
17 Mar 76	450,000			193,000	
18 Mar 76	450,000			46,000	
19 Mar 76	472,500	1		--	1
20 Mar 76	225,000			235,000	
21 Mar 76	225,000			530,000	
22 Mar 76	450,000	2		216,000	2
23 Mar 76	450,000	1		543,000	1
24 Mar 76	450,000			432,000	
25 Mar 76	405,000			432,000	
26 Mar 76	405,000			432,000	
27 Mar 76	450,000			432,000	
28 Mar 76	--			828,000	
29 Mar 76	450,000			624,000	
30 Mar 76	225,000			408,000	
31 Mar 76				696,000	
1 Apr 76				504,000	1
2 Apr 76				678,000	
3 Apr 76			1	810,000	1
4 Apr 76			1	378,000	1
5 Apr 76				--	1
6 Apr 76			1	432,000	1
7 Apr 76				504,000	

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* Calculated based on 750-gpm influent flow rate.

Table 4

Types of Samples Collected, Collection Dates, and Preparation Dates

Disposal Area Location	Sample Type	Collection Dates	Sample Filtration Dates
Sayreville, N. J.	Influent	2 Oct 75	1-5 Dec 75
	Effluent	2 Oct 75	1-5 Dec 75
	Background	2 Oct 75	1-5 Dec 75
Houston, Tex.	Influent	19 Nov 75	~ 15 Apr 76
	Effluent	19 Nov 75	~ 15 Apr 76
	Influent	20 Nov 75	~ 15 Apr 76
	Effluent	20 Nov 75	~ 15 Apr 76
	Influent	3 Dec 75	~ 15 Apr 76
	Effluent	3 Dec 75	~ 15 Apr 76
	Influent	4 Dec 75	~ 15 Apr 76
	Effluent	4 Dec 75	~ 15 Apr 76
	Background	2 Jan 76	~ 15 Apr 76
Grand Haven, Mich.	Influent	17 Dec 75	~ 15 Apr 76
	Effluent	17 Dec 75	~ 15 Apr 76
	Influent	18 Dec 75	~ 15 Apr 76
	Effluent	18 Dec 75	~ 15 Apr 76
	Influent	19 Dec 75	~ 15 Apr 76
	Effluent	19 Dec 75	~ 15 Apr 76
	Background	19 Dec 75	~ 15 Apr 76
Wilmington, N. C.	Background	15 Dec 75	14 Apr 76
	Influent	16 Dec 75	14 Apr 76
	Effluent	16 Dec 75	14 Apr 76
	Influent	22 Jan 76	2-4 Feb 76
	Effluent	22 Jan 76	2-4 Feb 76
	Background	22 Jan 76	2-4 Feb 76
	Influent	23 Jan 76	2-4 Feb 76
	Effluent	23 Jan 76	2-4 Feb 76
Richmond, Va.	Influent	27 Jan 76	17-18 Feb 76
	Effluent	27 Jan 76	17-18 Feb 76
	Background	27 Jan 76	17-18 Feb 76
	Influent	19 Feb 76	24-26 Feb 76
	Effluent	19 Feb 76	24-26 Feb 76
	Background	19 Feb 76	24-26 Feb 76
	Influent	20 Feb 76	24-26 Feb 76
	Effluent	20 Feb 76	24-26 Feb 76
	Influent	21 Feb 76	24-26 Feb 76
	Effluent	21 Feb 76	24-26 Feb 76

(Continued)

Table 4 (Concluded)

Disposal Area Location	Sample Type	Collection Dates	Sample Filtration Dates
Lake Charles, La.	Influent	4 Feb 76	18-19 Feb 76
	Effluent	4 Feb 76	18-19 Feb 76
	Background	4 Feb 76	23 Feb 76
	Influent	5 Feb 76	18-19 Feb 76
	Effluent	5 Feb 76	18-19 Feb 76
	Influent	5 Feb 76	18-19 Feb 76
	Effluent	5 Feb 76	18-19 Feb 76
Seattle, Wash.	Influent	16 Mar-23 Mar 76	Same as collection date
	Effluent	16 Mar- 6 Apr 76	Same as collection date
	Background	27 Mar-20 Apr 76	Same as collection date
Vicksburg, Miss.	Background	2 Mar 76	19 Apr 76
	Influent	17 Mar 76	18-22 Mar 76
	Effluent	17 Mar 76	18-22 Mar 76
	Effluent	12 Apr 76	21 Apr 76
	Influent	15 Apr 76	20 Apr 76
	Effluent	15 Apr 76	20 Apr 76
Southport, N. C.	Influent	6 May 76	12-14 May 76
	Effluent	6 May 76	12-14 May 76
	Background	6 May 76	12-14 May 76
	Influent	7 May 76	12-14 May 76
	Effluent	7 May 76	12-14 May 76
	Influent	17 May 76	25-27 May 76
	Effluent	17 May 76	25-27 May 76
	Background	17 May 76	25-27 May 76
	Influent	19 May 76	25-27 May 76
	Influent	20 May 76	25-27 May 76
	Effluent	20 May 76	25-27 May 76

Table 5

Physical and Chemical Analytical Methods and Instrumentation Used for the Characterization of Influent, Effluent, and Background Water Samples

Disposal Area	Total Solids	Nonfilterable Solids	Settleable Solids	Particle Size Analysis	
				Percent Sand, Silt, Clay	Clay-Size-Distribution
1. Sayreville, N. J.					Coulter Counter Model TA II, using 30-µm, 70-µm, or 100-µm aperture tubes (dependent on particle size range).
2. Houston, Tex.					Influents: Used "% Clay" fraction, dispersed with sodium hexametaphosphate and low frequency sonication. Effluents and Background water: Used total sample, dispersed by low-frequency sonication.
3. Grand Haven, Mich.	Gravimetric determination; brackish water samples, solids were centrifuged to approximately a 0.45-µm particle size, rinsed with deionized water, and recentrifuged to remove "dissolved solids"; *Std. Methods*, 13th ed., p. 286.[141]	Gravimetric Method; *EPA Manual* (1974),[142] p. 268.	Imhoff Cone; *Std. Methods*, 13th ed., p. 539.	Hydrometer Method; *Methods of Soil Analysis*, Part 2, p. 549.[86]	
4. Wilmington, N. C.	✓	✓	✓	✓	✓
5. Richmond, Va.	✓	✓	✓	✓	✓
6. Lake Charles, La.		✓	✓	✓	✓
8. Vicksburg, Miss.	✓	✓	✓	✓	✓
9. Southport, N. C.		✓	✓	✓	
7. Seattle, Wash.	✓	✓	✓		

(Continued)

✓ Denotes the disposal area for which samples were analyzed for a given parameter.

Table 5 (Continued)

Disposal Area	Metals (Ca, Mg, K, Na, Fe, Mn, Zn, Cd, Cu, Pb, Hg, Cr, Ti, V, As)	Chlorinated Pesticides, PCB's	Oil and Grease
1. Sayreville, N. J.	✓ A. Ca, Mg, K, Na: Perkin-Elmer Model 305B atomic absorption (AA) spectrophotometer with flame atomizer and deuterium background correction.	Hewlett-Packard Model 5750 gas chromatograph (GC) with Ni^{63} electron capture detector. Column: 1220 x 4 mm, packed with 5% QF-1 (Chromosorb W-HP, 80/100 mesh, Sargent Welsh). Carrier Gas: 95% argon, 5% methane. Reference Standards: DDT series, aldrin dieldrin, PCB's (single and multi-component).	✓ Petroleum ether extraction; gravimetric determination. *Std. Methods*, 13th ed., p. 254. Note: Solvent phase from separatory funnel was centrifuged for complete separation.
2. Houston, Tex.	✓ B. Fe, Mn, Cr: Direct sample injection into Perkin-Elmer Model 305B AA with heated graphite atomizer (HGA Model 2100) at 1250°C charring temperature. C. Zn, Cd, Cu, Ni, Pb, Ti, V, As: APDC-DDDC/MIBK extraction[143] and HGA analysis; As: with electrodeless discharge lamp (EDL) power supply.		✓
3. Grand Haven, Mich.	✓ D. Hg: Cold Vapor Method; Pye Unicam SP90-2 flameless AA. *Std. Methods*, 14th ed., p. 156.[144]	✓	✓
4. Wilmington, N. C.	✓ A. Ca, Mg, K, Na: Perkin-Elmer Model 306 AA spectrophotometer with flame atomizer and deuterium background correction. B. Fe, Mn, Zn, Ni (>0.03 ppm): Argon plasma emission spectrometer (Spectraspan 3).	✓	
5. Richmond, Va.	✓ C. Fe, Mn, Zn, Ni (<0.03 ppm): Cu, Pb: Perkin-Elmer Model 503 AA with HGA Model 2100. D. Cd, Hg: Zeeman effect AA spectrometer.	✓	
6. Lake Charles, La.	✓ E. Hg (large sample aliquot): Cold Vapor Method; Perkin-Elmer Model 306 flameless AA. *Std. Methods*, 14th ed., p. 156.	✓	
8. Vicksburg, Miss.	✓	✓	✓
9. Southport, N. C.	✓	✓	✓
7. Seattle, Wash.	✓ A. Perkin-Elmer Models 360 and 403 (all elements).	✓ Tracor Model 222 GC with linearized Ni^{63} electron capture detector. Confirmed on computerized Finnegan Model 3100 D GC-MS.	✓ Methylene chloride extraction; gravimetric determination. (procedure similar to *Std. Methods*, 13th ed., p. 254).

(Continued)

✓ Denotes the disposal area for which samples were analyzed for a given parameter.

Table 5 (Continued)

Disposal Area	Total Organic Carbon (Influent Solids)	Total and Organic Carbon (Effluents and Background <0.45 µm)	TKN (Organic N)	Ammonium
1. Sayreville, N. J.		Beckman Model 915A Total Organic Carbon (TOC) Analyzer with IR CO_2 detector.	✓ Digestion-acid titration after ammonium distillation; *Std. Methods*, 14th ed., p. 437.	✓ Sodium hydroxide distillation; Sulfuric acid titration; *Std. Methods*, 14th ed., pp. 168, 437.
2. Houston, Tex.	✓ Leco TC-12 automatic carbon analyzer with thermal conductivity detector.	✓	✓	✓
3. Grand Haven, Mich.		✓	✓	✓
4. Wilmington, N. C.	✓	✓ Dorhman/Envirotech TOC Analyzer with catalytic methane GC detector.	✓ High Solids: *Std. Methods*, 14th ed., p. 437.	✓ Technicon AA II; Phenate Method (colorimetric determination) *EPA Manual* (1974), p. 168.
5. Richmond, Va.	✓	✓ Beckman Model 915 TOC Analyzer with IR CO_2 detector.	✓ Low Solids: Technicon Auto Analyzer (AA) II digestion (Phenate Method); *EPA Manual* (1974), p. 168.	✓
6. Lake Charles, La.	✓	✓	✓	✓
8. Vicksburg, Miss.	✓	✓	✓	✓ Long-term site survey (Vicksburg, Miss.) *Std. Methods*, 14th ed., p. 437.
9. Southport, N. C.	✓	✓	✓	✓
7. Seattle, Wash.		✓ Beckman Model 915 TOC Analyzer with IR CO_2 detector.	✓	✓

(Continued)

✓ Denotes the disposal area for which samples were analyzed for a given parameter.

Table 5 (Continued)

Disposal Area	Nitrate and Nitrite N	Total P	Orthophosphate P	Total Sulfide	Sulfate	Chloride
1. Sayreville, N. J.	✓ Manual Cadmium Reduction Method. <u>Std. Methods</u>, 14th ed., p. 423.	✓ Manual acid digestion; Modified Ascorbic Acid Method.[145]		✓ Zinc acetate pretreatment; sulfuric acid acidification; iodometric titration, under N_2; <u>Std. Methods</u>, 13th ed., p. 551.	✓ Turbidimetric Method; <u>Std. Methods</u>, 14th ed., p. 496.	✓ Titrimetric method with mercuric nitrate and diphenylcarbazone indicator; <u>Std. Methods</u>, 14th ed., p. 304.
2. Houston, Tex.	✓	✓				
3. Grand Haven, Mich.	✓	✓		✓ Low-sulfide samples; Methylene Blue Colorimetric Method; <u>Std. Methods</u>, 14th ed., p. 503.	✓	✓
4. Wilmington, N. C.	✓ Technicon AA II; Automated Cadmium Reduction Method; <u>EPA Manual</u> (1974) p. 207.	✓ Manual digestion; Ascorbic Acid Method for high-solids samples; <u>Std. Methods</u>, 14th ed., p. 481.	✓ Manual Ascorbic Acid Method; <u>Std. Methods</u>, 14th ed., p. 481.	✓ Zinc acetate pretreatment, sulfuric acid acidification; iodometric titration, under N_2; <u>Std. Methods</u>, 13th ed., p. 551.		✓
5. Richmond, Va.	✓	✓	✓	✓		✓
6. Lake Charles, La.	✓	✓ Automated Technicon AA II digestion, Ascorbic Acid Method for low-solids samples; <u>EPA Manual</u> (1974) p. 256.	✓ Automated Technicon AA II Ascorbic Acid Method; <u>EPA Manual</u> (1974), p. 256.	✓		✓
8. Vicksburg, Miss.	✓	✓	✓	✓		✓
9. Southport, N. C.	✓	✓	✓	✓		✓
7. Seattle, Wash.	✓	✓	✓	✓ <u>Std. Methods</u>, 13th ed., p. 551. Soluble sulfide: Orion sulfide electrode, Gram's plot with cadmium nitrate	✓	✓

(Continued)

✓ Denotes the disposal area for which samples were analyzed for a given parameter.

Table 5 (Concluded)

Disposal Area	Alkalinity	Chemical Oxygen Demand	Cation Exchange Capacity and Exchangeable Ammonium N	Volatile Solids
1. Sayreville, N. J.	✓ Potentiometric titration; Std. Methods, 14th ed., p. 278.	Potassium dichromate oxidation; standard ferrous ammonium sulfate titration; EPA Manual (1974) p. 25.	CEC: Sodium Saturation Method, using sodium acetate; sodium leach with ammonium chloride; measurement of sodium in extract; Methods of Soil Analysis, Part 2, p. 899. Exchangeable Ammonium: Sodium Saturation Method, using sodium acetate to displace ammonium from exchange sites; Methods of Soil Analysis, Part 2, p. 899.	Ignition of solids at 550°C in a muffle furnace; EPA Manual (1974), p. 272.
2. Houston, Tex.	✓			
3. Grand Haven, Mich.	✓			
4. Wilmington, N. C.	✓		✓	
5. Richmond, Va.	✓		✓	
6. Lake Charles, La.	✓		✓	
8. Vicksburg, Miss.	✓		✓	
9. Southport, N. C.	✓		✓	
7. Seattle, Wash.	✓	✓	✓	✓

✓ Denotes the disposal area for which samples were analyzed for a given parameter.

(Sheet 5 of 5)

Table 6

Average Values for Field Data of Influents, Effluents, and Background Water from Nine Confined Land Disposal Areas

Location	No. of Samples	Water Temperature °C	Salinity °/oo	Conductivity mmho/cm	Dissolved O_2, mg/ℓ	pH
Sayreville, N. J.						
Influent	1	--	12.0	15.3	--	--
Effluent	1	17.5	14.0	17.6	6.3	--
Background	1	--	9.0	10.5	--	--
Houston, Tex.						
Influent	4	21.0	7.95	--	--	7.1
Effluent	4	19.5	8.9	--	7.85	7.7
Background	1	--	1.7	3.25	--	6.9
Grand Haven, Mich.						
Influent	4	1.5	0.7	0.52	--	--
Effluent	4	0.75	0.6	0.41	12.5	7.5
Background	1	3.5	--	0.39	11.5	7.6
Wilmington, N. C.						
Influent	3	8.5	2.8	3.2	2.3	6.6
Effluent	3	9.9	2.6	4.05	2.2	7.4
Background	2	--	--	--	--	5.5
Richmond, Va.						
Influent	4	10.6	0.25	0.29	7.75	6.7
Effluent	4	9.8	1.25	1.61	8.6	7.2
Background	2	--	0	0.18	--	6.7
Lake Charles, La.						
Influent	3	14.0	15.9	20.8	4.9	7.15
Effluent	3	17.3	18.6	25.3	3.7	7.25
Background	1	--	4.2	6.8	--	6.3
Seattle, Wash.						
Effluent, Pond 1	7*	7.1	18.5	30.1	5.4	5.8
Effluent, Pond 2	18	11.2	23.0	36.2	6.5	7.7
Background, surface	3	7.8	8.0	14.1	8.3	7.3
Background, 10 m	3	7.7	26.5	41.8	7.5	7.75
Vicksburg, Miss.						
Influent	2	18.5	0.15	0.65	3.0	7.0
Effluent	3	26.1	0.1	0.70	2.4	7.7
Background	1	--	--	--	--	--
Southport, N. C. (1)						
Influent	3	21.2	20.0	30.0	2.75	7.9
Effluent	3	24.0	21.3	32.0	3.3	7.65
Background	1	--	21.0	31.5	--	--
Southport, N. C. (2)						
Influent	3	22.4	15.2	23.7	2.8	7.55
Effluent	3	30.4	20.9	38.05	6.1	7.9
Background	1	--	--	--	--	--

* Daily composites.

Table 7

Average Values for Physical Parameters of Influent, Effluent, and Background Water Samples from Nine Confined Land Disposal Areas

Location	No. of Samples	Mechanical Particle Size, percent			Coulter Counter Particle Size, μm		Total Solids percent Weight	Nonfilterable Solids percent Weight	Settleable Solids mℓ/ℓ	Cation Exchange Capacity meq/100 g	Alkalinity mg/ℓ
		Clay (<2 μm)	Silt (2-50 μm)	Sand (>50 μm)	>50 percent	>80 percent					
Sayreville, N. J.											
Influent	1	--	--	--	--	--	--	--	--	--	680
Effluent	1	--	--	--	--	--	--	--	--	--	840
Background	1	--	--	--	--	--	--	--	--	--	500
Houston, Tex.											
Influent	4	--	--	--	--	--	--	--	--	--	524
Effluent	4	--	--	--	--	--	--	--	--	--	517
Background	1	--	--	--	--	--	--	--	--	--	92
Grand Haven, Mich.											
Influent	4	--	--	--	--	--	--	--	--	--	240
Effluent	4	--	--	--	--	--	--	--	--	--	218
Background	1	--	--	--	--	--	--	--	--	--	140
Wilmington, N. C.											
Influent	3	60	31	9	1.05	0.6	6.66	7.190	467	80.6	345.1
Effluent	3	61	35	4	2.0	1.2	2.34	0.901	417	120.6	480.3
Background	2	--	--	--	--	--	--	0.00448	<0.1	--	17.27
Richmond, Va.											
Influent	4	11	24	65	1.15	0.63	3.49	1.660	85	26.2	82.30
Effluent	4	--	--	--	3.75	1.9	0.0626	0.0464	0.7	65.9	60.50
Background	2	--	--	--	7.6	3.8	0.0114	0.0010	<0.1	--	40.77
Lake Charles, La.											
Influent	3	64.5	29.5	6	0.91	0.56	--	6.26	367	56.8	320.8
Effluent	3	58	30	12	3.1	1.65	--	1.218	120	66.2	199.2
Background	1	--	--	--	5.4	2.82	--	0.0036	<0.1	--	29.95
Seattle, Wash.											
Influent	5	--	--	--	--	--	6.7	--	350	70.0	348
Effluent, Pond 1	3	--	--	--	--	--	0.01917	0.02	0.6	--	183
Effluent, Pond 2	10	--	--	--	--	--	0.00857	<0.01	0.25	88.0	199
Background	7	--	--	--	--	--	--	--	--	--	--
Vicksburg, Miss.											
Influent	2	22.5	72.5	5	0.89	0.56	23.50	23.30	950	10.3	285.5
Effluent	3	--	--	--	8.9	3.4	0.297	0.248	13.5	--	237.3
Background	1	--	--	--	5.5	2.85	0.0428	0.00204	<0.1	--	290.0
Southport, N. C. (1)											
Influent	3	49	26	25	--	--	--	18.50	500	55.0	512
Effluent	3	--	--	--	--	--	--	0.0152	<0.1	--	178
Background	1	--	--	--	--	--	--	0.0094	<0.1	--	101
Southport, N. C. (2)											
Influent	3	41.5	50	8.5	--	--	--	17.10	693	57.9	1024
Effluent	3	--	--	--	--	--	--	0.1715	1.9	--	631
Background	1	--	--	--	--	--	--	0.02697	<0.1	--	112

Table 8

Average Values for Chemical Parameters of Influent, Effluent, and Background Water Samples from Nine Confined Land Disposal Areas

Location	No. of Samples	Total Carbon <0.45 µm	Organic Carbon			Oil and Grease		
			Total mg/ℓ	Solids mg/kg	<0.45 µm mg/ℓ	Total mg/ℓ	Solids mg/kg	<0.45 µm mg/ℓ
Sayreville, N. J.								
Influent	1	--	--	--	--	8.4	--	5.5
Effluent	1	--	--	--	--	12.2	--	--
Background	1	--	--	--	--	9.2	--	--
Houston, Tex.								
Influent	4	--	--	17,190	107	618.0	--	--
Effluent	4	--	--	29,820	68	73.5	--	--
Background	1	--	--	--	43	47.2	--	1.1
Grand Haven, Mich.								
Influent	4	57	--	--	28	39.5	--	2.5
Effluent	4	65	--	--	28	11.5	--	2.0
Background	1	58	--	--	31	6.4	--	1.1
Wilmington, N. C.								
Influent	3	--	3330	47,630	24	--	--	--
Effluent	3	--	475	51,700	25	--	--	--
Background	2	--	14	--	12	--	--	--
Richmond, Va.								
Influent	4	--	480	21,650	10	--	--	--
Effluent	4	--	10	--	6	--	--	--
Background	2	--	9	--	5	--	--	--
Lake Charles, La.								
Influent	3	--	993	14,550	11	--	--	--
Effluent	3	--	209	12,300	7	--	--	--
Background	1	--	12	--	9	--	--	--
Seattle, Wash.								
Influent	5	--	--	--	11	582	6060	21.3
Effluent, Pond 1	3	--	--	--	8	89	--	5.0
Effluent, Pond 2	10	--	--	--	10	23.7	--	5.25
Background	7	--	3.5	--	--	0.125	--	--
Vicksburg, Miss.								
Influent	2	--	1820	7,450	14	--	--	5.0
Effluent	3	--	16	--	8	3.9	--	--
Background	1	--	10	--	6	--	--	--
Southport, N. C. (1)								
Influent	3	--	5230	32,000	8	--	--	2.4
Effluent	3	--	5	--	7	5.9	--	--
Background	1	--	4	--	8	15.6	--	--
Southport, N. C. (2)								
Influent	3	--	6060	37,630	11	--	--	--
Effluent	3	--	31	--	11	--	--	--
Background	1	--	--	--	23	1.2	--	--

(Continued)

(Sheet 1 of 12)

Table 8 (Continued)

Location	No. of Samples	Total Chlorinated Pesticides, mg/ℓ*						Total PCB's mg/ℓ
		op' DDE	pp' DDE	op' DDD	pp' DDD	op' DDT	pp' DDT	
Sayreville, N. J.								
Influent	1	--	--	--	--	--	--	--
Effluent	1	--	--	--	--	--	--	--
Background	1	--	--	--	--	--	--	--
Houston, Tex.								
Influent	4	--	--	--	--	--	--	--
Effluent	4	--	--	--	--	--	--	--
Background	1	--	--	--	--	--	--	--
Grand Haven, Mich.								
Influent	4	0.10	0.27	0.23	0.52	0.75	0.71	10.67
Effluent	4	0.07	0.21	<0.01	<0.01	0.12	0.48	2.55
Background	1	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.1
Wilmington, N. C.								
Influent	3	0.08	0.35	<0.01	<0.01	<0.01	<0.01	1.28
Effluent	3	0.10	0.24	<0.01	<0.01	<0.01	<0.01	0.72
Background	2	<0.01	0.01	<0.01	<0.01	<0.01	<0.01	<0.1
Richmond, Va.								
Influent	4	0.26	0.77	<0.01	<0.01	<0.01	<0.01	4.75
Effluent	4	0.18	0.32	<0.01	<0.01	<0.01	<0.01	0.44
Background	2	0.08	0.24	<0.01	<0.01	<0.01	<0.01	<0.1
Lake Charles, La.								
Influent	3	0.13	0.67	<0.01	<0.01	<0.01	<0.01	4.98
Effluent	3	<0.01	0.96	<0.01	<0.01	<0.01	<0.01	<0.1
Background	1	0.28	1.57	<0.01	<0.01	<0.01	<0.01	0.30
Seattle, Wash.								
Influent	5	--	--	--	--	--	--	5.23
Effluent, Pond 1	3	--	--	--	--	--	--	0.0077
Effluent, Pond 2	10	--	--	--	--	--	--	0.00063
Background	7	--	--	--	--	--	--	0.00001
Vicksburg, Miss.								
Influent	2	0.17	0.92	<0.01	<0.01	<0.01	<0.01	8.4
Effluent	3	<0.01	0.02	<0.01	<0.01	<0.01	<0.01	<0.1
Background	1	--	--	--	--	--	--	--
Southport, N. C. (1)								
Influent	3	--	--	--	--	--	--	--
Effluent	3	--	--	--	--	--	--	--
Background	1	--	--	--	--	--	--	--
Southport, N. C. (2)								
Influent	3	0.016	0.019	0.76	0.65	4.71	4.02	5.88
Effluent	3	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.1
Background	1	--	--	--	--	--	--	--

(Continued)

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* Dieldrin, aldrin, lindane, heptachlor, heptachlor epoxide, and chlordane concentrations were also determined and all were below detection limits.

Table 8 (Continued)

Location	No. of Samples	Organic Nitrogen Total mg/ℓ	Organic Nitrogen Solids mg/kg	Organic Nitrogen <0.45 μm mg/ℓ	Ammonium-Nitrogen Total mg/ℓ	Ammonium-Nitrogen Exchangeable mg/kg	Ammonium-Nitrogen <0.45 μm mg/ℓ	Nitrate + Nitrite-N mg/ℓ <0.45 μm
Sayreville, N. J.								
Influent	1	<1.0	--	--	1.51	--	<1.0	0.28
Effluent	1	2.55	--	--	22.4	--	16.8	0.30
Background	1	1.6	--	--	1.97	--	<1.0	0.22
Houston, Tex.								
Influent	4	45.9	--	7.2	57.1	--	50.6	0.19
Effluent	4	6.4	--	1.1	53.8	--	49.7	0.22
Background	1	0.38	--	0.19	--	--	0.20	0.68
Grand Haven, Mich.								
Influent	4	59.7	--	2.8	30.2	--	20.5	0.59
Effluent	4	27.6	--	2.7	26.3	--	12.7	1.33
Background	1	0.60	--	0.65	<0.2	--	<0.2	1.32
Wilmington, N. C.								
Influent	3	242	3254	1.95	--	252	23.8	<0.01
Effluent	3	27.8	2878	1.95	11.4	458	22.6	0.28
Background	2	0.2	--	0.3	0.88		0.62	0.17
Richmond, Va.								
Influent	4	37.9	1765	1.6	--	145	9.00	0.30
Effluent	4	1.5	--	1.2	5.59	419	4.78	0.11
Background	2	0.2	--	<0.25	1.39	--	<0.05	<0.01
Lake Charles, La.								
Influent	3	82.3	1228	1.5	--	69.2	19.5	<0.01
Effluent	3	16.9	906	2.9	5.85	66.7	10.3	0.03
Background	1	0.2	--	0.9	1.15	--	0.41	0.04
Seattle, Wash.								
Influent	5	68	1116	3.0	--	10	9	0.27
Effluent, Pond 1	3	--	--	5.1	--	--	5.1	0.36
Effluent, Pond 2	10	0.2	--	0.4	--	122	6.6	0.35
Background	7	0.08	--	--	0.24	--	--	0.425
Vicksburg, Miss.								
Influent	2	157	660	1.9	--	108	6.95	0.36
Effluent	3	3.25	--	2.65	7.74	--	4.08	0.52
Background	1	0.45	--	0.45	0.36	--	0.54	1.98
Southport, N. C. (1)								
Influent	3	417	2340	1.75	--	45.3	13.9	0.014
Effluent	3	1.0	--	0.78	1.78	--	1.25	0.054
Background	1	2.35	--	0.9	<0.01	--	<0.01	<0.010
Southport, N. C. (2)								
Influent	3	483	2895	14.8	--	119.0	29.3	--
Effluent	3	4.7	--	4.6	17.7	119.0	14.1	--
Background	1	0.8	--	1.35	0.55	--	<0.01	--

(Continued)

Table 8 (Continued)

Location	No. of Samples	Orthophosphate-P, mg/ℓ <0.45 µm	Total Phosphorus			Total Sulfides	
			Total mg/ℓ	Solids mg/kg	<0.45 µm mg/ℓ	Total mg/ℓ	Solids mg/kg
Sayreville, N. J.							
Influent	1	--	1.74	--	0.10	--	--
Effluent	1	--	1.84	--	0.10	--	--
Background	1	--	2.81	--	0.13	--	--
Houston, Tex.							
Influent	4	--	189	--	0.25	--	--
Effluent	4	--	48.3	--	1.05	--	--
Background	1	--	0.86	--	0.50	--	--
Grand Haven, Mich.							
Influent	4	--	24.9	--	0.06	--	--
Effluent	4	--	8.66	--	<0.02	--	--
Background	1	--	0.76	--	<0.02	--	--
Wilmington, N. C.							
Influent	3	0.22	309	4285	0.26	3.8	32.6
Effluent	3	0.11	35.4	3900	0.14	2.6	148
Background	2	0.09	0.35	--	0.10	2.0	--
Richmond, Va.							
Influent	4	0.06	49.5	2650	0.08	3.0	62.6
Effluent	4	0.09	1.95	--	0.20	1.7	--
Background	2	0.11	0.33	--	0.15	1.9	--
Lake Charles, La.							
Influent	3	0.10	95.1	1400	0.14	26.3	317
Effluent	3	0.09	23.0	1400	0.1	18.4	327
Background	1	0.06	0.14	--	0.06	--	--
Seattle, Wash.							
Influent	5	0.37	94.5	1418	0.40	54	1128
Effluent, Pond 1	3	0.29	0.6	--	0.30	--	--
Effluent, Pond 2	10	0.05	0.25	--	0.05	<0.02	--
Background	7	--	0.125	--	--	<0.02	--
Vicksburg, Miss.							
Influent	2	0.09	347	1772	0.11	10.2	37.4
Effluent	3	0.09	3.15	--	0.12	1.2	--
Background	1	0.15	0.15	--	0.15	1.4	--
Southport, N. C. (1)							
Influent	3	0.41	231	1105	<0.5	102	529
Effluent	3	0.17	0.40	--	--	1.6	--
Background	1	<0.03	0.07	--	<0.5	0.8	--
Southport, N. C. (2)							
Influent	3	4.31	129	818	6.12	122	782
Effluent	3	1.00	1.85	--	1.30	2.7	--
Background	1	0.03	0.18	--	0.10	0.2	--

(Continued)

Note: All solids values are expressed as mg/kg dry weight.

(Sheet 4 of 12)

Table 8 (Continued)

Location	No. of Samples	Calcium			Magnesium		
		Total mg/ℓ	Solids mg/kg	<0.45 µm mg/ℓ	Total mg/ℓ	Solids mg/kg	<0.45 µm mg/ℓ
Sayreville, N. J.							
Influent	1	--	--	--	--	--	--
Effluent	1	--	--	--	--	--	--
Background	1	--	--	--	--	--	--
Houston, Tex.							
Influent	4	2070	--	148	1550	--	314
Effluent	4	232	--	153	603	--	317
Background	1	42.5	--	35.3	53.6	--	32.9
Grand Haven, Mich.							
Influent	4	107.3	--	59.0	284	--	169.5
Effluent	4	80.7	--	54.7	68.1	--	15.5
Background	1	56.2	--	42.6	23.0	--	13.9
Wilmington, N. C.							
Influent	3	477	3,695	174	683	7,900	90
Effluent	3	404	2,895	384	163	5,635	120
Background	2	6.9	--	7.6	7.9	--	6.3
Richmond, Va.							
Influent	4	67.3	4,205	12.3	91.4	4,720	3.3
Effluent	4	54.8	4,850	17.4	7.6	7,080	4.3
Background	2	15.3	--	12.0	3.5	--	2.9
Lake Charles, La.							
Influent	3	389	2,215	260	968	9,130	417
Effluent	3	310	3,695	250	731	13,415	497
Background	1	55	--	49	159	--	150
Seattle, Wash.							
Influent	5	--	18,200	--	--	20,900	--
Effluent, Pond 1	3	--	--	--	--	--	--
Effluent, Pond 2	10	--	12,600	--	--	14,500	--
Background	7	--	--	--	--	--	--
Vicksburg, Miss.							
Influent	2	4980	17,250	62	1685	10,840	29.5
Effluent	3	111	27,750	54	60.4	12,000	25
Background	1	59	--	68	31.4	--	33
Southport, N. C. (1)							
Influent	3	6270	32,440	365	2450	9,070	1135
Effluent	3	385	--	375	1165	--	1100
Background	1	340	--	330	1200	--	1100
Southport, N. C. (2)							
Influent	3	6225	35,200	340	2447	9,150	1065
Effluent	3	423	--	370	870	--	1135
Background	1	390	--	390	1300	--	1300

(Continued)

Table 8 (Continued)

		Potassium			Sodium			
Location	No. of Samples	Total mg/ℓ	Solids mg/kg	<0.45 µm mg/ℓ	Total mg/ℓ	Solids mg/kg	<0.45 µm mg/ℓ	Chloride mg/ℓ
Sayreville, N. J.								
Influent	1	--	--	--	--	--	--	6,220
Effluent	1	--	--	--	--	--	--	7,500
Background	1	--	--	--	--	--	--	4,080
Houston, Tex.								
Influent	4	1485	--	142	2825	--	2440	5,105
Effluent	4	970	--	155	2715	--	2445	5,080
Background	1	144	--	31.6	1465	--	880	1,040
Grand Haven, Mich.								
Influent	4	1499	--	154	152	--	92.9	150
Effluent	4	588	--	141	123	--	50.7	80
Background	1	338	--	142	16.8	--	9.8	21
Wilmington, N. C.								
Influent	3	798	10,305	41.8	732	5,205	832	1,170
Effluent	3	130	8,805	54.6	1045	30,910	909	1,317
Background	2	5.6	--	6.2	246	--	10	120
Richmond, Va.								
Influent	4	306	18,800	2.6	105.5	6,670	7.2	8
Effluent	4	12.2	18,200	2.9	9.6	240	6.0	20
Background	2	2.9	--	1.3	6.9	--	6.2	3
Lake Charles, La.								
Influent	3	1045	13,500	200	6885	--	7000	13,870
Effluent	3	488	16,200	197	6070	--	7100	11,870
Background	1	64.3	--	39	--	--	1400	2,080
Seattle, Wash.								
Influent	5	--	3,300	--	--	11,500	--	16,100
Effluent, Pond 1	3	--	--	--	--	--	--	15,650
Effluent, Pond 2	10	--	10,500	--	--	61,800	--	12,500
Background	7	--	--	--	--	--	--	--
Vicksburg, Miss.								
Influent	2	6120	30,900	6.1	2520	11,250	65	85
Effluent	3	45.5	15,200	9.0	131	28,000	58	72
Background	1	5	--	9.0	--	--	32	35
Southport, N. C. (1)								
Influent	3	2245	10,570	405	9850	--	9235	16,030
Effluent	3	355	--	430	8800	--	8465	14,570
Background	1	320	--	370	8000	--	7600	20,500
Southport, N. C. (2)								
Influent	3	2190	10,995	313	8160	--	8500	15,170
Effluent	3	340	--	340	9710	--	9265	16,230
Background	1	320	--	380	9800	--	9700	17,600

(Continued)

Table 8 (Continued)

Location	No. of Samples	Iron			Manganese		
		Total mg/ℓ	Solids mg/kg	<0.45 μm mg/ℓ	Total mg/ℓ	Solids mg/kg	<0.45 μm mg/ℓ
Sayreville, N. J.							
Influent	1	--	--	0.057	--	--	0.0039
Effluent	1	28.2	--	0.074	23.6	--	0.0038
Background	1	--	--	0.073	--	--	0.0043
Houston, Tex.							
Influent	4	917	--	0.062	42.9	--	0.0042
Effluent	4	800	--	0.055	39.6	--	0.0034
Background	1	63.6	--	0.006	0.07	--	0.030
Grand Haven, Mich.							
Influent	4	54.8	--	0.101	1.04	--	0.152
Effluent	4	40.9	--	0.147	0.39	--	0.029
Background	1	5.21	--	0.059	0.25	--	0.009
Wilmington, N. C.							
Influent	3	3916	52,500	12.4	68.8	877	4.42
Effluent	3	364	42,500	4.5	10.9	811	4.51
Background	2	4.2	--	0.27	0.12	--	0.043
Richmond, Va.							
Influent	4	786	40,400	1.25	19.5	1002	0.620
Effluent	4	35.6	24,100	0.24	1.05	1195	0.280
Background	2	1.2	--	0.34	0.05	--	0.064
Lake Charles, La.							
Influent	3	2089	33,300	8.60	55.2	726	10.4
Effluent	3	480	39,600	1.56	18.1	1540	6.43
Background	1	1.29	--	0.20	0.192	--	0.184
Seattle, Wash.							
Influent	5	2980	46,470	0.275	29.7	427	0.197
Effluent, Pond 1	3	1.97	--	0.19	0.337	--	0.256
Effluent, Pond 2	10	4.25	41,300	0.24	0.96	928	0.81
Background	7	0.44	--	--	0.060	--	--
Vicksburg, Miss.							
Influent	2	9840	52,950	10.5	283.4	1450	3.16
Effluent	3	93.9	39,400	0.013	3.13	1135	0.81
Background	1	1.84	--	<0.001	0.403	--	0.016
Southport, N. C. (1)							
Influent	3	6510	36,110	3.11	98.2	511	2.55
Effluent	3	1.45	--	1.28	0.627	--	0.618
Background	1	1.35	--	1.43	0.080	--	0.082
Southport, N. C. (2)							
Influent	3	6780	40,330	1.48	75.4	426	2.45
Effluent	3	84.0	--	0.855	2.05	--	1.58
Background	1	1.12	--	0.881	0.074	--	0.054

(Continued)

Table 8 (Continued)

		Zinc			Cadmium		
Location	No. of Samples	Total mg/ℓ	Solids mg/kg	<0.45 µm mg/ℓ	Total mg/ℓ	Solids mg/kg	<0.45 µm mg/ℓ
Sayreville, N. J.							
Influent	1	--	--	0.0079	--	--	0.00046
Effluent	1	1.01	--	0.0079	0.75	--	0.00039
Background	1	--	--	0.0065	--	--	0.00035
Houston, Tex.							
Influent	4	--	--	--	1.15	--	0.00058
Effluent	4	--	--	--	0.11	--	0.00033
Background	1	0.016	--	0.0007	0.005	--	0.00035
Grand Haven, Mich.							
Influent	4	36.9	--	0.0079	6.04	--	0.00075
Effluent	4	1.69	--	0.0081	0.34	--	0.00045
Background	1	0.305	--	0.0040	0.0034	--	0.0003
Wilmington, N. C.							
Influent	3	13.9	191	0.040	0.0090	0.11	0.0023
Effluent	3	1.7	165	0.090	0.0142	0.78	0.0009
Background	2	0.13	--	0.017	0.0017	--	0.0006
Richmond, Va.							
Influent	4	5.2	249	0.005	0.0146	0.78	0.0020
Effluent	4	0.29	210	0.017	0.0051	4.86	0.0014
Background	2	0.05	--	0.007	--	--	0.0026
Lake Charles, La.							
Influent	3	6.56	105	0.002	0.010	0.122	0.007
Effluent	3	1.73	136	0.002	0.003	0.046	0.003
Background	1	0.053	--	0.005	--	--	0.0011
Seattle, Wash.							
Influent	5	68.7	936	0.007	0.41	5.81	<0.002
Effluent, Pond 1	3	0.104	--	0.026	0.002	--	<0.002
Effluent, Pond 2	10	0.237	1.30	0.108	0.004	6.83	0.002
Background	7	0.01	--	--	<0.002	--	--
Vicksburg, Miss.							
Influent	2	31.4	168	0.073	0.027	8.1	0.0002
Effluent	3	2.66	1385	0.004	<0.002	--	0.0002
Background	1	0.74	--	0.010	<0.0002	--	--
Southport, N. C. (1)							
Influent	3	14.4	87.3	0.229	0.59	4.2	0.0110
Effluent	3	1.27	--	0.099	0.0095	--	0.0099
Background	1	1.22	--	0.093	0.0096	--	0.0098
Southport, N. C. (2)							
Influent	3	17.9	117.8	0.109	3.18	20.5	0.0081
Effluent	3	0.197	--	0.108	0.014	--	0.0101
Background	1	1.28	--	0.121	0.0100	--	0.0120

(Continued)

Table 8 (Continued)

Location	No. of Samples	Copper Total mg/ℓ	Copper Solids mg/kg	Copper <0.45 μm mg/ℓ
Sayreville, N. J.				
Influent	1	--	--	0.0065
Effluent	1	3.38	--	0.0065
Background	1	--	--	0.0057
Houston, Tex.				
Influent	4	12.75	--	0.0055
Effluent	4	1.20	--	0.0050
Background	1	0.160	--	0.0035
Grand Haven, Mich.				
Influent	4	11.8	--	0.0041
Effluent	4	0.50	--	0.0045
Background	1	0.052	--	0.0052
Wilmington, N. C.				
Influent	3	2.61	37.1	0.004
Effluent	3	0.290	34.0	0.012
Background	2	0.011	--	0.003
Richmond, Va.				
Influent	4	0.93	43.5	0.004
Effluent	4	0.08	131	0.006
Background	2	0.07	--	0.005
Lake Charles, La.				
Influent	3	1.79	28.5	0.0025
Effluent	3	0.39	30.2	0.010
Background	1	0.089	--	0.006
Seattle, Wash.				
Influent	5	9.05	135	0.052
Effluent, Pond 1	3	0.077	--	0.057
Effluent, Pond 2	10	0.053	198	0.042
Background	7	0.019	--	--
Vicksburg, Miss.				
Influent	2	5.67	26.4	0.002
Effluent	3	0.092	36.9	0.004
Background	1	--	--	0.006
Southport, N. C. (1)				
Influent	3	1.92	12.3	0.014
Effluent	3	0.025	--	0.013
Background	1	0.026	--	0.026
Southport, N. C. (2)				
Influent	3	4.05	27.9	0.061
Effluent	3	0.083	--	0.055
Background	1	0.024	--	0.028

(Continued)

Table 8 (Continued)

Location	No. of Samples	Nickel			Lead		
		Total mg/ℓ	Solids mg/kg	<0.45 µm mg/ℓ	Total mg/ℓ	Solids mg/kg	<0.45 µm mg/ℓ
Sayreville, N. J.							
Influent	1	--	--	0.0028	--	--	0.0037
Effluent	1	0.34	--	0.0053	3.38	--	0.0047
Background	1	--	--	0.0037	--	--	0.0035
Houston, Tex.							
Influent	4	7.05	--	0.0059	69.2	--	0.0057
Effluent	4	0.58	--	0.0049	6.41	--	0.0052
Background	1	1.50	--	0.0060	0.018	--	0.0002
Grand Haven, Mich.							
Influent	4	10.4	--	0.007	15.3	--	0.0003
Effluent	4	0.39	--	0.005	0.91	--	0.0003
Background	1	0.013	--	0.004	0.049	--	0.0002
Wilmington, N. C.							
Influent	3	3.56	50.0	0.004	4.4	60.1	0.004
Effluent	3	0.53	45.9	0.005	0.60	47.0	<0.001
Background	2	0.017	--	0.006	0.012	--	0.002
Richmond, Va.							
Influent	4	1.00	46.1	<0.003	1.59	78.8	0.001
Effluent	4	0.046	42.0	0.003	0.087	142	<0.001
Background	2	0.018	--	0.005	0.003	--	<0.001
Lake Charles, La.							
Influent	3	1.98	20.5	0.006	2.12	33.5	<0.001
Effluent	3	0.95	34.7	0.005	0.472	37.9	<0.001
Background	1	0.011	--	0.003	0.012	--	0.005
Seattle, Wash.							
Influent	5	3.85	61.8	0.02	13.5	201	--
Effluent, Pond 1	3	--	--	--	--	--	--
Effluent, Pond 2	10	0.02	25.8	0.02	--	135	--
Background	7	<0.01	--	--	--	--	--
Vicksburg, Miss.							
Influent	2	16.4	84.5	<0.003	11.6	68.6	<0.001
Effluent	3	0.121	57.9	<0.003	0.06	12.2	<0.001
Background	1	0.018	--	0.004	<0.001	--	<0.001
Southport, N. C. (1)							
Influent	3	6.1	39.6	0.033	4.5	24.6	<0.001
Effluent	3	0.15	--	0.037	0.039	--	0.001
Background	1	--	--	0.033	0.002	--	0.001
Southport, N. C. (2)							
Influent	3	6.4	38.2	0.033	0.74	10.5	0.001
Effluent	3	0.079	--	0.027	0.015	--	0.003
Background	1	--	--	0.036	0.002	--	0.003

(Continued)

Table 8 (Continued)

Location	No. of Samples	Mercury			Chromium	
		Total mg/ℓ	Solids mg/kg	<0.45 µm mg/ℓ	Total mg/ℓ	<0.45 µm mg/ℓ
Sayreville, N. J.						
Influent	1	0.0099	--	0.0007	--	0.0032
Effluent	1	0.0108	--	0.0009	0.34	0.0038
Background	1	0.0163	--	0.0007	--	0.0029
Houston, Tex.						
Influent	4	--	--	--	--	--
Effluent	4	--	--	--	--	--
Background	1	--	--	--	--	--
Grand Haven, Mich.						
Influent	4	--	--	--	63.8	0.004
Effluent	4	--	--	--	0.26	0.005
Background	1	--	--	--	0.013	0.003
Wilmington, N. C.						
Influent	3	0.042	0.44	0.003	--	--
Effluent	3	0.185	0.08	0.002	--	--
Background	2	0.0006	--	0.002	--	--
Richmond, Va.						
Influent	4	0.0024	0.18	<0.0002	--	--
Effluent	4	0.0002	0.10	<0.0002	--	--
Background	2	0.0001	--	<0.0002	--	--
Lake Charles, La.						
Influent	3	0.0063	0.10	<0.0002	--	--
Effluent	3	0.0022	0.17	<0.0002	--	--
Background	1	0.0030	--	<0.0002	--	--
Seattle, Wash.						
Influent	5	0.040	0.64	<0.0002	--	--
Effluent, Pond 1	3	0.0005	--	0.0002	0.036	0.025
Effluent, Pond 2	10	0.00015	--	0.0001	0.028	0.025
Background	7	0.00025	--	--	0.020	--
Vicksburg, Miss.						
Influent	2	0.132	0.9	<0.0002	--	--
Effluent	3	0.0544	2.1	<0.0002	--	--
Background	1	<0.0002	--	<0.0002	--	--
Southport, N. C. (1)						
Influent	3	0.06	0.43	0.0032	--	--
Effluent	3	0.0087	--	0.0025	--	--
Background	1	0.0086	--	0.0017	--	--
Southport, N. C. (2)						
Influent	3	--	--	0.0035	--	--
Effluent	3	--	--	0.0024	--	--
Background	1	--	--	0.0009	--	--

(Continued)

Table 8 (Concluded)

Location	No. of Samples	Titanium		Vanadium		Arsenic	
		Total mg/ℓ	<0.45 μm mg/ℓ	Total mg/ℓ	<0.45 μm mg/ℓ	Total mg/ℓ	<0.45 μm mg/ℓ
Sayreville, N. J.							
Influent	1	--	0.215	--	0.066	--	0.00047
Effluent	1	0.35	0.235	0.68	0.047	0.03	0.00023
Background	1	--	0.232	--	0.039	--	0.00035
Houston, Tex.							
Influent	4	3.3	0.0294	4.28	0.0284	0.266	0.00036
Effluent	4	0.36	0.0281	0.28	0.0224	0.148	0.00014
Background	1	0.0105	0.0001	0.32	0.0040	0.013	0.00080
Grand Haven, Mich.							
Influent	4	--	--	2.49	0.005	4.55	0.0004
Effluent	4	--	--	0.23	0.0055	0.32	0.0003
Background	1	--	--	0.029	0.004	0.0040	0.0003
Wilmington, N. C.							
Influent	3	--	--	--	--	--	--
Effluent	3	--	--	--	--	--	--
Background	2	--	--	--	--	--	--
Richmond, Va.							
Influent	4	--	--	--	--	--	--
Effluent	4	--	--	--	--	--	--
Background	2	--	--	--	--	--	--
Lake Charles, La.							
Influent	3	--	--	--	--	--	--
Effluent	3	--	--	--	--	--	--
Background	1	--	--	--	--	--	--
Seattle, Wash.							
Influent	5	--	--	--	--	1.20	0.064
Effluent, Pond 1	3	--	--	--	--	0.010	0.012
Effluent, Pond 2	10	--	--	--	--	0.009	0.006
Background	7	--	--	--	--	0.0025	--
Vicksburg, Miss.							
Influent	2	--	--	--	--	--	--
Effluent	3	--	--	--	--	--	--
Background	1	--	--	--	--	--	--
Southport, N. C. (1)							
Influent	3	--	--	--	--	--	--
Effluent	3	--	--	--	--	--	--
Background	1	--	--	--	--	--	--
Southport, N. C. (2)							
Influent	3	--	--	--	--	--	--
Effluent	3	--	--	--	--	--	--
Background	1	--	--	--	--	--	--

(Sheet 12 of 12)

Table 9

Statistical Character of Background Water, Influent, and Effluent Samples for Different Confined Disposal Areas (Sites 2-9)

Parameter	Number of Samples			Range			Mean			Standard Deviation			Probability
	Background Water	Influent	Effluent	Background Water	Influent	Effluent	Background Water	Influent	Effluent	Background Water	Influent	Effluent	F Value Exceeded
Water Temp, °C	1	24	26	3.5-3.5	1.0-23.7	0.0-34.0	3.5	14.9	16.8	0.0	7.299	10.16	0.308
Salinity, 0/00	4	23	23	0.0-21.0	0.0-21.5	0.0-22.0	6.7	8.2	9.6	10.50	7.800	9.202	0.862
Conductivity, mmhos/cm	5	21	22	0.39-31.5	0.0-32.1	0.0-39.2	8.4	11.3	14.0	14.25	12.39	15.62	0.788
Dissolved O_2, mg/ℓ	1	17	23	11.5-11.5	0.25-9.3	0.6-12.5	11.5	3.8	5.3	0.0	2.885	3.402	0.048 *
Slurry pH	5	16	16	5.5-7.6	6.25-8.3	6.9-8.1	6.6	7.16	7.51	0.775	0.516	0.316	0.0025 **
Particle size, Clay, %	0	17	2	- -	4 -68	58 -61	- -	43.2	59.5	- -	20.15	2.121	0.280
Silt, %	0	17	2	- -	17 -78	30 -35	- -	38.1	32.5	- -	16.68	3.536	0.649
Sand, %	0	16	2	- -	0.0-79.0	4.0-12.0	- -	19.9	8.0	- -	22.50	5.657	0.479
Coulter Counter >50%, μm	3	9	10	5.4-7.6	0.77-1.2	0.2-10.5	6.2	0.97	4.75	1.242	0.131	3.545	0.0035 **
>80%, μm	3	9	10	2.82-3.8	0.5-0.65	0.56-3.8	3.15	0.58	2.22	0.557	0.043	1.171	0.0001 **
Total solids, % wt	3	12	17	0.009-0.043	1.94-32.0	0.005-2.85	0.022	8.64	0.345	0.018	8.416	0.785	0.0006 **
Nonfilterable solids, % wt.	8	17	24	0.001-0.027	0.58-32.0	0.004-3.27	0.007	11.70	0.329	0.009	10.61	0.760	0.0000 **
Settleable solids, mℓ/ℓ	7	22	24	0.1-0.1	45 -999	0.1-950	0.1	452	69	0.0	288.8	205.6	0.0000 **

(Continued)

* Influent, effluent, and background water values are significantly different at $p \leq 0.05$.

** Influent, effluent, and background water values are significantly different at $p \leq 0.01$.

Table 9 (Continued)

Parameter	Number of Samples			Range			Mean			Standard Deviation			Probability F Value Exceeded
	Background Water	Influent	Effluent	Background Water	Influent	Effluent	Background Water	Influent	Effluent	Background Water	Influent	Effluent	
Total C, <0.45μm mg/ℓ	1	3	3	58 -58	30 -85	60 -70	58	57	65	0.0	27.54	5.000	0.872
Organic C, Total, mg/ℓ	13	16	13	2 -14	35.0-11500	4 -1060	6.5	3880	151	4.521	3405	303.1	0.0000 **
Solid, mg/kg	1	20	7	10 -10	974 -53400	11 -53200	10	25100	20800	0.0	16026	23677	0.385
<0.45μm, mg/ℓ	10	29	35	4 -43	3 -185	3 -120	15	27	19	12.89	43.32	22.31	0.456
Oil & Grease, Total, mg/ℓ	11	12	18	0.1-47.2	25.0-1497	2.4-196	6.5	458	27.5	14.32	433.0	50.99	0.0000 **
Solid, mg/kg	0	5	0	- -	2928 -8492	- -	- -	6060.	- -	- -	2121	- -	- -
<0.45μm, mg/ℓ	2	11	9	1.1-1.1	1.8-48.0	0.32-13.0	1.1	11.2	3.1	0.0	16.88	3.997	0.300
Total Chlorinated Pesticides													
op'DDE, mg/ℓ	5	14	13	<0.01-0.28	<0.01-0.53	<0.01-0.50	0.08	0.13	0.07	0.117	0.139	0.136	0.555
pp'DDE, mg/ℓ	5	14	13	<0.01-1.57	<0.01-1.72	<0.01-2.87	0.37	0.47	0.37	0.679	0.527	0.773	0.915
op'DDD, mg/ℓ	1	14	13	<0.01-<0.01	<0.01-1.28	<0.01-<0.01	<0.01	0.19	<0.01	0.0	0.368	0.0	0.183
pp'DDD, mg/ℓ	0	14	13	- -	<0.01-1.04	<0.01-<0.01	- -	0.21	<0.01	- -	0.368	0.0	0.050 *
op'DDT, mg/ℓ	1	14	13	<0.01-<0.01	<0.01-5.4	<0.01-0.23	<0.01	1.1	0.02	0.0	1.993	0.061	0.152
pp'DDT, mg/ℓ	1	14	13	<0.01-<0.01	<0.01-5.94	<0.01-0.96	<0.01	0.68	0.07	0.0	1.627	0.263	0.416
Total PCB, mg/ℓ	12	20	23	<0.01-0.3	<0.1-21	<0.1-7.66	0.058	5.81	0.50	0.090	5.451	1.607	0.0000 **

(Continued)

* Influent, effluent, and background water values are significantly different at $p \leq 0.05$.

** Influent, effluent, and background water values are significantly different at $p \leq 0.01$.

(Sheet 2 of 7)

Table 9 (Continued)

Parameter	Number of Samples			Range			Mean			Standard Deviation			Probability F Value Exceeded
	Background Water	Influent	Effluent	Background Water	Influent	Effluent	Background Water	Influent	Effluent	Background Water	Influent	Effluent	
Organic N, Total, mg/ℓ	17	28	26	<0.01-2.35	3.6-839	0.1-74.5	0.376	168	9.7	0.556	205.8	18.017	0.0000 **
Solid, mg/kg	0	21	3	- -	532 -3870	906 -3042	- -	1820	2220	- -	1087	1150	0.562
<0.45µm, mg/ℓ	10	28	34	0.1-1.35	0.1-27.6	0.1-6.7	0.55	4.3	1.6	0.397	5.871	1.627	0.0083 **
NH_3-N, Total, mg/ℓ	16	7	21	0.01-1.54	7.3-86.0	0.82-80.3	0.53	45.6	19.6	0.459	28.46	22.93	0.0000 **
Exch, mg/kg	0	18	7	- -	2.4-339	58.5-458	- -	110	196	- -	94.12	167.8	0.115
<0.45µm, mg/ℓ	10	29	35	0.01-0.82	0.66-71.7	0.74-70.9	0.27	20.8	13.6	0.270	19.03	15.73	0.0034 **
NO_3+NO_2-N, mg/ℓ	16	26	32	0.01-1.98	0.01-0.82	0.01-1.83	0.46	0.18	0.35	0.523	0.234	0.396	0.066
Total P, Total, mg/ℓ	17	28	34	0.07-0.86	12.8-496	0.11-82.1	0.26	155	11.7	0.236	133.6	21.85	0.0000 **
Solid, mg/kg	0	21	3	- -	639 -4400	1400 -4000	- -	1850	3070	- -	1179	1447	0.117
<0.45µm, mg/ℓ	10	29	32	0.02-0.5	0.03-9.47	0.01-1.53	0.18	0.86	0.33	0.175	2.047	0.499	0.221
Orthophosphate P, <0.45µm, mg/ℓ	8	22	28	0.03-0.16	0.04-5.89	0.01-1.04	0.08	0.79	0.18	0.051	1.585	0.294	0.074
Alkalinity, mg/ℓ as $CaCO_3$	10	29	35	16.27-290	51.38-1520	29.75-670	88.10	412	287	83.08	317.4	184.5	0.0015 **
Chloride, mg/ℓ	10	29	34	5.0-20600	5.0-19200	5.0-16400	4150	8290	7810	7909	7326	6365	0.261
Total Sulfide, Solids, mg/kg	0	21	3	- -	17.8-3090	94.1-327	- -	493	208	- -	679.5	116.6	0.484

(Continued)

* Influent, effluent, and background water values are significantly different at $p \leq 0.05$.

** Influent, effluent, and background water values are significantly different at $p \leq 0.01$.

Table 9 (Continued)

Parameter	Number of Samples			Range			Mean			Standard Deviation			Probability F Value Exceeded
	Background Water	Influent	Effluent	Background Water	Influent	Effluent	Background Water	Influent	Effluent	Background Water	Influent	Effluent	
Cation Exch. Cap, meq./100g	0	19	6	- -	2.37-88.2	65.9-120.6	- -	50.9	82.5	- -	25.16	21.55	0.011 *
Calcium,													
Total, mg/ℓ	10	24	25	4.8-390	45.7-11500	16.8-560	98.7	2450	250	142.3	3420	163.6	0.0013 **
Solids, mg/kg	0	22	7	- -	1150 -37900	1190-26100	- -	16300	9930	- -	13722	10844	0.274
<0.45μm, mg/ℓ	10	24	25	4.1-390	8.0-416	13.0-532	95.4	181.1	204.8	141.6	140.7	166.3	0.165
Magnesium,													
Total, mg/ℓ	10	24	25	2.5-1300	26.5-1320	3.15-1200	279.0	1270	464	514.4	1057	416.8	0.0003 *
Solids, mg/kg	0	22	8	- -	933 -37800	4700.0-16300	- -	11200	10200	- -	8239	4853	0.746
<0.45μm, mg/ℓ	10	24	25	1.5-1300	2.6-1300	2.6-1200	265	415	398	497.1	432.8	447.9	0.661
Potassium,													
Total, mg/ℓ	10	24	25	2.2-338	128 -6360	4.6-1145	121	1770	390	148.3	1644	362.3	0.0000 **
Solids, mg/kg	0	20	8	- -	3100 -43500	8330-18200	- -	13200	14200	- -	8655	3647	0.762
<0.45μm, mg/ℓ	10	24	25	1.2-380	1.6-450	1.5-440	98.6	163	166	151.6	137.0	144.9	0.418
Sodium,													
Total, mg/ℓ	8	19	23	6.5-9800	85.0-9900	6.5-11300	2470	3900	3540	4026	3422	3963	0.772
Solids, mg/kg	0	11	5	- -	2394 -13100	240-43200	- -	9430	23600	- -	3583	18176	0.021*
<0.45μm, mg/ℓ	9	24	25	6.1-9700	6.2-9500	6.0-9400	2183	3620	3490	3737	3806	3798	0.607

(Continued)

* Influent, effluent, and background water values are significantly different at $p \leq 0.05$.

** Influent, effluent, and background water values are significantly different at $p \leq 0.01$.

(Sheet 4 of 7)

Table 9 (Continued)

Parameter	Number of Samples			Range			Mean			Standard Deviation			Probability F Value Exceeded
	Background Water	Influent	Effluent	Background Water	Influent	Effluent	Background Water	Influent	Effluent	Background Water	Influent	Effluent	
Iron,													
Total, mg/ℓ	17	29	34	0.38-63.6	46.1-12600	1.14-1290	5.19	3400	193	15.13	3565.	343.0	0.0000 **
Solids, mg/kg	0	22	8	- -	24300-81600	24100-48300	- -	42300	38300	- -	13117	7068	0.421
<0.45μm, mg/ℓ	10	29	35	0.001-1.43	0.043-15.9	0.01-10.1	0.378	3.52	0.814	0.457	5.009	1.816	0.0037 **
Manganese,													
Total, mg/ℓ	17	29	35	0.04-0.40	0.8-310	0.21-48.5	0.107	63.1	7.9	0.095	73.00	13.61	0.0000 **
Solids, mg/kg	0	22	8	- -	250-2110	683-2170	- -	682	1160	- -	383.9	470.9	0.0081 **
<0.45μm, mg/ℓ	10	29	35	0.002-0.184	0.004-14.4	0.002-7.95	0.059	2.35	1.45	0.057	3.485	2.090	0.057 *
Zinc,													
Total, mg/ℓ	17	25	30	0.006-1.28	0.6-206	0.026-5.49	0.238	27.5	1.03	0.422	40.28	1.481	0.0003 **
Solids, mg/kg	0	22	8	- -	55.8-1960	31.7-3660	- -	323	621	- -	435	1232	0.325
<0.45μm, mg/ℓ	10	25	30	0.001-0.121	0.001-0.496	0.002-0.228	0.028	0.055	0.064	0.042	0.10	0.069	0.472
Cadmium,													
Total, mg/ℓ	13	26	32	<0.0002-0.01	0.002-7.17	0.001-0.37	0.003	1.39	0.051	0.003	2.036	0.102	0.0002 **
Solid, mg/kg	0	17	4	- -	0.048-45.3	0.046-4.87	- -	7.1	1.62	- -	10.44	2.216	0.319
<0.45μm, mg/ℓ	9	29	35	<0.0002-0.012	0.0002-0.015	0.0001-0.011	0.003	0.004	0.003	0.004	0.004	0.003	0.824

(Continued)

* Influent, effluent, and background water values are significantly different at $p \leq 0.05$.

** Influent, effluent, and background water values are significantly different at $p \leq 0.01$.

Table 9 (Continued)

Parameter	Number of Samples			Range			Mean			Standard Deviation			Probability F Value Exceeded
	Background Water	Influent	Effluent	Background Water	Influent	Effluent	Background Water	Influent	Effluent	Background Water	Influent	Effluent	
Copper,													
Total, mg/ℓ	15	29	35	0.003-0.16	0.1-18.2	0.02-1.59	0.038	6.09	0.28	0.042	5.327	0.414	0.0000 **
Solid, mg/kg	0	22	8	- -	6.0-165	26.0-131	- -	52.2	46.3	- -	48.43	35.00	0.753
<0.45μm, mg/ℓ	10	29	35	0.001-0.028	0.001-0.106	0.001-0.1	0.009	0.019	0.021	0.010	0.026	0.022	0.323
Nickel,													
Total, mg/ℓ	14	29	30	<0.01-1.5	0.21-18.2	<0.01-1.70	0.120	5.8	0.30	0.397	4.555	0.414	0.0000 **
Solid, mg/kg	0	22	8	- -	15.4-124	25.3-74.6	- -	47.3	47.1	- -	26.06	15.18	0.982
<0.45μm, mg/ℓ	10	29	30	0.003-0.036	0.003-0.047	0.002-0.043	0.011	0.014	0.012	0.013	0.013	0.013	0.820
Lead,													
Total, mg/ℓ	9	28	23	0.001-0.049	0.24-86.5	0.001-7.57	0.011	16.2	1.38	0.015	23.88	2.437	0.0033 **
Solid, mg/kg	0	21	8	- -	5.7-327	1.0-142	- -	81.5	43.7	- -	88.65	43.58	0.262
<0.45μm, mg/ℓ	10	24	25	<0.001-0.005	<0.001-0.012	<0.001-0.007	0.002	0.002	0.002	0.002	0.003	0.002	0.725
Mercury,													
Total, mg/ℓ	14	18	24	<0.0002-0.009	0.001-0.243	<0.0002-0.367	0.001	0.044	0.024	0.002	0.059	0.080	0.161
Solid, mg/kg	0	18	6	- -	0.07-1.66	0.08-3.2	- -	0.46	0.79	- -	0.438	1.234	0.334
<0.45μm, mg/ℓ	8	19	28	<0.0002-0.004	0.0002-0.008	<0.0002-0.006	0.001	0.001	0.001	0.001	0.002	0.002	0.895

(Continued)

** Influent, effluent, and background water values are significantly different at $p \leq 0.01$.

(Sheet 6 of 7)

Table 9 (Concluded)

Parameter	Number of Samples			Range			Mean			Standard Deviation			Probability F Value Exceeded
	Background Water	Influent	Effluent	Background Water	Influent	Effluent	Background Water	Influent	Effluent	Background Water	Influent	Effluent	
Chromium, Total, mg/ℓ	8	3	8	0.009-0.026	56.7-76.6	0.024-0.58	0.018	63.8	0.12	0.005	11.11	0.190	0.0000 **
<0.45µm, mg/ℓ	1	3	8	0.003-0.003	0.003-0.005	0.004-0.033†	0.003	0.004	0.017†	0.0	0.001	0.011	0.132
Titanium, Total, mg/ℓ	1	4	4	0.01-0.01	2.1-4.35	2.55-0.50	0.010	3.31	0.36	0.0	0.956	0.117	0.0017 **
<0.45µm, mg/ℓ	1	4	4	0.0001-0.0001	0.025-0.038	0.020-0.036	0.0001	0.029	0.028	0.0	0.006	0.009	0.030 *
Vanadium, Total, mg/ℓ	2	7	7	0.029-0.32	2.29-5.23	0.76-0.47	0.175	3.52	0.26	0.206	1.321	0.134	0.0000 **
<0.45µm, mg/ℓ	2	7	7	0.004-0.004	0.004-0.039	0.003-0.027	0.004	0.018	0.015	0.0	0.013	0.010	0.317
Arsenic, Total, mg/ℓ	9	12	17	0.001-0.013	0.181-6.02	0.003-0.41	0.004	1.73	0.096	0.004	1.979	0.126	0.0006 **
<0.45µm, mg/ℓ	2	12	17	0.0003-0.001	0.0001-0.117	0.0001-0.021	0.001	0.027	0.004	0.000	0.043	0.006	0.077

* Influent, effluent and background water values are significantly different at $p \leq 0.05$.

** Influent, effluent and background water values are significantly different at $p \leq 0.01$.

† High effluent concentrations resulted from inclusion of high effluent values from the Seattle, Washington site, which lacked comparable influent data.

(Sheet 7 of 7)

Table 10

Geochemical Phase Partitioning of Elements in Influent and Effluent Solids from Four Confined Land Disposal Areas*

Partitioning Phase	Calcium		Magnesium		Potassium		Sodium		Iron**	
	mg/kg	%	mg/kg	%	mg/kg	%	mg/kg	%	mg/kg	%
A. Exchangeable†										
Influent	2,120 (1,550 to 2,887)	30.4	1,638 (181 to 3,600)	17.8	955 (105 to 1,570)	6.8	3,080 (20.7 to 9,000)	31.9	483 (32.1 to 1,428)	1.1
Effluent	2,468 (1,748 to 3,232)	37.3	1,660 (372 to 3,300)	18.2	911 (202 to 1,394)	5.8	4,944 (68.7 to 12,790)	54.8	147 (3.3 to 579)	0.3
B. Carbonate††										
Influent	1,708 (259 to 5,200)	14.3	502 (268 to 1,200)	5.6	126 (58.0 to 186)	0.9	257 (87.7 to 450)	4.3	2,332 (58.5 to 8,100)	5.0
Effluent	1,185 (362 to 2,700)	12.8	585 (75.2 to 2,000)	4.9	107 (26.2 to 271)	0.6	285 (11.1 to 705)	9.0	2,675 (151 to 6,800)	4.4
C. Easily reducible‡										
Influent	899 (469 to 1,500)	12.3	282 (133 to 360)	3.7	54.6 (30.2 to 81.0)	0.4	27.4 (5.0 to 60.0)	0.6	3,088 (1,501 to 7,314)	6.9
Effluent	641 (379 to 984)	9.9	276 (52.9 to 474)	3.7	54.0 (1.96 to 87.8)	0.4	25.9 (4.58 to 53.2)	0.8	4,062 (1,901 to 8,395)	7.4
D. Remaining phases										
Influent	5,942 (1,209 to 20,800)	43.0	6,641 (3,929 to 16,100)	72.9	13,630 (9,561 to 17,770)	91.9	6,205 (0.0 to 24,770)	63.2	36,470 (30,420 to 42,340)	87.0
Effluent	6,639 (847 to 26,250)	40.0	7,202 (2,917 to 16,800)	73.2	14,120 (9,732 to 18,250)	93.2	6,235 (0.0 to 24,790)	35.4	47,860 (39,620 to 59,000)	87.9
E. Total phases										
Influent	10,670	100	9,060	100	14,760	100	9,570	100	42,380	100
Effluent	10,930	100	9,720	100	15,190	100	11,490	100	54,740	100
Major partitioning phase increases										
Influent	Carbonate Easily reducible		--		--		--		Exchangeable	
Effluent	Exchangeable		--		--		Exchangeable		Easily reducible	

(Continued)

Note: All mg/kg values are based on dry-weight solids.
Percent calculations are based on the average of the percentage calculations for each site.

* Based on 5 influent and effluent samples, except for mercury (4), chromium (1), and arsenic (1).

** Data for influent-effluent and the partitioning phases are significantly different at $p \le 0.1$ using the F-test, for either mg/kg or percent values.

† Ammonium acetate extractable.

†† 1 M acetic acid extractable.

‡ 0.1 M hydroxylamine hydrochloride in 0.01 M nitric acid extractable.

(Sheet 1 of 3)

Table 10 (Continued)

Partitioning Phase	Manganese mg/kg	Manganese %	Zinc** mg/kg	Zinc** %	Lead mg/kg	Lead %	Cadmium** mg/kg	Cadmium** %	Copper mg/kg	Copper %
A. Exchangeable†										
Influent	203 (12.0 to 312)	22.5	5.7 (0.5 to 12.1)	3.2	2.1 (0.00 to 4.7)	5.8	0.03 (0.002 to 0.07)	20.1	0.13 (0.00 to 0.22)	0.3
Effluent	225 (11.0 to 392)	16.4	10.0 (3.2 to 24.4)	2.7	2.5 (0.00 to 7.3)	3.0	0.29 (0.00 to 1.3)	18.0	1.1 (0.00 to 4.6)	1.1
B. Carbonate††										
Influent	167 (70.0 to 312)	19.6	21.6 (9.1 to 41.0)	10.8	0.34 (0.00 to 1.7)	0.2	0.07 (0.008 to 0.11)	21.4	1.8 (0.40 to 3.2)	5.3
Effluent	246 (91.0 to 446)	20.3	199 (4.8 to 585)	33.1	12.7 (0.00 to 28.9)	11.4	1.7 (0.007 to 5.5)	56.7	11.5 (0.00 to 26.0)	12.4
C. Easily reducible‡										
Influent	102 (18.0 to 179)	12.0	23.3 (9.5 to 34.2)	11.0	11.5 (0.00 to 33.5)	19.7	0.07 (0.00 to 0.15)	9.2	4.6 (0.50 to 9.9)	11.5
Effluent	281 (133 to 777)	20.1	43.1 (13.6 to 105)	9.0	19.0 (0.00 to 39.2)	16.8	0.44 (0.00 to 2.1)	11.8	9.8 (1.5 to 20.0)	12.1
D. Remainimg phases										
Influent	370 (239 to 659)	45.9	172 (97.5 to 385)	75.0	62.1 (17.9 to 217)	74.3	0.86 (0.00 to 3.7)	49.3	47.3 (20.8 to 123)	82.9
Effluent	486 (289 to 751)	43.2	191 (103 to 275)	55.2	88.2 (17.9 to 254)	68.8	0.93 (0.00 to 2.4)	13.5	52.9 (26.9 to 100)	74.4
E. Total phases										
Influent	842	100	222	100	76.2	100	1.05	100	54.0	100
Effluent	1240	100	443	100	122	100	3.35	100	75.4	100
Major partitioning phase increases										
Influent	--		--		--		--		--	
Effluent	Carbonate Easily reducible		Carbonate		Carbonate		Carbonate Easily reducible		Exchangeable Carbonate Easily reducible	

(Continued)

** Data for influent-effluent and the partitioning phases are significantly different at $p \leq 0.1$ using the F-test, for either mg/kg or percent values.

† Ammonium acetate extractable.

†† 1 M acetic acid extractable.

‡ 0.1 M hydroxylamine hydrochloride in 0.01 M nitric acid extractable.

(Sheet 2 of 3)

Table 10 (Concluded)

Partitioning Phase	Nickel mg/kg	Nickel %	Mercury mg/kg	Mercury %	Chromium mg/kg	Chromium %	Arsenic mg/kg	Arsenic %
A. Exchangeable†								
Influent	3.0 (1.2 to 3.8)	5.6	0.001 (0.00 to 0.005)	0.1	<0.05	<0.1	0.15	1.3
Effluent	4.2 (3.0 to 6.0)	5.3	0.006 (0.00 to 0.02)	1.0	0.2	0.1	7.1	14.6
B. Carbonate††								
Influent	4.6 (2.3 to 7.7)	7.4	0.00	<0.1	3.2	3.3	<0.15	<0.3
Effluent	5.7 (1.4 to 15.0)	6.1	0.00	<0.1	2.5	1.3	<0.15	<0.3
C. Easily reducible‡								
Influent	3.3 (2.4 to 4.4)	5.6	0.00	<0.1	4.4	4.5	<0.15	<0.3
Effluent	2.3 (0.00 to 4.1)	2.8	0.055 (0.00 to 0.23)	4.3	1.0	0.5	<0.15	<0.3
D. Remaining phases								
Influent	52.6 (28.4 to 79.5)	81.4	0.72 (0.52 to 0.99)	99.9	90.5	92.2	11.0	98.7
Effluent	75.0 (40.0 to 100)	85.8	0.73 (0.29 to 1.1)	94.7	190.3	98.1	41.4	85.4
E. Total phases								
Influent	63.6	100	0.72	100	98.1	100	11.2	100
Effluent	87.4	100	0.79	100	194	100	48.5	100
Major partitioning phase increases								
Influent	Easily reducible		--		Easily reducible		--	
Effluent	Organic-sulfide-residual		--		Organic-sulfide-residual		Exchangeable Organic-sulfide-residual	

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† Ammonium acetate extractable.

†† 1 M acetic acid extractable.

‡ 0.1 M hydroxylamine hydrochloride in 0.01 M nitric acid extractable.

(Sheet 3 of 3)

Table 11

Relationship Between Total Solids and Dissolved Oxygen in Effluents

Location and Sampling Dates	Total Effluent Nonfilterable Solids Percent by Weight	Dissolved Oxygen, mg/ℓ
Wilmington, N. C.		
16 Dec 75	0.052	0.6
22 Jan 76	2.050	4.0
23 Jan 76	0.600	2.05
Richmond, Va.		
27 Jan 76	0.013	10.0
19 Feb 76	0.028	8.6
20 Feb 76	0.014	9.3
21 Feb 76	0.130	3.1
Lake Charles, La.		
4 Feb 76	3.27	1.5
5 Feb 76	0.0036	5.75
6 Feb 76	0.380	3.8
Vicksburg, Miss.		
17 Mar 76	0.520	1.5
12 Apr 76	0.150	3.0
15 Apr 76	0.073	3.05
Seattle, Wash.		
16 Mar 76	0.006	7.5
19 Mar 76	0.010	6.6
22 Mar 76	0.006	5.5
23 Mar 76	0.011	5.0
3 Apr 76	0.006	3.8
4 Apr 76	0.015	3.7
Southport, N. C.		
6 May 76	0.013	3.5
6 May 76	0.008	2.4
7 May 76	0.025	3.95
17 May 76	0.048	4.7
19 May 76	0.366	6.6
20 May 76	0.100	7.0

Table 12

Average Eh and pH Values of Freshly Dredged Sediments in Eight Confined Upland Disposal Areas

	Sediment Eh*		Sediment pH	
Location	No. of Measurements	Average (Range)	No. of Measurements	Average (Range)
Sayreville, N. J.	6	+040 (+005 to +075)	3	7.15 (7.1 to 7.2)
Houston, Tex.	28	-110 (-285 to +085)	22	7.05 (6.65 to 7.35)
Grand Haven, Mich.	--	--	--	--
Wilmington, N. C.	26	+070 (-180 to +370)	28	6.75 (5.0 to 7.5)
Richmond, Va.	7	+320 (+020 to +505)	-- --	-- --
Lake Charles, La.	9	+080 (-255 to +245)	6	7.2 (6.9 to 7.5)
Seattle, Wash. (Pond 1)	32	-093 (-260 to +125)	11	7.9 (6.9 to 9.05)
Seattle, Wash. (Pond 2)	8	+271 (+105 to +465)	3	6.1 (5.75 to 6.45)
Vicksburg, Miss.	22	-045 (-295 to +155)	14	6.85 (6.5 to 7.05)
Southport, N. C.	24	-330 (-410 to -130)	11	7.25 (6.9 to 7.7)

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* Eh in millivolts; measured at 10 to 15 cm depth in freshly deposited sediment.

Table 13

Interaction of Residence Time, Temperature, Turbidity, pH, and Concentrations of Nitrogen and Phosphorus in Brown Lake Disposal Area, Vicksburg, Miss.

Sample Identification	No. Days from Dredging Initiation	NH_4 - N mg/ℓ*	TKN mg/ℓ*	Total P mg/ℓ*	Ortho-phosphate mg/ℓ*	Water Temperature °C	Water pH*	Flow Rate (Estimate) ℓ/min	Turbidity**	Effluent Residence Time, days†
Background lake water (2 Mar 76)	- 4	0.82			0.15				Low	
Influent (8 Mar 76)	2	12.0	246.5	--	--					
Influent (15 Mar 76)	9	1.4	84.7	0.16	0.06					
Influent (16 Mar 76)	10	6.4	59.9	0.29	0.23					
Influent (17 Mar 76)††	11	8.55			0.07	14.5	6.85			
Influent (29 Mar 76)	23	4.61	367.8	0.09	0.07					
Influent (15 Apr 76)††	40	5.35	--	--	0.10	22.5	7.2			
Effluent (8 Mar 76)	2	0.8	5.5	0.10	0.06					
Effluent (9 Mar 76)	3	0.6	3.5	0.16	0.05					
Effluent (10 Mar 76)	4	0.8	3.9	0.10	0.08					
Effluent (11 Mar 76)	5	0.8	3.1	0.14	0.05			40		
Effluent (12 Mar 76)	6	1.0	13.4	0.15	0.08		7.9	30		
Effluent (15 Mar 76)	9	0.8	5.1		0.06			1000	High	

(Continued)

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* Total sample.

** Visual observation of sample; high turbidity indicates 10-cm water column is not transparent.

† Postdredging drainage time.

†† Days on which composite daily samples were obtained.

Table 13 (Concluded)

Sample Identification	No. Days After Initiation	$NH_4 - N$ mg/ℓ	TKN mg/ℓ	Total P mg/ℓ	Ortho-phosphate mg/ℓ	Water Temperature °C	Water pH	Flow Rate (Estimate) ℓ/min	Turbidity	Effluent Residence Time, days
Effluent (16 Mar 76)	10	3.2	27.5	0.16	0.11					
Effluent (17 Mar 76)††	11	4.1	30.7	0.17	0.09	15.8	7.3			
Effluent (19 Mar 76)	13	--	--	--	0.13					
Effluent (29 Mar 76)	23	0.47	5.7	0.13	0.13					
Effluent (31 Mar 76)	25	0.71	8.2	0.23	0.19					
Effluent (2 Apr 76)	27	0.64	7.1	0.15	0.09	30.0	7.9			
Effluent (5 Apr 76)	30	0.67	25.7	0.18	0.12					
Effluent (7 Apr 76)	32	0.51	6.5	0.11	0.09					
Effluent (12 Apr 76)††	37	0.79	9.5	0.12	0.11	29.0				
Effluent (15 Apr 76)†† (afternoon)	40	0.25	5.4	0.20	0.15	33.5	8.1			
Postdredging drainage										
Effluent (19 Apr 76) (afternoon)	44	0.19	9.3	0.13	0.13	31.0	8.05	60	High	4
Effluent (20 Apr 76) (morning)	45	0.08	2.7	0.20	0.14	22.0	8.5	60	Low	5
Effluent (21 Apr 76) (morning)	46	0.02	7.6	0.47	0.40	21.0	9.15	40	Low	6
Effluent (22 Apr 76) (morning)	47	--	--	--	--	24.5	8.65	30	Low	7
Effluent (23 Apr 76) (afternoon)	48	0.18	8.4	0.30	0.06	32.5	7.8	20	High	8

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†† Days on which composite daily samples were obtained.

Table 14

Effect of Retention Time on the Removal of Soluble Phase Ammonium and Nitrate Nitrogen from Treatment Area Effluents

Location and Sampling Dates	Treatment Area Size, acres	Effluent Retention Time*	Influent $NH_3 - N$ mg/ℓ	Effluent $NH_3 - N$ mg/ℓ	Effluent $NO_3 - N$ mg/ℓ
Lake Charles, La. 4 Feb 76	150	Short	24.7	10.2	<0.01
Lake Charles, La. 5 Feb 76	150	Long	14.4	7.2	<0.01
Lake Charles, La. 6 Feb 76	150	Medium	19.4	13.4	0.08
Seattle, Wash. (Pond 1) Avg: 4 Apr 76 to 6 Apr 76	1.9	Short	--	5.1	0.36
Seattle, Wash. 4 Apr 76 to 6 Apr 76	1.9	Medium	--	5.3	0.31
Wilmington, N. C. Avg: 16 Dec 76 22 Jan 76, 23 Jan 76 (avg)	400 (thick dormant vegetation)	--	23.8	22.6	0.28

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* Short: less than 1 day residence time; long: greater than 4 days residency for most of effluent.

Table 15

Comparative Analytical Data for Chemical Concentrations in Bottom Sediments, Pore Water, Standard Elutriate Test Filtrates, and Disposal Area Influents and Effluents from Seattle, Washington

Sample Identification	Organic Carbon		Oil and Grease			PCB's (1242, 1254, 1260)			Organic Nitrogen		
	Total mg/ℓ	<0.45 µm mg/ℓ	Total mg/ℓ	Solids mg/kg	<0.45 µm mg/ℓ	Total mg/ℓ	Solids mg/kg	<0.45 µm mg/ℓ	Total mg/ℓ	Solids mg/kg	<0.45 µm mg/ℓ
Slip 1 sediment	--	58	711	1638	--	16.5	38.5	0.425	540	1225	5.8
Disposal area influent	--	11	582	6060	21.3	5.23	96.2	0.024	68	1116	3.0
Slip 1 elutriate water*	3	--	<1	--	--	<0.0001	--	--	0.13	--	--
Slip 1 standard elutriate test**	--	22.5	--	--	5.35	--	--	0.048	--	--	1.65
Diluted sediment pore water (1/4)†	--	[14.5]	--	--	--	--	--	0.106	--	--	[1.45]
Disposal area Pond 1 effluent	--	8	--	--	5.0	0.0077	32.7	0.00064	--	--	5.1
Disposal area Pond 2 effluent	--	10	--	--	5.25	0.00063	--	0.00016	0.2	--	0.4

Sample Identification	Ammonium Nitrogen			Nitrate + Nitrite Nitrogen	Total Phosphorus		
	Total†† mg/ℓ	Exchangeable mg/kg (mg/ℓ)	<0.45 µm mg/ℓ	<0.45 µm, mg/ℓ	Total mg/ℓ	Solids mg/kg	<0.45 µm mg/ℓ
Slip 1 sediment	28.8	47 (20.8)	8.0	0.57	540	1220	0.75
Disposal area influent	--	10 (0.7)	9.0	0.27	94.5	1418	0.40
Slip 1 elutriate water*	0.04	--	--	0.41	0.10	--	--
Slip 1 standard elutriate test**	--	--	3.0	0.50	--	--	0.26
Diluted sediment pore water (1/4)†	--	--	2.0	0.38	--	--	0.24
Disposal area Pond 1 effluent	--	--	5.1	0.36	0.6	--	0.30
Disposal area Pond 2 effluent	6.6	122 (<0.1)	6.6	0.35	0.25	--	0.05

(Continued)

Note: All solids data are mg/kg dry weight.

* Unfiltered background water from the dredging site; mixed in a 4:1 ratio with Slip 1 sediment.

** The water phase was filtered through a 0.45-µm membrane filter; aliquots used for the oil and grease and PCB analyses were centrifuged to approximate 0.45-µm filtration.

† Values in brackets are uncorrected for the soluble phase contribution from the background dilution water.

†† Exchangeable ammonium-N + <0.45 µm ammonium-N.

Table 15 (Concluded)

Sample Identification	Iron Total mg/ℓ	Iron Solids mg/kg	Iron <0.45 μm mg/ℓ	Manganese Total mg/ℓ	Manganese Solids mg/kg	Manganese <0.45 μm mg/ℓ	Zinc Total mg/ℓ	Zinc Solids mg/kg	Zinc <0.45 μm mg/ℓ
Slip 1 sediment	22100	50,200	10.6	228	511	3.76	430	992	0.026
Disposal area influent	2980	46,470	0.275	29.7	427	0.197	68.7	936	0.007
Slip 1 elutriate water*	1.30	--	--	0.080	--	--	0.020	--	--
Slip 1 standard elutriate test**	--	--	0.38	--	--	1.94	--	--	0.006
Diluted sediment pore water (1/4)†	--	--	[2.65]	--	--	[0.94]	--	--	[0.007]
Disposal area Pond 1 effluent	1.97	--	0.19	0.337	--	0.256	0.104	--	0.026
Disposal area Pond 2 effluent	4.25	41,300	0.24	0.96	928	0.81	0.237	1.3	0.108

Sample Identification	Cadmium Total mg/ℓ	Cadmium Solids mg/kg	Cadmium <0.45 μm mg/ℓ	Copper Total mg/ℓ	Copper Solids mg/kg	Copper <0.45 μm mg/ℓ	Nickel Total mg/ℓ	Nickel Solids mg/kg	Nickel <0.45 μm mg/ℓ
Slip 1 sediment	2.05	4.8	0.005	45	102	0.0073	--	--	<0.01
Disposal area influent	0.41	5.8	<0.002	9.05	135	0.052	3.85	61.8	0.02
Slip 1 elutriate water*	0.008	--	--	0.0072	--	--	<0.01	--	--
Slip 1 standard elutriate test**	--	--	0.005	--	--	0.0088	--	--	<0.01
Diluted sediment pore water (1/4)†	--	--	[0.001]	--	--	[0.0018]	--	--	<0.01
Disposal area Pond 1 effluent	0.002	--	<0.002	0.077	--	0.057	--	--	--
Disposal area Pond 2 effluent	0.004	6.8	0.002	0.053	198	0.042	0.02	25.8	0.02

Sample Identification	Mercury Total mg/ℓ	Mercury Solids mg/kg	Mercury <0.45 μm mg/ℓ	Chromium Total mg/ℓ	Chromium Solids mg/kg	Chromium <0.45 μm mg/ℓ	Arsenic Total mg/ℓ	Arsenic Solids mg/kg	Arsenic <0.45 μm mg/ℓ
Slip 1 sediment	0.1	0.2	0.0004	23	52	0.037	7	16	0.024
Disposal area influent	0.04	0.64	<0.0002	--	--	--	1.2	17.2	0.064
Slip 1 elutriate water*	0.0004	--	--	0.016	--	--	0.002	--	--
Slip 1 standard elutriate test**	--	--	0.0002	--	--	0.045	--	--	0.013
Diluted sediment pore water (1/4)†	--	--	[0.0001]	--	--	[0.009]	--	--	[0.006]
Disposal area Pond 1 effluent	0.0005	--	0.0002	0.036	--	0.025	0.010	--	0.012
Disposal area Pond 2 effluent	0.00015	--	0.0001	0.028	190	0.025	0.009	23	0.006

---

* Unfiltered background water from the dredging site; mixed in a 4:1 ratio with Slip 1 sediment.

** The water phase was filtered through a 0.45-μm membrane filter; aliquots used for the oil and grease and PCB analyses were centrifuged to approximate 0.45-μm filtration.

† Values in brackets are uncorrected for the soluble phase contribution from the background dilution water.

Table 16

Summary of the Standard Elutriate and Diluted Sediment Pore-Water Test Data of Sediments from Slip 1, Seattle, Wash.

Parameter	Diluted (1:4) Sediment Pore Water		Standard Elutriate (SE) Test		Remarks
	Pond 1	Pond 2	Pond 1	Pond 2	
Organic C	Fair (>)*	Good (>)*	Fair (>)	Fair (>)	Unrepresentative sediment samples or analytical errors.
Oil and grease	--	--	Good (∿)	Good (∿)	Surface oil films must be considered for good results.
PCB's	Poor (>>)	Poor (>>)	Poor (>>)	Poor (>>)	Elutriate test was best; unrepresentative sediment samples; increased SE residence time is suggested.
Organic N	Poor (<)*	Poor (>)*	Poor (<)	Poor (>)	Longer SE residence time is suggested; the residence time seems critical for breakdown of organic N to ammonium.
Ammonium N	Fair (<)	Poor (<)	Fair (<)	Fair (<)	Longer SE residence time is suggested.
Nitrate N	Good (∿)	Good (∿)	Good (>)	Good (>)	Nitrate concentration originates from elutriate (surface) water; SE not recommended unless disposal area conditions can be closely duplicated (eg., aeration, residence time).
Total P	Good (<)	Poor (>)	Good (<)	Poor (>)	Probably good for most sites except where heavy iron precipitation occurs; longer SE residence time is suggested to duplicate pond 2 effluent.
Iron	Poor (>>)*	Poor (>>)*	Fair (>)	Fair (>)	Iron precipitation is rapid; longer SE residence time or use of a filter size smaller than 0.45 μm is suggested; Fe release from dikes was possible.
Manganese	Poor (>)*	Good (>)*	Poor (>>)	Fair (>)	Longer SE residence time is suggested under carefully controlled Eh; release of Mn from dike sediments was possible at this site; test may be more meaningful at other sites.

(Continued)

Note: Good = < ±1.5 times effluent values from the respective pond.
Fair = ±1.5 to ±3 times effluent value from the respective pond.
Poor = > ±3 times effluent value from the respective pond.
(>) = value is greater than the effluent concentration from the respective pond; double symbol indicates a very large difference.
(<) = value is less than the effluent concentration from the respective pond; double symbol indicates a very large difference.
(∿) = value is similar to the effluent concentration from the respective pond.

* These values were not corrected for soluble phase concentrations in the background water at the dredging site; these comparisons represent the sediment pore-water (< 0.45-μm) filtrate concentrations, divided by 4.

Table 16 (Concluded)

Parameter	Diluted (1:4) Sediment Pore Water		Standard Elutriate (SE) Test		Remarks
	Pond 1	Pond 2	Pond 1	Pond 2	
Zinc	Poor (<)*	Poor (<<)*	Poor (<)	Poor (<<)	Longer SE residence time is suggested; release of Zn from dike sediments was possible at this site.
Cadmium	Good (?)*	Fair (<)*	Poor (>)	Fair (>)	Both the pore-water dilution and SE seem to be fair; variance may result from unrepresentative sediment samples.
Copper	Poor (<<)*	Poor (<<)*	Poor (<)	Poor (<)	Longer SE residence time is suggested.
Nickel	--	Poor (<)	--	Poor (<)	Longer SE residence time is suggested.
Mercury	Fair (<)*	Good (∿)*	Good (∿)	Fair (>)	Both the pore dilution and SE seem to be good.
Chromium	Fair (<)*	Fair (<)*	Fair (>)	Fair (>)	The SE seems a little better although both tests seem to be fair to good.
Arsenic	Fair (<)*	Good (∿)*	Good (∿)	Fair (>)	Both tests seem to be good and variance may be caused by unrepresentative sediment samples or analytical error; a more representative SE residence time should be best.

Photo 1. Sayreville, N. J., disposal area; cross-dike between compartments 2 and 3 and thick vegetation (*Phragmites communis*) in compartment 2

Photo 2. Sayreville, N. J., disposal area; view from sluice in compartment 3 toward influent discharge pipes near bridge in background

Photo 3. Houston, Tex., disposal area; east section of disposal area. Effluent sluice is just out of picture to the right

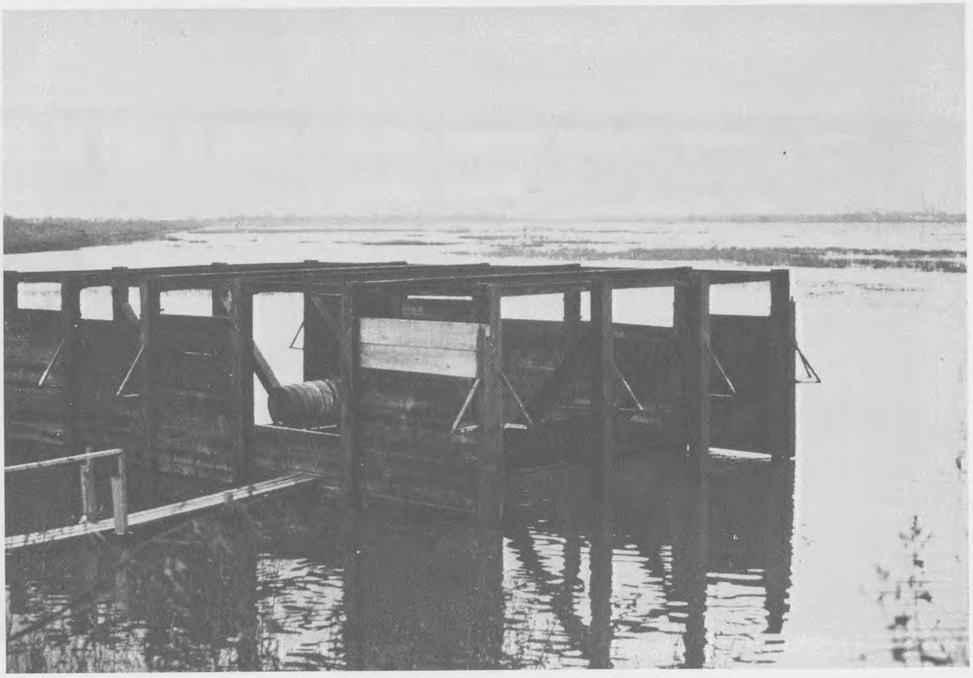

Photo 4. Houston, Tex., disposal area; view from effluent sluice looking toward influent discharge area

Photo 5. Grand Haven, Mich., disposal area; postdisposal view of containment area showing solid waste within the site and stockpiles of limestone, coal, and salt in the background beyond the dike. Influent pipe is shown to the upper left

Photo 6. Grand Haven, Mich., disposal area; view of site during dredging operation and sampling showing subfreezing conditions

Photo 7. Wilmington, N. C., disposal area; view of large overland flow system showing dead and dormant vegetation

Photo 8. Wilmington, N. C., disposal area; shallow ponding in borrow pits along inside of dike adjacent to effluent sluice boxes and weirs

Photo 9. Richmond, Va., disposal area; view of overland flow in compartment 1 showing trees partly buried by sandy sediment

Photo 10. Richmond, Va., disposal area; view of ponded area in compartment 3 showing sluice used for effluent discharge

Photo 11. Lake Charles, La., disposal area; large, extensively ponded disposal area (No. 22) along the Calcasieu River

Photo 12. Lake Charles, La., disposal area; discharge over weir from disposal area No. 22

Photo 13. Seattle, Wash., disposal area; influent discharge from pneumatic pipeline dredge into compartment 1

Photo 14. Seattle, Wash., disposal area; turbid water in compartment 2 resulting from precipitation of colloidal iron complexes in the aerated surface water. Effluent pumpout was from far right-hand corner

Photo 15. Seattle, Wash., disposal area; clear water in compartment 1 resulting from inhibition of iron hydroxide precipitation by highly reduced dredged material on bottom

Photo 16. Vicksburg, Miss., disposal area; small ponded containment area with effluent sluice in foreground

Photo 17. Southport, N. C., disposal area; view of predisposal condition of area showing thick stands of Spartina patens in foreground and forest in background

Photo 18. Southport, N. C., disposal area; view of ponded area near effluent sluice showing extent of vegetation cover by dredged material during trip 1

Photo 19. Southport, N. C., disposal area; view of ponded area near effluent sluice showing extent of vegetation cover by dredged material during trip 2

Photo 20. Southport, N. C., disposal area; view of effluent collection at end of sluice discharge pipe

APPENDIX A: DREDGING LOGS

Table A1

Dredging Log for the Sampling Period at the Houston, Tex., Disposal Area*

Sampling Date	Slurry Solids Volume percent	Sediment Classification	In Situ Sediment Volume yd^3	Daily Pumping Time hours	Estimated Daily Slurry Volume yd^3	24-Hour Average Influent Flow Rate gpm
19 Nov 75	15	Silt and clay	17,330	11.1	115,850	16,250
20 Nov 75	16	Oily silt and clay	24,600	14.75	154,185	21,650
3 Dec 75	7	Silt and clay	12,540	17.6	183,815	25,800
4 Dec 75	7	Oily silt and clay	12,315	18.0	188,160	26,400

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* Data from personal communication, William Humphreys, T. L. James Co., Kenner, La., Aug 1977.

Table A2

Dredging Log for the Sampling Period at the Grand Haven, Mich., Disposal Area*

Sampling Date	In Situ Sediment Volume** yd^3	No. of Daily Hopper Loads	Estimated Daily Slurry Volume†	24-Hour Average Influent Flow Rate gpm
16 Dec 75	4880	14	19,500	2740
17 Dec 75	4395	11	17,600	2465
18 Dec 75	3730	9	14,900	2095
19 Dec 75	4670	11	18,700	2620

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* Data from personal communication, Bruce Sabol, U. S. Army Engineer District, Detroit, Aug 1977.

** Calculated from the hopper bin sediment volume.

† Based on an average water: sediment ratio of 4.

Table A3

Dredging Log for the Sampling Period at the Wilmington, N. C., Disposal Area*

Sampling Date	Sediment Classification	In Situ Sediment Volume yd^3	Daily Pumping Time hours
16 Dec 75	Mud, silt	12,520	17.6
22 Jan 76	Mud, silt	9,820	7.25
23 Jan 76	--	--	--

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* Data from personal communication, Tucker Russell, U. S. Army Engineer District, Wilmington, Aug 1977.

Table A4

Dredging Log for the Sampling Period at the Richmond, Va., Disposal Area*

Sampling Date	In Situ Sediment Volume, yd^3	Daily Pumping Time, hours
27 Jan 76	1575	7.1
19 Feb 76	3110	21.5
20 Feb 76	3235	20.75
21 Feb 76	3405	17.75

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* Data from personal communication, Rivers Wescott, U. S. Army Engineer District, Norfolk, Aug 1977.

Table A5

Dredging Log for the Sampling Period at the Lake Charles, La., Disposal Area*

Sampling Date	Slurry Solids Volume percent	Sediment Classification	In Situ Sediment Volume yd^3	Daily Pumping Time hours	Estimated Daily Slurry Volume yd^3	24-Hour Average Influent Flow Rate gpm
4 Feb 76	13	Sandy-silt	32,590	18.5	242,000	33,975
5 Feb 76	16	90 percent silt, 10 percent sand	16,300	8.0	104,600	14,690
6 Feb 76	17	90 percent silt, 10 percent sand	43,800	19.9	260,800	36,575

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* Data from personal communication, Donald Roeder American Dredging Co., Philadelphia, Pa., Aug 1977.

Table A6

Dredging Log for the Sampling Period at the Southport, N. C., Disposal Area*

Sampling Date	Sediment Classification	In Situ Sediment Volume yd^3	Daily Pumping Time hours
6 May 76	Sand, shell	7,202	19.9
7 May 76	Mud, silt, sand	24,878	21.7
17 May 76	Mud, silt	8,075	4.2
19 May 76	Mud, sand, shell	6,267	7.4
20 May 76	Mud, silt, sand	7,919	10.6

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* Data from personal communication, Tucker Russell, U. S. Army Engineer District, Wilmington, Aug 1977.

APPENDIX B: FIELD AND ANALYTICAL LABORATORY DATA

Table B1

Field Data for Influents, Effluents, and Sediments from National Lead Industries' Disposal Area No. 4 and for Background Water from the Raritan River, Sayreville, New Jersey

Sample Identification	Sample Time 24 hr	Water Temperature °C		Salinity %		Conductivity mmhos/cm		Dissolved O_2 mg/ℓ		Slurry or Sediment pH		Sediment Eh mV	
		Reps	Mean	Reps	Mean	Reps	Mean	Reps	Mean	Reps	Mean	Reps	Mean
Background surface water (2 Oct 75)	1130	--	--	9.0	9.0	10.5	10.5	--	--	--	--	--	--
(10 Feb 76)	--	--	--	8.0	8.0	13.4	13.4	--	--	6.7	6.7	--	--
Influent (2 Oct 75)	1100	--	--	12.0	12.0	15.3	15.3	--	--	--	--	--	--
Effluent (2 Oct 75)	1030	17.5	17.5	14.0	14.0	17.6	17.6	6.3	6.3	--	--	--	--
Disposal area sediment (2 Oct 75)	1100	--	--	--	--	--	--	--	--	7.1 7.1 7.2	7.15	6*	+038 (+005 to +075)**

---

* Number of replicate measurements.

** Ranges in parentheses.

Table B2

Field Data for Influents, Effluents, and Sediments from the Clinton Disposal Area and Background Water from the Houston Ship Channel, Houston, Texas

Sample Identification	Sample Time 24 hr	Water Temperature °C		Salinity, ‰		Conductivity mmhos/cm		Dissolved O_2 mg/ℓ		Slurry or Sediment pH		Sediment Eh, mV	
		Reps	Mean	Reps	Mean	Reps	Mean	Reps	Mean	Reps	Mean	Reps	Mean
Background surface water* (2 Jan 76)	0930	--	--	1.7	1.7	3.25	3.25	--	--	--	6.9 7.7**	--	--
Influent (19 Nov 75)	1500	23.0	23.0	--	--	--	--	--	--	7.35	7.35	--	--
Influent (20 Nov 75)	0930	22.0	22.0	7.3	7.3	--	--	--	--	--	--	--	--
Influent (3 Dec 75)	1145	20.0	19.0	8.0	8.0	--	--	--	--	7.1	7.05	--	--
	1530	19.0		7.5						7.1			
	1730	18.0		8.5						6.9			
Influent (4 Dec 75)	0930	19.0	19.0	8.5	8.85	--	--	--	--	7.0	6.95	--	--
	1130	18.5		9.0						6.9			
	1330	19.0		9.0						7.0			
Effluent (19 Nov 75)	1600	23.5	23.5	--	--	--	--	4.8	4.8	--	--	--	--
Effluent (20 Nov 75)	1200	21.5	21.5	7.5	7.5	--	--	10.0	10.0	--	--	--	--
Effluent (3 Dec 75)	1230	15.5	16.5	9.5	9.5	--	--	11.0	11.5	8.0	8.05	--	--
	1600	18.0		9.5				11.2		8.1			
	1700	16.5		9.5				12.3		8.1			
Effluent (4 Dec 75)	1030	16.0	16.5	10.0	9.65	--	--	8.0	5.1	7.5	7.3	--	--
	1200	16.0		9.5				7.0		7.2			
	1430	18.0		9.5				0.3		7.2			
Disposal area sediment (19 Nov 75)	1700	--	--	--	--	--	--	--	--	2†	7.0	8†	-060
(20 Nov 75)	1400	--	--	--	--	--	--	--	--	20†	7.05 (6.65 to 7.35)††	20†	-130 (-285 to +085)††

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* Surface water; parameters measured 15 Jan 77; stored at 4°C.

** Filtered sample (<0.45 μm).

† Number of replicate measurements

†† Ranges in parentheses

Table B3

Field Data for Influents and Effluents from the Verplank Coal Company Disposal Area and Background Water from the Grand River, Grand Haven, Michigan

Sample Identification	Sample Time 24 hr	Water Temperature °C		Salinity, %		Conductivity mmhos/cm		Dissolved O_2 mg/ℓ		Slurry	pH
		Reps	Mean	Reps	Mean	Reps	Mean	Reps	Mean	Reps	Mean
Background surface water*	--	4.0	3.5	--	--	0.45	0.39	11.5	11.5	7.6	7.6
(16 Dec 75)(17 Dec 75)	--	3.0				0.33		--		--	
Influent (16 Dec 75)*	--	--	--			0.58	0.58	--	--	--	--
Influent (18 Dec 75)	A.M.	1.5	1.5	0.5	0.5	0.27	0.30	--	--	--	--
		1.0		0.5		0.32					
		1.5		0.5		0.27					
Influent (19 Dec 75)	A.M.	1.0	1.0	1.0	0.85	0.60	0.53	--	--	--	--
		1.0		0.75		0.49					
		1.0		0.75		0.49					
Influent (19 Dec 75)	P.M.	1.0	1.5	1.0	0.85	0.60	0.65	--	--	--	--
		2.0		0.5		0.65					
		2.0		1.0		0.70					
Effluent (16 Dec 75)*	--	4.0	2.0	--	--	0.73	0.67	12.5	12.5	7.5	7.5
(17 Dec 75)		0		--		0.61		--		--	
Effluent (18 Dec 75)	A.M.	0	0	0.5	0.5	0.27	0.29	--	--	--	--
		0		0.5		0.30					
		0		0.5		0.29					
Effluent (19 Dec 75)	A.M.	0	0	0.5	0.5	0.30	0.31	--	--	--	--
		0		0.5		0.32					
		0		0.5		0.30					
Effluent (19 Dec 75)	P.M.	0.5	1.0	1.0	0.85	0.35	0.37	--	--	--	--
		1.5		0.5		0.37					
		1.5		1.0		0.39					

* Samples collected by the Detroit District, CE.

Table B4

Field Data for Influents, Effluents, and Sediments from the Eagle Island Disposal Area and Background Water from the Anchorage Basin in the Cape Fear River, Wilmington, North Carolina

Sample Identification	Sample Time 24 hr	Water Temperature °C Reps	Water Temperature °C Mean	Salinity, % Reps	Salinity, % Mean	Conductivity mmhos/cm Reps	Conductivity mmhos/cm Mean	Dissolved O_2 mg/ℓ Reps	Dissolved O_2 mg/ℓ Mean	Slurry or Sediment pH Reps	Slurry or Sediment pH Mean	Sediment Eh, mV Reps	Sediment Eh, mV Mean
Background surface water (22 Jan 26)	--	--	--	--	--	--	--	--	--	5.5	5.5	--	--
Influent (16 Dec 75)	1300	12.1	12.8	3.8	4.3	5.0	5.7	0.6	0.6	7.0	7.0	--	--
	--	13.0		4.2		6.0		0.6		7.0			
	1430	13.4		4.9		6.05		0.6		7.0			
Influent (22 Jan 76)	1300	6.0	6.3	2.0	2.15	2.00	2.05	5.5	5.45	6.5	6.25	--	--
	1515	6.5		2.3		2.07		5.4		6.0			
Influent (23 Jan 76)	0830	6.5	6.5	1.75	1.9	1.90	1.85	1.0	0.8	6.5	6.5	--	--
	1115	6.5		2.0		1.80		0.6		6.5			
Effluent (16 Dec 75)	0830	13.0	13.3	--	--	7.0	6.35	0.6	0.6	7.3	7.3	--	--
	--	13.5				6.0		0.6		7.3			
	1130	13.5				6.0		0.6		7.3			
Effluent (22 Jan 76)	1030	6.8	6.8	2.5	2.5	2.85	2.85	4.0	4.0	7.5	7.5	--	--
Effluent (23 Jan 76)	1145	7.4	9.5	2.4	2.65	2.85	3.00	1.1	2.05	7.35	7.45	--	--
	--	11.5		2.9		3.15		3.0		7.5			
	--							2.0					
Disposal area sediment (16 Dec 75)	--	--	--	--	--	--	--	--	--	20*	6.6	18*	+055
Disposal area sediment (23 Jan 76)	--	--	--	--	--	--	--	--	--	8*	7.1	8*	+075
										(5.0 to 7.5)**		(-180 to +370)**	

* Number of replicate measurements
** Ranges in parentheses

Table B5

Field Data for Influents, Effluents, and Sediments from the Deepwater Terminal Disposal Area and Background Water from the James River, Richmond, Virginia

Sample Identification	Sample Time 24 hr	Water Temperature °C		Salinity, %		Conductivity mmhos/cm		Dissolved O_2 mg/ℓ		Slurry pH		Sediment Eh, mV	
		Reps	Mean	Reps	Mean	Reps	Mean	Reps	Mean	Reps	Mean	Reps	Mean
Background surface water (27 Jan 76)	--	--	--	0	0	0.18	0.18	--	--	--	6.7	--	--
(19 Feb 76)	--	--				--		--		6.7		--	
Influent (27 Jan 76)	A.M.	6.0	6.0	0	0	0.075	0.085	8.0	10.0	--	--	--	--
	A.M.	6.0		0		0.10		12.0					
	A.M.	6.0		0		0.085		10.0					
Influent (19 Feb 76)	1100	12.0	12.2	0	0	~ 0	~ 0	7.8	8.6	6.2	6.65	--	--
	1230	12.0		0		~ 0		9.0		6.8			
	1300	12.5		0		~ 0		9.1		7.0			
Influent (20 Feb 76)	1130	12.0	12.0	0	0	~ 0	~ 0	9.2	9.3	--	--	--	--
	1230	12.0		0		~ 0		9.3					
	1300	12.0		0		~ 0		9.5					
Influent (21 Feb 76)	1130	12.0	12.0	1.0	1.0	1.2	1.07	2.6	3.1	6.7	6.7	--	--
	1215	12.0		1.0		1.0		0.6					
	1300	12.0		0.9		1.0		6.2					
Effluent (27 Jan 76)*	P.M.	8.0	8.0	0	0	0.195	0.195	--	--	--	--	--	--
Effluent (19 Feb 76)	1330	12.0	11.8	0	0	~ 0	~ 0	9.5	9.6	7.2	7.15	--	--
	1400	12.0		0		~ 0		9.6		7.3			
	1430	11.5		0		~ 0		9.7		6.9			
Effluent (20 Feb 76)	1415	9.0	9.5	1.0	1.2	1.75	1.75	10.4	10.4	--	--	--	--
	1430	9.5		1.25		1.75		10.4					
	1515	10.0		1.25		1.75		10.3					
Effluent (21 Feb 76)	1430	10.0	10.0	3.5	3.7	4.4	4.5	6.0	5.8	7.2	7.2	--	--
	1500	10.0		3.75		4.5		6.0					
	1545	10.0		3.75		4.6		5.5					
Disposal area sediment (20 Feb 76)	--	--	--	--	--	--	--	--	--	--	--	7**	+319 (+020 to +505)†

* Collected at second crossdike of 3 compartment disposal area.

** Number of replicate measurements

† Ranges in parentheses

Table B6

Field Data for Influents, Effluents, and Sediments from the Disposal Area No. 22 and Background Water from the Calcasieu River near Lake Charles, Louisiana

Sample Identification	Sample Time 24 hr	Water Temperature °C		Salinity, %		Conductivity mmhos/cm		Dissolved O_2 mg/ℓ		Slurry or Sediment pH		Sediment Eh, mV	
		Reps	Mean	Reps	Mean	Reps	Mean	Reps	Mean	Reps	Mean	Reps	Mean
Background surface water (4 Feb 76)	--	--	--	4.2	4.2	6.8	6.8	--	--	6.3	6.3	--	--
Influent (4 Feb 76)	1000	13.5	14.0	15.0	15.4	19.5	20.35	4.4	4.55	7.2	7.2	--	--
	--	14.5		15.75		21.0		5.5					
	1100	14.0		15.5		20.5		3.8					
Influent (5 Feb 76)	1200	13.0	13.0	15.5	15.5	20.0	19.0	9.0	8.05	7.0	7.0	--	--
	--	13.0		15.0		19.0		8.1					
	1400	13.0		16.0		21.0		7.0					
Influent (6 Feb 76)	1145	15.0	15.0	17.0	16.85	23.0	23.0	2.0	2.1	7.3	7.3	--	--
	--	15.0		17.0		23.0		2.4					
	1415	15.0		16.5		23.0		1.9					
Effluent (4 Feb 76)*	1200	16.5	16.5	17.25	17.15	24.0	23.75	1.4	1.5	6.9	6.9	--	--
	1500	16.5		17.0		23.5		1.6					
Effluent (5 Feb 76)	0900	21.0	21.0	22.0	20.5	28.0	28.25	6.0	5.75	7.4	7.4	--	--
	1000	21.0		19.0		28.5		5.5					
Effluent (6 Feb 76)	0930	14.0	14.5	18.0	18.25	23.0	23.9	3.8	3.8	7.45	7.45	--	--
	--	15.0		18.5		24.7		4.0					
	1130	14.5		18.25		24.0		3.6					
Disposal area sediment (4 Feb 76)	--	--	--	--	--	--	--	--	--	6**	7.2 (6.9 to 7.5)†	9**	+078 (-255 to +245)†

* Collected at discharge pipes after short-circuit flow of only about 300 yd.

** Number of replicate measurements

† Ranges in parentheses

Table B7a

Field Data for the Duwamish Waterway Disposal Area Pond No. 1 Effluent and Sediment, Seattle, Washington

Date Collected	Effluent								Sediment			
	Conductivity mmhos/cm		Dissolved O_2 mg/ℓ		Temperature °C		pH		Eh, mV		pH	
	Reps	Mean	Reps	Mean	Reps	Mean	Reps	Mean	Reps	Mean	Reps	Mean
15 Mar 76												
16 Mar 76												
17 Mar 76												
18 Mar 76												
19 Mar 76												
20 Mar 76												
21 Mar 76												
22 Mar 76												
23 Mar 76												
24 Mar 76	6	35.0	6	1.7	6	8.2	6	5.8	--	--	--	--
25 Mar 76	1	38.0	1	3.0	1	7.0	1	8.2	--	--	--	--
26 Mar 76	16	25.9	3	5.2	16	6.9	16	5.8	--	--	--	--
27 Mar 76	2	25.0	1	7.5	2	6.0	1	5.7	--	--	--	--
28 Mar 76	--	--	--	--	--	--	9	5.7	--	--	--	--
29 Mar 76	9	23.8	9	7.2	9	6.8	--	--	--	--	--	--
30 Mar 76*	5	33.0	3	7.9	5	7.5	--	--	--	--	--	--
31 Mar 76												
1 Apr 76												
2 Apr 76												
3 Apr 76												
4 Apr 76												
5 Apr 76	--	--	--	--	--	--	--	--	32	-093 (-260 to +125)**	11	7.9 (6.9 to 9.05)**

* End of dredging operation.
** Ranges in parentheses.

Table B7b

Field Data for the Duwamish Waterway Disposal Area Pond No. 2

Effluent and Sediment, Seattle, Washington

Date Collected	Effluent								Sediment			
	Conductivity mmhos/cm		Dissolved O_2 mg/ℓ		Temperature °C		pH		Eh, mV		pH	
	Reps	Mean	Reps	Mean	Reps	Mean	Reps	Mean	Reps	Mean	Reps	Mean
15 Mar 76	14	25.6	13	8.1	14	10.2	14	7.5	--	--	--	--
16 Mar 76	24	29.3	24	7.5	24	10.8	13	7.5	--	--	--	--
17 Mar 76	22	31.5	24	7.0	11	10.8	--	--	--	--	--	--
18 Mar 76	24	31.6	24	6.7	--	--	--	--	--	--	--	--
19 Mar 76	24	31.5	24	6.6	--	--	--	--	--	--	--	--
20 Mar 76	24	33.8	24	7.1	--	--	--	--	--	--	--	--
21 Mar 76	24	33.2	24	5.9	--	--	--	--	--	--	--	--
22 Mar 76	24	37.0	24	5.5	--	--	--	--	--	--	--	--
23 Mar 76	24	37.9	24	5.0	--	--	--	--	--	--	--	--
24 Mar 76	24	40.2	24	6.1	--	--	--	--	--	--	--	--
25 Mar 76	21	39.8	21	6.6	12	11.2	12	8.0	--	--	--	--
26 Mar 76	24	41.5	24	7.5	24	11.2	24	7.6	--	--	--	--
27 Mar 76	24	39.8	24	7.8	24	10.3	24	7.5	--	--	--	--
28 Mar 76	24	40.0	24	6.9	24	11.0	24	7.5	--	--	--	--
29 Mar 76	16	36.1	16	7.0	16	10.8	16	7.9	--	--	--	--
30 Mar 76	--	--	--	--	--	--	--	--	--	--	--	--
31 Mar 76	--	--	--	--	--	--	--	--	--	--	--	--
1 Apr 76*	--	--	--	--	--	--	--	--	--	--	--	--
2 Apr 76*	--	--	--	--	--	--	--	--	--	--	--	--
3 Apr 76*	8	41.1	8	3.8	8	13.0	8	7.8	--	--	--	--
4 Apr 76*	22	39.9	21	3.7	12	12.4	--	--	--	--	--	--
5 Apr 76*	8	41.6	--	--	--	--	--	--	8	+271	3	6.1
									(+105 to +465)**		(5.75 to 6.45)**	

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* Pump-down of Pond No. 2 to about half of initial volume.
** Ranges in parentheses.

Table B7c

Field Data for the Duwamish Waterway Background Water Samples, Seattle, Washington

Date Collected	Depth m	Daily Reps	Conductivity mmhos/cm	Dissolved O_2 mg/ℓ	Temperature °C	pH
18 March 76	0	6	15.2	8.9	8.1	7.35
	10	6	42.3	8.0	7.5	7.85
22 March 76	0	7	11.9	8.3	7.6	7.3
	10	7	41.9	7.4	7.8	7.75
23 March 76	0	7	15.3	7.7	7.7	7.3
	10	5	41.3	7.1	7.8	7.7

Table B8

Field Data for Influents, Effluents, and Sediments from the Waterways Experiment Station Disposal Area and Background Water from Brown Lake, Vicksburg, Mississippi

Sample Identification	Sample Time 24 hr	Water Temperature °C		Salinity, %		Conductivity mmhos/cm		Dissolved O_2 mg/ℓ		Slurry or Sediment pH		Sediment Eh, mV	
		Reps	Mean	Reps	Mean	Reps	Mean	Reps	Mean	Reps	Mean	Reps	Mean
Influent, 17 Mar 76	1000	14.0	14.5	0.25	0.25	0.48	0.50	5.8	2.5	6.9	6.85	--	--
	1245	15.0		0.25		0.49		1.0		6.8			
	1300	15.0		0.25		0.52		0.7		6.8			
Influent, 15 Apr 76	1415	23.0	22.5	0	0.05	0.80	0.79	3.3	3.45	7.15	7.2	--	--
	1430	22.0		0		0.80		3.4		7.4			
	1445	22.5		0		0.78		3.6		7.05			
Effluent, 17 Mar 76	0930	12.5	15.8	0.25	0.25	0.45	0.48	1.2	1.15	7.5	7.3	--	--
	0945	13.0		0.25		0.46		0.8		--			
	1030	14.5		0.25		0.48		0.6		6.6			
	1330	23.0		0.30		0.53		2.0		7.8			
Effluent, 9 Apr 76	1345	29.0	29.0	0	0	0.80	0.73	2.6	3.0	--	--	--	--
	1415	29.0		0		0.79		2.5					
	1430	29.0		0		0.60		3.8					
Effluent 15 Apr 76	1500	34.0	33.5	0.1	0.05	0.80	0.88	3.2	3.05	8.2	8.1	--	--
	1515	34.0		0		0.90		2.9		8.1			
	1530	32.0		0		0.92		3.0		8.05			
Disposal area sediment													
23 Mar 76	--	--	--	--	--	--	--	--	--	8*	6.8	6*	-110
25 Mar 76	--	--	--	--	--	--	--	--	--	2*	7.0	5*	-080
29 Mar 76	--	--	--	--	--	--	--	--	--	4*	6.9	6*	-040
31 Mar 76	--	--	--	--	--	--	--	--	--	--	--	5*	+065
										(6.5 to 7.05)**		(-295 to +155)**	

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* Number of replicate measurements

** Ranges in parentheses

Table B9a

Field Data for Influents and Effluents from the Oak Island Disposal Area and Background Water from the Atlantic Intracoastal Waterway, Initial Trip, Southport, North Carolina

Sample Identification	Sample Time 24 hr	Water Temperature °C		Salinity, %		Conductivity mmhos/cm		Dissolved O_2 mg/ℓ		Slurry, pH	
		Reps	Mean	Reps	Mean	Reps	Mean	Reps	Mean	Reps	Mean
Background surface water, 6 May 76	--	--	--	21.0	21.0	31.5	31.5	--	--	--	--
Influent, 6 May 76	1115	21.0	21.0	17.8	17.8	27.0	27.0	2.0	1.85	8.0	8.0
	1230	21.0						1.8			
	1300	21.0						1.8			
Influent, 6 May 76	1430	21.0	21.0	20.8	20.8	31.0	31.0	1.9	3.25	8.3	8.3
	1500	21.0						3.8			
	1530	21.0						4.1			
Influent, 7 May 76	0730	22.0	21.5	21.5	21.5	32.1	32.1	3.1	3.15	7.4	7.4
	0800	21.0						3.6			
	0815	21.0						2.8			
Effluent, 6 May 76	1630	25.0	25.0	20.6	20.6	31.0	31.0	3.5	3.5	7.6	7.6
	1645	25.0						3.5			
	1700	25.0						3.5			
Effluent, 7 May 76	0700	24.0	24.0	21.7	21.7	32.5	32.5	2.4	2.4	7.6	7.6
	0715	24.0						2.4			
	0730	24.0						2.4			
Effluent, 7 May 76	0900	23.0	23.0	21.6	21.6	32.5	32.5	3.8	3.95	7.7	7.7
	0915	23.0						4.4			
	0930	23.0						3.6			

Table B9b

Field Data for Influents, Effluents, and Sediments from the Oak Island Disposal Area and Background Water from the Atlantic Intracoastal Waterway, Final Trip, Southport, North Carolina

Sample Identification	Sample Time 24 hr	Water Temperature °C		Salinity, %		Conductivity mmhos/cm		Dissolved O_2 mg/ℓ		Slurry or Sediment pH		Sediment Eh, mV	
		Reps	Mean	Reps	Mean	Reps	Mean	Reps	Mean	Reps	Mean	Reps	Mean
Influent, 17 May 76	1215	23.0	23.7	16.0	13.35	27.0	22.6	8.5	6.95	--	--	--	--
	1245	24.0		12.0		21.0		6.2					
	1345	24.0		12.0		19.8		6.2					
Influent, 19 May 76	1430	20.0	22.0	16.8	15.25	26.0	22.65	0.8	1.15	--	--	--	--
	1500	22.0		17.7		28.8		0.8					
	1615	23.0		16.0		19.1		0.8					
	1645	23.0		10.5		16.7		2.2					
Influent, 20 May 76	1030	21.0	21.4	18.2	17.05	28.0	25.85	0.8	0.25	--	7.55	--	--
	1100	21.0		17.0		25.5		0.2		--			
	1130	21.0		16.6		25.1		0.0		7.3			
	1230	22.5		16.4		24.8		0.0		7.8			
Effluent, 17 May 76	1425	34.0	34.0	22.0	22.0	39.0	39.0	4.2	4.7	--	--	--	--
	1445							5.0					
	1500							4.8					
Effluent, 17 May 76	1645	30.0	30.0	20.0	20.0	36.0	36.0	6.6	6.6	--	--	--	--
Effluent, 20 May 76	0930	22.0	27.3	22.1	20.8	33.1	39.2	8.2	7.0	7.8	7.9	--	--
	1245	30.0		19.8		42.0		6.8		8.0			
	1400	30.0		20.5		42.5		6.0		--			
Disposal area sediment 21 May 76	1000	--	--	--	--	--	--	--	--	11* (6.9 to 7.7)**	7.25	24* (-410 to -130)**	-330
	1500												

* Number of replicate measurements

** Ranges in parentheses

Table B10

Chemical Composition of National Lead Industries' Disposal Area No. 4 Influents and Effluents, and Raritan River Background Water, Sayreville, New Jersey

Sample Identification	Oil and Grease		Organic-N††		Ammonium-N		$NO_3 + NO_2$ -N	Total-P		Alkalinity as $CaCO_3$	Chloride
	Total mg/ℓ	<0.45 µm mg/ℓ	Total mg/ℓ	<0.45 µm mg/ℓ	Total mg/ℓ	<0.45 µm mg/ℓ	<0.45 µm mg/ℓ	Total mg/ℓ	<0.45 µm mg/ℓ	mg/ℓ	mg/ℓ
Background water (10-2-75) (surface water)	9.2	--	1.6	--	1.97	<1.0	0.22	2.81	0.13	500	4080
Influent sediment* (10-2-75)	--	--	758	--	58.8	--	--	802	--	--	--
Influent water** (10-2-75)	8.4	5.5	<1.0	--	1.51	<1.0	0.28	1.74	0.10	680	6220
Effluent-cross-dike weir (10-2-75)	15.6	10.9	2.55	--	21.4	9.24	0.48	1.93	0.12	600	7500
Effluent sediment† (10-2-75)	--	--	95.5	--	23.3	--	--	527.0	--	--	--
Effluent-final sluice (10-2-75)	12.2	--	2.55	--	22.4	16.8	0.36	1.84	0.10	840	7500

	Iron			Manganese			Zinc			Titanium		
	Total mg/ℓ	Solids mg/kg§	<0.45 µm mg/ℓ	Total mg/ℓ	Solids mg/kg§	<0.45 µm mg/ℓ	Total mg/ℓ	Solids mg/kg§	<0.45 µm mg/ℓ	Total mg/ℓ	Solids mg/kg§	<0.45 µm mg/ℓ
Background water (10-2-75)	--	--	0.073	--	--	0.0043	--	--	0.0065	--	--	0.232
Influent sediment* (10-2-75)	--	13,500	0.054	--	754	0.0045	--	297	0.0108	--	23.3	0.225
Influent water** (10-2-75)	--	--	0.057	--	--	0.0039	--	--	0.0079	--	--	0.215
Effluent-cross-dike weir (10-2-75)	--	--	0.064	--	--	0.0069	--	--	0.0116	--	--	0.246
Effluent sediment† (10-2-75)	--	--	0.059	--	--	0.0075	--	--	0.0070	--	--	0.222
Effluent-final sluice (10-2-75)	28.2	--	0.074	23.6	--	0.0038	1.01	--	0.0079	0.35	--	0.235

	Cadmium			Copper			Nickel			Lead		
	Total mg/ℓ	Solids mg/kg§	<0.45 µm mg/ℓ	Total mg/ℓ	Solids mg/kg§	<0.45 µm mg/ℓ	Total mg/ℓ	Solids mg/kg§	<0.45 µm mg/ℓ	Total mg/ℓ	Solids mg/kg§	<0.45 µm mg/ℓ
Background water (10-2-75)	--	--	0.00035	--	--	0.0057	--	--	0.0037	--	--	0.0035
Influent sediment* (10-2-75)	--	19.8	0.00056	--	198	0.0076	--	19.8	0.0083	--	198	0.0045
Influent water** (10-2-75)	--	--	0.00046	--	--	0.0065	--	--	0.0028	--	--	0.0037
Effluent-cross-dike weir (10-2-75)	--	--	0.00038	--	--	0.0059	--	--	0.0046	--	--	0.0064
Effluent sediment† (10-2-75)	--	--	0.00056	--	--	0.0078	--	--	0.0039	--	--	0.0039
Effluent-final sluice (10-2-75)	0.75	--	0.00039	3.38	--	0.0065	0.34	--	0.0053	3.38	--	0.0047

	Mercury			Chromium			Arsenic			Vanadium		
	Total mg/ℓ	Solids mg/kg§	<0.45 µm mg/ℓ	Total mg/ℓ	Solids mg/kg§	<0.45 µm mg/ℓ	Total mg/ℓ	Solids mg/kg§	<0.45 µm mg/ℓ	Total mg/ℓ	Solids mg/kg§	<0.45 µm mg/ℓ
Background water (10-2-75)	0.0163	--	0.0007	--	--	0.0029	--	--	0.00035	--	--	0.039
Influent sediment (10-2-75)	0.620	--	--	--	79.3	0.0036	--	3.32	0.00064	--	94.7	0.040
Influent water** (10-2-75)	0.0099	--	0.0007	--	--	0.0032	--	--	0.00047	--	--	0.066
Effluent-cross-dike weir (10-2-75)	0.0255	--	0.0009	--	--	0.0045	--	--	0.00016	--	--	0.046
Effluent sediment (10-2-75)	0.920	--	--	--	--	0.0048	--	--	0.00036	--	--	0.058
Effluent-final sluice (10-2-75)	0.0108	--	0.0009	0.34	--	0.0038	0.034	--	0.00023	0.68	--	0.047

Note: Dashes (--) indicate missing data.
* Fine-grained sediment collected 5 meters from influent discharge pipe.
** Ponded water, collected from immediately beneath discharge pipe.
† Fine-grained sediment collected within the disposal area, adjacent to the final effluent sluice box.
†† TKN - (<0.45 µm) ammonium-N.
§ Dry weight.

Table B11

Chemical Composition of the Clinton Disposal Area Influents and Effluents, and Houston Ship Channel Background Water, Houston, Texas

Sample Identification	Total Organic Carbon		Oil and Grease		Organic-N†		Ammonium-N		$NO_3 + NO_2$ -N	Total-P	
	Solids mg/kg*	<0.45 µm mg/ℓ**	Total mg/ℓ	<0.45 µm mg/ℓ	Total mg/ℓ	<0.45 µm mg/ℓ	Total mg/ℓ	<0.45 µm mg/ℓ	<0.45 µm mg/ℓ	Total mg/ℓ	<0.45 µm mg/ℓ
Background water (1-2-76) (surface water)	--	43	47.2	1.1	0.38	0.19	--	0.20	0.68	0.86	0.50
Influent (11-19-75)	7,360	19	644	--	19.9	3.9	7.30	4.20	0.20	75.7	0.54
Influent (11-20-75)	22,400	52	508	--	44.9	3.35	64.6	58.3	0.17	318	0.20
Influent (12-3-75)	13,900	172	555	--	67.6	9.8	70.5	71.7	0.17	250	0.11
Influent (12-4-75)	25,100	185	766	--	51.3	11.75	86.0	68.2	0.22	113	0.13
Effluent (11-19-75)	--	46	21.7	--	5.05	2.1	41.8	31.8	0.21	40.0	1.11
Effluent (11-20-75)	--	55	51.2	--	2.4	0.8	34.4	38.5	0.22	40.5	1.38
Effluent (12-3-75)	--	120	25.0	--	7.0	1.4	58.8	57.7	0.25	37.0	1.53
Effluent (12-4-75)	29,820	54	196	--	11.05	<0.2	80.3	70.9	0.21	75.8	0.17

	Alkalinity as $CaCO_3$	Chloride	Sulfate	Calcium		Magnesium		Potassium		Sodium	
	mg/ℓ	mg/ℓ	mg/ℓ	Total mg/ℓ	<0.45 µm mg/ℓ	Total mg/ℓ	<0.45 µm mg/ℓ	Total mg/ℓ	<0.45 µm mg/ℓ	Total mg/ℓ	<0.45 µm mg/ℓ
Background water (1-2-76) (surface water)	92	1040	174	42.5	35.3	53.6	32.9	144	31.6	1465	880
Influent (11-19-75)	244	4900	--	1810	152	1895	272	797	118	2000	2100
Influent (11-20-75)	496	4610	--	1815	138	1430	289	1230	141	2045	2610
Influent (12-3-75)	698	5250	--	2055	150	1095	340	2710	126	3605	2555
Influent (12-4-75)	658	5660	--	2595	150	1770	354	1210	182	3650	2485
Effluent (11-19-75)	387	4510	--	217	160	576	275	1045	130	1960	2480
Effluent (11-20-75)	417	4950	--	224	158	598	286	1110	151	2300	2130
Effluent (12-3-75)	600	5200	--	220	141	688	335	1145	160	2945	2640
Effluent (12-4-75)	662	5660	--	267	151	550	372	580	181	3660	2530

(Continued)

Note: Dashes (--) indicate missing data.

* Dry weight.

** Unfiltered clear supernatant after several weeks of settling.

† TKN - (0.45 µm) ammonium-N.

Table B11 (Concluded)

	Iron		Manganese		Zinc		Cadmium		Copper	
Sample Identification	Total mg/ℓ	<0.45 μm mg/ℓ	Total mg/ℓ	<0.45 μm mg/ℓ	Total mg/ℓ	<0.45 μm mg/ℓ	Total mg/ℓ	<0.45 μm mg/ℓ	Total mg/ℓ	<0.45 μm mg/ℓ
Background water (1-2-76) (surface water)	63.6	0.006	0.07	0.030	0.016	0.0007	0.005	0.00035	0.160	0.0035
Influent (11-19-75)	683	0.056	49.2	0.0038	--	--	1.095	0.00061	8.15	0.0058
Influent (11-20-75)	699	0.043	34.2	0.0036	--	--	1.095	0.00037	10.6	0.0029
Influent (12-3-75)	950	0.069	39.0	0.0038	--	--	1.365	0.00066	14.05	0.0065
Influent (12-4-75)	1337	0.081	49.0	0.0055	--	--	1.060	0.00070	18.2	0.0070
Effluent (11-19-75)	547	0.066	37.9	0.0027	--	--	0.095	0.00030	0.665	0.0041
Effluent (11-20-75)	900	0.045	48.5	0.0046	--	--	0.110	0.00054	1.30	0.0069
Effluent (12-3-75)	711	0.046	31.0	0.0025	--	--	0.117	0.00029	1.25	0.0050
Effluent (12-4-75)	1043	0.061	40.9	0.0037	--	--	0.116	0.00019	1.59	0.0039

	Nickel		Lead		Titanium		Vanadium		Arsenic	
	Total mg/ℓ	<0.45 μm mg/ℓ	Total mg/ℓ	<0.45 μm mg/ℓ	Total mg/ℓ	<0.45 μm mg/ℓ	Total mg/ℓ	<0.45 μm mg/ℓ	Total mg/ℓ	<0.45 μm mg/ℓ
Background water (1-2-76) (surface water)	1.50	0.0060	0.018	0.0002	0.0105	0.0001	0.32	0.0040	0.013	0.00080
Influent (11-19-75)	3.7	0.0048	57.4	0.0048	3.1	0.0270	2.40	0.0247	0.247	0.00024
Influent (11-20-75)	4.75	0.0047	55.9	0.0046	2.1	0.0251	4.75	0.0241	0.181	0.00011
Influent (12-3-75)	8.8	0.0060	77.0	0.0065	3.7	0.0381	4.75	0.0259	0.289	0.00048
Influent (12-4-75)	11.0	0.0083	86.5	0.0069	4.35	0.0275	5.23	0.0390	0.348	0.00059
Effluent (11-19-75)	0.285	0.0037	4.75	0.0039	0.50	0.0206	0.24	0.0163	0.089	0.00010
Effluent (11-20-75)	0.62	0.0054	6.89	0.0058	0.41	0.0350	0.37	0.0266	0.222	0.00025
Effluent (12-3-75)	0.63	0.0043	6.43	0.0054	0.255	0.0362	0.23	0.0206	0.153	0.00014
Effluent (12-4-75)	0.79	0.0064	7.57	0.0056	0.27	0.0204	0.265	0.0261	0.127	0.00008

Table B12

Chemical Composition of Verplank's Coal and Dock Co. Disposal Area Influents and Effluents, and Grand River Background Water, Grand Haven, Michigan

Sample Identification	Total Carbon <0.45 µm mg/ℓ	Organic Carbon <0.45 µm mg/ℓ	Oil and Grease Total mg/ℓ	Oil and Grease <0.45 µm mg/ℓ	Chlorinated Pesticide Residues* op'DDE mg/ℓ	pp'DDE mg/ℓ	op'DDT mg/ℓ	pp'DDT mg/ℓ	op'DDD mg/ℓ	pp'DDD mg/ℓ	Total PCB's (1242, 1254, 1260) mg/ℓ
Background water (12-19-76) (surface water)	58	31	6.4	1.1	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.1
Influent (12-17-76)**	--	--	--	--	--	--	--	--	--	--	6.00
Influent (12-18-76)†	30	26	29.0	2.1	0.07	0.16	1.39	1.04	0.45	1.04	21.00
Influent (12-19-76)†	55	19	25.0	3.5	--	--	--	--	--	--	--
Influent (12-19-76)†	85	39	64.5	2.0	0.13	0.38	0.11	0.37	<0.01	<0.01	5.00
Effluent (12-17-76)**	--	--	--	--	--	--	--	--	--	--	<0.1
Effluent (12-18-76)	65	20	5.5	1.1	--	--	--	--	--	--	--
Effluent (12-19-76)	60	22	6.5	2.0	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.1
Effluent (12-19-76)	70	42	22.5	3.0	0.14	0.41	0.23	0.96	<0.01	<0.01	7.66

Sample Identification	Organic-N†† Total mg/ℓ	Organic-N†† <0.45 µm mg/ℓ	Ammonium-N Total mg/ℓ	Ammonium-N <0.45 µm mg/ℓ	NO_3 + NO_2 -N <0.45 µm mg/ℓ	Total-P Total mg/ℓ	Total-P <0.45 µm mg/ℓ	Total Alkalinity as $CaCO_3$ mg/ℓ	Chloride <0.45 µm mg/ℓ	Sulfate <0.45 µm mg/ℓ	Sulfide <0.45 µm mg/ℓ
Background water (12-1-76) (surface water)	0.60	0.65	<0.2	<0.2	1.32	0.76	<0.02	140	21	37.8	<0.05
Influent (12-18-76)†	101	3.4	35.6	31.6	0.30	24.5	0.08	247	195	53.9	<0.05
Influent (12-19-76)†	50.1	2.8	24.1	17.6	0.82	22.3	0.08	210	105	23.8	<0.05
Influent (12-19-76)†	28.1	2.2	30.9	12.4	0.65	28.0	0.03	263	150	33.7	<0.05
Effluent (12-18-76)	3.9	0.6	4.2	3.4	1.25	0.86	<0.02	184	50	27.9	<0.05
Effluent (12-19-76)	4.3	0.8	10.3	7.0	1.83	1.31	<0.02	193	70	10.6	<0.05
Effluent (12-19-76)	74.5	6.7	64.3	27.7	0.90	23.8	<0.02	276	120	124.2	<0.05

(Continued)

Note: Dashes (--) indicate missing data.

* Dieldrin, aldrin, lindane, heptachlor, heptachlor epoxide, and chlordane were below detection limits (<0.01 mg/ℓ).

** Samples collected by CE Detroit District at influent discharge pipe.

† Influent samples were collected at inflow pipe into hopper dredge.

†† TKN - (<0.45 µm) ammonium-N.

Table B12 (Concluded)

Sample Identification	Calcium		Magnesium		Potassium		Sodium		Iron	
	Total mg/ℓ	<0.45 µm mg/ℓ	Total mg/ℓ	<0.45 µm mg/ℓ	Total mg/ℓ	<0.45 µm mg/ℓ	Total mg/ℓ	<0.45 µm mg/ℓ	Total mg/ℓ	<0.45 µm mg/ℓ
Background water (12-19-76) (surface water)	56.2	42.6	23.0	13.9	338	142	16.8	9.8	5.21	0.059
Influent (12-18-76)†	130.1	66.3	277.5	181.5	1287	136.5	150.5	86.9	46.1	0.068
Influent (12-19-76)†	96.5	54.7	310.5	152.5	2373	147	161.5	96.4	55.2	0.093
Influent (12-19-76)†	95.4	56.1	265	174.5	837	178.5	143.5	95.4	63.2	0.143
Effluent (12-18-76)	82.8	45.2	27.6	13.9	166.5	94.0	125.5	35.9	38.0	0.025
Effluent (12-19-76)	57.0	45.5	22.7	14.2	850	158.5	125.5	41.0	44.2	0.378
Effluent (12-19-76)	102.2	73.3	154	18.5	747	171	117	75.3	40.6	0.038

Sample Identification	Manganese		Zinc		Cadmium		Copper		Nickel	
	Total mg/ℓ	<0.45 µm mg/ℓ	Total mg/ℓ	<0.45 µm mg/ℓ	Total mg/ℓ	<0.45 µm mg/ℓ	Total mg/ℓ	<0.45 µm mg/ℓ	Total mg/ℓ	<0.45 µm mg/ℓ
Background water (12-19-76) (surface water)	0.25	0.009	0.305	0.0040	0.0034	0.0003	0.052	0.0052	0.013	0.004
Influent (12-18-76)†	1.28	0.378	35.5	0.0109	5.09	0.0006	9.97	0.0073	9.16	0.006
Influent (12-19-76)†	0.80	0.013	57.5	0.0092	5.87	0.0008	12.7	0.0016	8.92	0.006
Influent (12-19-76)†	1.03	0.066	17.8	0.0038	7.17	0.0009	12.6	0.0034	13.2	0.008
Effluent (12-18-76)	0.72	0.015	0.15	0.0027	0.32	0.0004	0.61	0.012	0.056	0.002
Effluent (12-19-76)	0.24	0.003	0.26	0.0040	0.34	0.0005	0.43	0.0010	0.10	0.003
Effluent (12-19-76)	0.21	0.069	4.67	0.0175	0.37	0.0004	0.47	0.0006	1.02	0.010

Sample Identification	Lead		Chromium		Vanadium		Arsenic	
	Total mg/ℓ	<0.45 µm mg/ℓ	Total mg/ℓ	<0.45 µm mg/ℓ	Total mg/ℓ	<0.45 µm mg/ℓ	Total mg/ℓ	<0.45 µm mg/ℓ
Background water (12-19-76) (surface water)	0.049	0.0002	0.013	0.003	0.029	0.004	0.004	0.0003
Influent (12-18-76)†	13.1	0.0002	56.7	0.004	2.29	0.004	2.47	0.00025
Influent (12-19-76)†	25.1	0.0003	76.6	0.003	2.74	0.006	5.15	0.00045
Influent (12-19-76)†	7.68	0.0003	58.1	0.005	2.45	0.005	6.02	0.0004
Effluent (12-18-76)	0.79	0.0005	0.14	0.006	0.14	0.0025	0.405	0.00025
Effluent (12-19-76)	1.36	0.0002	0.079	0.004	0.076	0.0035	0.31	0.00025
Effluent (12-19-76)	0.575	0.0001	0.58	0.004	0.47	0.011	0.235	0.00045

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† Influent samples were collected at inflow pipe into hopper dredge.

Table B13

Physical and Chemical Composition of the Eagle Island Disposal Area Influents and Effluents and Cape Fear River Anchorage Basin Background Water, Wilmington, North Carolina

Sample Identification	Particle Size (Mechanical) % <2 µm	Particle Size (Mechanical) % 2-50 µm	Particle Size (Mechanical) % >50 µm	Coulter Counter Particle Size, µm* >50%	Coulter Counter Particle Size, µm* >80%	Total Solids %	Nonfilterable Solids, %	Settleable Solids mℓ/ℓ
Background water (12-15-75) (Surface water)	--	--	--	--	--	--	0.00397	--
Background water (1-22-76) (Surface water)	--	--	--	--	--	--	0.00500	<0.1
Influent (12-16-75)	65.5	34.5	0	1.05	0.6	--	9.650	550
Influent (1-22-76)	60	29	11	--	--	6.59	5.900	200
Influent (1-23-76)	55	30	15	--	--	6.73	6.030	650
Effluent (12-16-75)	--	--	--	2.0	1.2	--	0.0521	0.9
Effluent (1-22-76)	61	35	4	--	--	2.85	2.050	300
Effluent (1-23-76)	--	--	--	--	--	1.83	0.600	950

	Total Organic Carbon Total mg/ℓ	Total Organic Carbon Solids mg/kg†	Total Organic Carbon <0.45 µm mg/ℓ	Chlorinated Pesticides** op' DDE mg/ℓ	Chlorinated Pesticides** pp' DDE mg/ℓ	Total PCB (1242, 1254, 1260) mg/ℓ
Background water (12-15-75)	14	--	11	<0.01	0.01	<0.1
Background water	14	--	12	<0.01	0.01	<0.1
Influent (12-16-75)	3705	38,200	20	--	--	--
Influent (1-22-76)	3045	51,300	20	<0.01	0.09	2.56
Influent (1-23-76)	3250	53,400	31	0.15	0.61	<0.1
Effluent (12-16-75)	26	--	20	--	--	--
Effluent (1-22-76)	1055	50,200	27	0.07	0.27	<0.1
Effluent (1-23-76)	345	53,200	27	0.12	0.21	1.44

	Organic-N†† Total mg/ℓ	Organic-N†† Solids mg/kg†	Organic-N†† <0.45 µm mg/ℓ	Ammonium-N Total mg/ℓ	Ammonium-N Exchangeable‡ mg/kg† (mg/ℓ)‡‡	Ammonium-N <0.45 µm mg/ℓ
Background water (12-15-75)	0.1	--	0.1	0.72	--	0.41
Background water (1-22-76)	0.2	--	0.5	1.03	--	0.82
Influent (12-16-75)	376	3870	2.4	--	339 (32.7)	30.0
Influent (1-22-76)	168	2838	1.5	--	193 (11.4)	12.7
Influent (1-23-76)	182	3055	--	--	223 (13.4)	28.8
Effluent (12-16-75)	7.0	--	2.5	11.4	--	14.8
Effluent (1-22-76)	56.8	2713	1.8	--	458 (9.4)	27.5
Effluent (1-23-76)	19.6	3042	1.5	--	--	25.6

	$NO_3 + NO_2$ -N <0.45 µm mg/ℓ	Orthophosphate-P Total mg/ℓ	Orthophosphate-P <0.45 µm mg/ℓ	Total-P Total mg/ℓ	Total-P Solids mg/kg†	Total-P <0.45 µm mg/ℓ
Background water (12-15-75)	0.29	0.24	0.07	0.31	--	0.09
Background water (1-22-76	0.04	0.32	0.10	0.38	--	0.10
Influent (12-16-75)	<0.01	--	0.09	420	4350	0.10
Influent (1-22-76)	<0.01	--	0.17	260	4400	0.19
Influent (1-23-76)	<0.01	--	0.41	248	4100	0.49
Effluent (12-16-75)	0.50	0.90	0.06	1.11	--	0.09
Effluent (1-22-76)	<0.01	--	0.06	82.1	4000	0.14
Effluent (1-23-76)	0.34	--	0.20	23.0	3800	0.20

(Continued)

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* Influents: determined for <2-µm fraction from mechanical particle-size analysis, sodium hexametaphosphate was added for dispersion; effluents: determined for total effluent, no dispersive agent added; dispersed by sonication.

** op' DDD, pp' DDD, op' DDT, pp' DDT, dieldrin, aldrin, lindane, heptachlor, heptachlor epoxide, and chlordane are below detection limits.

† Dry weight.

†† TKN - (<0.45 µm) ammonium-N.

‡ Sodium acetate extractable.

‡‡ Values in parentheses as (mg/ℓ) exchangeable ammonium-N in total sample.

Table B13 (Concluded)

Sample Identification	Alkalinity as $CaCO_3$ mg/ℓ	Chloride mg/ℓ	Total Sulfide		Cation Exchange Capacity meq/100 g†	Total C:N:P Ratio
			Total mg/ℓ	Solids mg/kg†		
Background water (12-15-75)	18.25	195	2.0	--	--	45.2:1.6:1
Background water (1-22-76)	16.27	45	--	--	--	36.8:2.7:1
Influent (12-16-75)	386.1	2250	3.3	17.8	88.2	9.1:1:1
Influent (1-22-76)	104.8	415	3.5	32.2	80.0	16.8:0.7:1
Influent (1-23-76)	544.4	845	4.55	47.8	73.5	13.1:0.9:1
Effluent (12-16-75)	270.8	1750	1.5	--	--	23.4:19.6:1
Effluent (1-22-76)	556.5	1060	3.6	94.1	120.6	12.9:1:1
Effluent (1-23-76)	613.6	1140	2.8	201.8	--	15:2:1

	Calcium			Magnesium			Potassium		
	Total mg/ℓ	Solids mg/kg†	<0.45 µm mg/ℓ	Total mg/ℓ	Solids mg/kg†	<0.45 µm mg/ℓ	Total mg/ℓ	Solids mg/kg†	<0.45 µm mg/ℓ
Background water (12-15-75)	8.9	--	11.0	13.3	--	11.0	8.9	--	11.0
Background water (1-22-76)	4.8	--	4.1	2.5	--	1.5	2.2	--	1.3
Influent (12-16-75)	349	2820	85.0	1180	10,545	180	1200	11,885	60.0
Influent (1-22-76)	469	4590	21.0	437	7,090	20.0	597	9,885	17.0
Influent (1-23-76)	612	3670	416	431	6,065	69.2	597	9,140	48.4
Effluent (12-16-75)	100	--	90.0	172	--	180	51.2	--	49.0
Effluent (1-22-76)	560	1990	530	188	4,875	90.0	226	8,330	55.8
Effluent (1-23-76)	553	3800	532	129	6,395	91.0	114	9,280	58.7

	Sodium			Iron			Manganese		
	Total mg/ℓ	Solids mg/kg†	<0.45 µm mg/ℓ	Total mg/ℓ	Solids mg/kg†	<0.45 µm mg/ℓ	Total mg/ℓ	Solids mg/kg†	<0.45 µm mg/ℓ
Background water (12-15-75)	480	--	--	3.6	--	0.285	0.104	--	0.037
Background water (1-22-76)	12.0	--	10.0	4.7	--	0.239	0.140	--	0.049
Influent (12-16-75)	--	--	1600	6070	62,800	13.8	104	1005	7.53
Influent (1-22-76)	546	4,625	290	2895	48,900	7.46	48.0	785	1.83
Influent (1-23-76)	918	5,780	606	2785	45,900	15.9	54.3	839	3.89
Effluent (12-16-75)	--	--	1300	37.2	--	0.237	6.14	--	6.30
Effluent (1-22-76)	1130	21,660	702	755	36,700	3.06	17.3	683	3.35
Effluent (1-23-76)	962	40,165	725	300	48,300	10.1	9.4	938	3.89

	Zinc			Cadmium			Copper		
	Total mg/ℓ	Solids mg/kg†	<0.45 µm mg/ℓ	Total mg/ℓ	Solids mg/kg†	<0.45 µm mg/ℓ	Total mg/ℓ	Solids mg/kg†	<0.45 µm mg/ℓ
Background water (12-15-75)	0.15	--	0.014	--	--	0.0004	0.011	--	0.004
Background water (1-22-76)	0.11	--	0.019	0.0017	--	0.0007	0.010	--	0.002
Influent (12-16-75)	19.6	202	0.065	--	--	0.0012	3.09	32.0	0.003
Influent (1-22-76)	10.8	183	0.050	0.013	0.164	0.0035	2.25	38.1	0.004
Influent (1-23-76)	11.3	188	0.010	0.005	0.048	0.0022	2.49	41.3	0.004
Effluent (12-16-75)	0.85	--	0.034	--	--	0.0006	0.020	--	0.004
Effluent (1-22-76)	3.0	146	0.0095	0.025	1.18	0.0010	0.58	28.1	0.003
Effluent (1-23-76)	1.29	183	0.216	0.003	0.383	0.0012	0.26	39.9	0.028

	Nickel			Lead			Mercury		
	Total mg/ℓ	Solids mg/kg†	<0.45 µm mg/ℓ	Total mg/ℓ	Solids mg/kg†	<0.45 µm mg/ℓ	Total mg/ℓ	Solids mg/kg†	<0.45 µm mg/ℓ
Background water (12-15-75)	--	--	0.004	--	--	<0.001	0.0012§	--	0.0042
Background water (1-22-76)	0.017	--	0.007	0.012	--	0.003	<0.0002§	--	<0.0002§
Influent (12-16-75)	3.89	38.8	<0.003	6.4	66.3	<0.001	0.110	1.07	0.0079
Influent (1-22-76)	3.19	53.2	0.007	3.4	57.6	0.012	0.009	0.15	<0.0002§
Influent (1-23-76)	3.61	57.9	0.006	3.4	56.4	<0.001	0.007	0.11	<0.0002§
Effluent (12-16-75)	0.024	--	0.006	--	--	<0.001	0.367	--	0.0062
Effluent (1-22-76)	0.91	37.4	0.006	0.90	43.9	<0.001	0.002	0.08	<0.0002§
Effluent (1-23-76)	0.65	54.3	0.004	0.30	50.0	<0.001	--	--	<0.0002§

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† Dry weight.

§ Cold vapor technique.

Table B14

Physical and Chemical Composition of Deep Water Terminal Disposal Area Influents and Effluents and James River Background Water, Richmond, Virginia

Sample Identification	Particle Size (Mechanical) % <2 µm	% 2-50 µm	% >50 µm	Coulter Counter Particle Size, µm* >50%	>80%	Total Solids %	Nonfilterable Solids, %	Settleable Solids mℓ/ℓ
Background water (1-27-76) (surface water)	--	--	--	--	--	0.0134	0.00126	<0.1
Background water (2-19-76) (surface water)	--	--	--	7.6	3.8	0.0093	0.00065	<0.1
Influent (1-27-76)	1	2	97	1.45	0.70	4.02	1.70**	140
Influent (2-19-76)	4	17	79	0.88	0.55	4.41	1.06**	45/980†
Influent (2-20-76)	13	27	60	1.05	0.60	1.94	0.58**	55/980†
Influent (2-21-76)	26	49	25	1.20	0.65	3.60	3.30	100/980†
Effluent (1-27-76)††	--	--	--	--	--	0.0389	0.0133	<0.1
Effluent (2-19-76)	--	--	--	3.0	1.7	0.0409	0.0284	0.4
Effluent (2-20-76)	--	--	--	1.9	1.05	0.0252	0.0144	0.2
Effluent (2-21-76)	--	--	--	6.4	3.0	0.1452	0.1295	2.0

	Total Organic Carbon			Total Chlorinated Pesticides‡‡		Total PCB
	Total mg/ℓ	Solids mg/kg‡	<0.45 µm mg/ℓ	op' DDE mg/ℓ	pp' DDE mg/ℓ	(1242, 1254, 1260) mg/ℓ
Background water (1-27-76)	6	--	4	--	--	--
Background water (2-19-76)	11	--	6	0.08	0.24	<0.1
Influent (1-27-76)	635	36,900	10	--	--	--
Influent (2-19-76)	115	10,200	6	0.05	0.18	<0.1
Influent (2-20-76)	35	5,600	4	0.20	0.41	0.53
Influent (2-21-76)	1135	33,900	21	0.53	1.72	13.71
Effluent (1-27-76)††	11	--	6	--	--	--
Effluent (2-19-76)	8	--	4	0.01	0.19	<0.1
Effluent (2-20-76)	10	--	8	0.04	0.20	<0.1
Effluent (2-21-76)	--	--	7	0.50	0.58	1.33

	Organic-N§			Ammonium-N			$NO_3 + NO_2$ -N
	Total mg/ℓ	Solids mg/kg‡	<0.45 µm mg/ℓ	Total mg/ℓ	Exchangeable§§ mg/kg‡ (mg/ℓ)#	<0.45 µm mg/ℓ	<0.45 µm mg/ℓ
Background water (1-27-76)	0.3	--	<0.25	1.23	--	<0.05	<0.01
Background water (2-19-76)	<0.25	--	<0.25	1.54	--	<0.05	<0.01
Influent (1-27-76)	62.1	3605	1.0	--	279 (4.7)	14.1	1.20
Influent (2-19-76)	7.7	650	0.9	--	34.8 (0.4)	4.46	<0.01
Influent (2-20-76)	3.6	532	0.55	--	7.1 (0.1)	1.23	<0.01
Influent (2-21-76)	78.0	2265	3.8	--	259 (8.5)	16.2	<0.01
Effluent (1-27-76)††	1.1	--	1.1	2.94	--	2.40	0.40
Effluent (2-19-76)	1.9	--	0.9	2.94	--	3.08	<0.01
Effluent (2-20-76)	0.15	--	0.6	2.67	--	1.64	0.04
Effluent (2-21-76)	2.6	--	2.2	13.8	419 (0.5)	12.0	<0.01

(Continued)

Note: dashes (--) indicate missing data.

* Influents: determined for <2-µm fraction from mechanical particle-size analysis, sodium hexametaphosphate added for dispersion; effluents: determined for total effluent, no dispersive agent added; dispersed by sonication.

** Gravel and larger sized rocks excluded.

† Solids formed an upper stratum of dispersed colloids.

†† Collected at second crossdike (3 equal compartment disposal area).

‡ Dry weight.

‡‡ op' DDD, pp' DDD, op' DDT, pp' DDT, dieldrin, aldrin, lindane, heptachlor, heptachlor epoxide, and chlordane are below detection limits.

§ TKN - (<0.45 µm) ammonium-N.

§§ Sodium acetate extractable.

\# Values in parentheses (mg/ℓ) exchangeable ammonium-N in total sample.

Table B14 (Continued)

Sample Identification	Orthophosphate-P Total mg/ℓ	Orthophosphate-P <0.45 μm mg/ℓ	Total-P Total mg/ℓ	Total-P Solids mg/kg‡	Total-P <0.45 μm mg/ℓ	Alkalinity as $CaCO_3$ mg/ℓ	Chloride mg/ℓ	Sulfide Total mg/ℓ	Sulfide Solids mg/kg‡
Background water (1-27-76)	0.45	0.16	0.48	--	0.21	39.67	<5	1.9	--
Background water (2-19-76)	0.10	0.05	0.17	--	0.09	41.86	5	1.9	--
Influent (1-27-76)	--	0.06	64.7	3800	0.08	129.0	10	2.8	73.8
Influent (2-19-76)	--	0.04	14.9	1400	0.07	59.91	<5	2.5	48.8
Influent (2-20-76)	--	0.07	12.8	2200	0.08	51.38	10	1.8	36.3
Influent (2-21-76)	--	0.08	105.7	3200	0.08	88.90	10	4.8	91.6
Effluent (1-27-76)††	0.65	0.07	0.82	--	0.09	86.30	40	1.6	--
Effluent (2-19-76)	0.89	0.06	0.93	--	0.50	29.75	25	1.8	--
Effluent (2-20-76)	0.21	0.10	0.55	--	0.11	55.15	10	1.7	--
Effluent (2-21-76)	2.04	0.11	5.51	--	0.11	70.80	5	--	--

Sample Identification	Cation Exchange Capacity meq/100 g‡	C:N:P Ratio	Calcium Total mg/ℓ	Calcium Solids mg/kg‡	Calcium <0.45 μm mg/ℓ	Magnesium Total mg/ℓ	Magnesium Solids mg/kg‡	Magnesium <0.45 μm mg/ℓ
Background water (1-27-76)	--	12.5:0.6:1	14.7	--	14.0	3.5	--	2.7
Background water (2-19-76)	--	64.7:<0.5:1	15.9	--	10.0	3.5	--	3.0
Influent (1-27-76)	50.0	9.8:1.2:1	85.1	4140	15.0	90.8	5070	4.7
Influent (2-19-76)	10.2	7.7:0.8:1	68.3	5315	12.0	40.2	3520	3.0
Influent (2-20-76)	8.1	2.7:0.4:1	45.7	5465	14.0	26.5	4080	2.8
Influent (2-21-76)	36.6	10.7:0.9:1	70.2	1890	8.0	208	6220	2.6
Effluent (1-27-76)††	--	13.4:4.3:1	29.4	--	29.0	10.4	--	8.0
Effluent (2-19-76)	--	8.6:5.4:1	57.9	--	14.0	3.15	--	3.1
Effluent (2-20-76)	--	18.2:3.3:1	115	--	13.0	5.7	--	3.5
Effluent (2-21-76)	65.9	-- :2.6:1	16.8	4850	13.7	11.3	7080	2.6

Sample Identification	Potassium Total mg/ℓ	Potassium Solids mg/kg‡	Potassium <0.45 μm mg/ℓ	Sodium Total mg/ℓ	Sodium Solids mg/kg‡	Sodium <0.45 μm mg/ℓ	Iron Total mg/ℓ	Iron Solids mg/kg‡	Iron <0.45 μm mg/ℓ
Background water (1-27-76)	2.8	--	1.3	6.5	--	6.3	1.1	--	0.153
Background water (2-19-76)	3.0	--	1.2	7.3	--	6.1	1.3	--	0.531
Influent (1-27-76)	316	18,300	3.9	--	--	6.3	946	55,500	2.58
Influent (2-19-76)	185	17,200	2.4	126	10,950	10.0	260	24,500	0.227
Influent (2-20-76)	128	21,800	1.6	--	--	6.3	160	27,600	0.113
Influent (2-21-76)	593	17,900	2.6	85.0	2,395	6.2	1779	53,800	2.09
Effluent (1-27-76)††	8.8	--	5.6	8.4	--	5.3	15.5	--	0.096
Effluent (2-19-76)	9.5	--	2.1	14.4	--	6.3	17.1	--	0.196
Effluent (2-20-76)	4.6	--	1.5	9.2	--	6.4	12.8	--	0.110
Effluent (2-21-76)	25.8	18,200	2.4	6.5	240	6.0	97.2	24,100	0.563

Sample Identification	Manganese Total mg/ℓ	Manganese Solids mg/kg‡	Manganese <0.45 μm mg/ℓ	Zinc Total mg/ℓ	Zinc Solids mg/kg‡	Zinc <0.45 μm mg/ℓ	Cadmium Total mg/ℓ	Cadmium Solids mg/kg‡	Cadmium <0.45 μm mg/ℓ
Background water (1-27-76)	0.04	--	0.002	0.030	--	0.005	--	--	0.0013
Background water (2-19-76)	0.06	--	0.125	0.074	--	0.009	--	--	0.0039
Influent (1-27-76)	29.4	1650	1.43	7.1	415	0.004	--	--	0.0010
Influent (2-19-76)	5.90	519	0.432	1.3	123	0.005	0.0031	--	0.0045
Influent (2-20-76)	4.33	683	0.374	0.60	104	0.005	--	--	0.0021
Influent (2-21-76)	38.4	1155	0.233	11.7	354	0.007	0.0260	0.779	0.0003
Effluent (1-27-76)††	1.18	--	0.569	0.243	--	0.040	--	--	0.0017
Effluent (2-19-76)	0.80	--	0.222	0.244	--	0.012	0.0034	--	0.0013
Effluent (2-20-76)	0.53	--	0.220	0.112	--	0.007	--	--	0.0019
Effluent (2-21-76)	1.69	1195	0.119	0.55	210	0.009	0.0068	4.865	0.0005

(Continued)

Table B14 (Concluded)

Sample Identification	Copper Total mg/ℓ	Copper Solids mg/kg‡	Copper <0.45 μm mg/ℓ	Nickel Total mg/ℓ	Nickel Solids mg/kg‡	Nickel <0.45 μm mg/ℓ
Background water (1-27-76)	--	--	0.001	0.010	--	0.006
Background water (2-19-76)	0.07	--	0.008	0.026	--	0.004
Influent (1-27-76)	1.18	69.0	0.002	1.18	66.4	0.004
Influent (2-19-76)	0.24	22.4	0.005	0.42	29.1	0.004
Influent (2-20-76)	0.10	16.0	0.003	0.21	24.2	<0.003
Influent (2-21-76)	2.19	68.5	0.005	2.17	64.6	<0.003
Effluent (1-27-76)††	0.06	--	0.006	0.020	--	0.003
Effluent (2-19-76)	0.07	--	0.005	0.024	--	0.004
Effluent (2-20-76)	0.05	--	0.005	0.017	--	0.003
Effluent (2-21-76)	0.17	131	0.007	0.123	42.0	<0.003

	Lead Total mg/ℓ	Lead Solids mg/kg‡	Lead <0.45 μm mg/ℓ	Mercury Total mg/ℓ	Mercury Solids mg/kg‡	Mercury <0.45 μm mg/ℓ
Background water (1-27-76)	0.004	--	<0.001	<0.0002	--	<0.0002##
Background water (2-19-76)	0.002	--	<0.001	0.0002	--	<0.0002##
Influent (1-27-76)	2.10	123.0	0.003	0.0010	0.25	<0.0002##
Influent (2-19-76)	0.48	45.0	<0.001	0.0007	0.07	<0.0002##
Influent (2-20-76)	0.24	40.4	<0.001	0.0012	0.21	<0.0002##
Influent (2-21-76)	3.52	106.5	0.002	0.0067	0.20	<0.0002##
Effluent (1-27-76)††	0.028	--	0.001	<0.0002	--	<0.0002##
Effluent (2-19-76)	0.060	--	<0.001	0.0002	--	<0.0002##
Effluent (2-20-76)	0.020	--	<0.001	<0.0002	--	<0.0002##
Effluent (2-21-76)	0.24	142	<0.001	0.0004	0.10	<0.0002##

Cold vapor method.

(Sheet 3 of 3)

Table B15

Physical and Chemical Composition of Disposal Area No. 22 Influents and Effluents and Calcasieu River Background Water, near Lake Charles, Louisiana

Sample Identification	Particle Size (Mechanical) % <2 µm	% 2-50 µm	% >50 µm	Coulter Counter Particle Size, µm* >50%	>80%	Nonfilterable Solids, %	Settleable Solids mℓ/ℓ
Background water (2-24-76) (surface water)	--	--	--	5.4	2.82	0.0036	<0.1
Influent (2-4-76)	68	27	5	0.92	0.58	6.48	300
Influent (2-5-76)	67	30	3	1.05	0.6	5.22	300
Influent (2-6-76)	58.5	31	10.5	0.77	0.5	7.08	500
Effluent (2-4-76)**	58	30	12	0.88	0.56	3.27	300
Effluent (2-5-76)	--	--	--	5.4	3.0	0.00355	<0.1
Effluent (2-6-76)	--	--	--	3.0	1.4	0.38	60

	Total Organic Carbon Total mg/ℓ	Solids mg/kg††	<0.45 µm mg/ℓ	Chlorinated Pesticides† op'DDE mg/ℓ	pp'DDE mg/ℓ	Total PCB (1242, 1254, 1260) mg/ℓ
Background water (2-4-76)	12	--	9	0.28	1.57	0.30
Influent (2-4-76)	1090	16,700	8	<0.01	0.15	8.96
Influent (2-5-76)	--	--	7	0.22	0.50	<0.1
Influent (2-6-76)	895	12,400	18	0.17	1.36	5.99
Effluent (2-4-76)**	405	12,300	3	<0.01	<0.01	<0.1
Effluent (2-5-76)	12	--	5	<0.01	<0.01	<0.1
Effluent (2-6-76)	--	--	13	<0.01	2.87	<0.1

	Organic-N‡ Total mg/ℓ	Solids mg/kg††	<0.45 µm mg/ℓ	Ammonium-N Total mg/ℓ	Exchangeable‡‡ mg/kg†† (mg/ℓ)§	<0.45 µm mg/ℓ	$NO_3 + NO_2$ -N <0.45 µm mg/ℓ
Background water (2-4-76)	0.2	--	0.9	1.15	--	0.41	0.04
Influent (2-4-76)	104	1589	3.1	--	78.0 (5.1)	24.7	<0.01
Influent (2-5-76)	--	--	0.7	--	45.2 (2.4)	14.4	<0.01
Influent (2-6-76)	60.6	867	0.6	--	84.4 (6.0)	19.4	<0.01
Effluent (2-4-76)**	31.9	906	2.7	--	58.4 (1.9)	10.2	<0.01
Effluent (2-5-76)	1.8	--	5.2	5.85	--	7.2	<0.01
Effluent (2-6-76)	--	--	0.8	--	74.9 (0.3)	13.4	0.08

	Ortho-PO_4-P <0.45 µm mg/ℓ	Total-P Total mg/ℓ	Solids mg/kg††	<0.45 µm mg/ℓ	Alkalinity as $CaCO_3$ mg/ℓ	Chloride mg/ℓ	Sulfide Total mg/ℓ	Solids mg/kg††
Background water (2-4-76)	0.06	0.14	--	0.06	29.95	2,080	--	--
Influent (2-4-76)	0.12	90.9	1400	0.15	423.8	12,200	22.8	272
Influent (2-5-76)	0.07	--	--	0.12	232.5	19,200	24.2	329
Influent (2-6-76)	0.11	99.3	1400	0.15	306.1	10,200	32.0	349
Effluent (2-4-76)**	0.15	45.9	1400	0.15	254.1	13,700	18.4	327
Effluent (2-5-76)	0.06	0.11	--	0.06	107.8	11,700	--	--
Effluent (2-6-76)	0.05	--	--	0.11	235.7	10,200	--	--

(Continued)

Note: dashes (--) indicate missing data.

* Influents: determined for <2-µm fraction from mechanical particle-size analysis, sodium hexametaphosphate added for dispersion; effluents: determined for total effluent, no dispersive agent added, dispersed by sonication.

** Effluent was obtained from discharge pipes near influent pipe; approximately 300 yd of overland flow and little ponding.

† op'DDD, pp'DDD, op'DDT, pp'DDT, dieldrin, aldrin, lindane, heptachlor, heptachlor epoxide, and chlordane are below detection limits.

†† Dry weight.

‡ TKN - (<0.45 µm) ammonium-N.

‡‡ Sodium acetate extractable.

§ Values in parentheses (mg/ℓ) exchangeable ammonium-N in total sample.

Table B15 (Concluded)

Sample Identification	Cation Exchange Capacity meq/100 g†† Soil	Calcium			Magnesium		
		Total mg/ℓ	Solids mg/kg††	<0.45 μm mg/ℓ	Total mg/ℓ	Solids mg/kg††	<0.45 μm mg/ℓ
Background water (2-4-76)	--	55	--	49	159	--	150
Influent (2-4-76)	55.7	430	2595	280	966	8,840	420
Influent (2-5-76)	58.3	326	1150	280	1010	8,455	600
Influent (2-6-76)	56.5	410	2900	220	928	10,090	230
Effluent (2-4-76)**	66.1	392	3125	300	778	13,130	360
Effluent (2-5-76)	--	292	--	220	667	--	430
Effluent (2-6-76)	66.2	245	4270	230	749	13,700	700

	Potassium			Sodium			Iron		
	Total mg/ℓ	Solids mg/kg††	<0.45 μm mg/ℓ	Total mg/ℓ	Solids mg/kg††	<0.45 μm mg/ℓ	Total mg/ℓ	Solids mg/kg††	<0.45 μm mg/ℓ
Background water (2-4-76)	64.3	--	39	--	--	1400	1.29	--	0.200
Influent (2-4-76)	1115	14,200	210	7700	--	7300	2067	31,700	14.6
Influent (2-5-76)	802	11,500	210	6065	--	7100	1831	35,000	4.15
Influent (2-6-76)	1220	14,900	180	--	--	6500	2368	33,300	7.04
Effluent (2-4-76)**	797	17,600	230	7535	--	7500	1288	39,300	3.88
Effluent (2-5-76)	432	--	180	--	--	7200	7.19	--	0.400
Effluent (2-6-76)	235	14,800	180	4600	--	6600	152	39,900	0.402

	Manganese			Zinc			Cadmium		
	Total mg/ℓ	Solids mg/kg††	<0.45 μm mg/ℓ	Total mg/ℓ	Solids mg/kg††	<0.45 μm mg/ℓ	Total mg/ℓ	Solids mg/kg††	<0.45 μm mg/ℓ
Background water (2-4-76)	0.192	--	0.184	0.053	--	0.005	--	--	0.0011
Influent (2-4-76)	60.2	720	14.4	7.05	109	0.002	0.0173	0.122	0.0100
Influent (2-5-76)	45.5	736	7.45	5.74	110	0.0013	0.0020	--	0.0024
Influent (2-6-76)	59.8	723	9.31	6.89	97.3	0.003	--	--	0.0080
Effluent (2-4-76)**	37.5	912	7.95	4.40	135	0.002	0.0048	0.046	0.0034
Effluent (2-5-76)	5.00	--	7.75	0.262	--	0.002	--	--	0.0050
Effluent (2-6-76)	11.8	2165	3.60	0.522	137	0.002	0.0014	--	0.0027

	Copper			Nickel		
	Total mg/ℓ	Solids mg/kg††	<0.45 μm mg/ℓ	Total mg/ℓ	Solids mg/kg††	<0.45 μm mg/ℓ
Background water (2-4-76)	0.089	--	0.006	0.011	--	0.003
Influent (2-4-76)	1.91	29.4	0.003	2.11	24.4	0.003
Influent (2-5-76)	1.45	27.7	0.0025	1.99	21.8	<0.003
Influent (2-6-76)	2.02	28.5	0.002	1.84	15.4	0.014
Effluent (2-4-76)**	1.01	30.7	0.002	1.70	25.3	0.004
Effluent (2-5-76)	0.037	--	0.002	0.187	--	0.005
Effluent (2-6-76)	0.138	29.7	0.026	0.968	44.0	0.006

	Lead			Mercury		
	Total mg/ℓ	Solids mg/kg††	<0.45 μm mg/ℓ	Total mg/ℓ	Solids mg/kg††	<0.45 μm mg/ℓ
Background water (2-4-76)	0.012	--	0.005	0.0030	--	<0.0002§§
Influent (2-4-76)	1.98	30.6	<0.001	--	--	<0.0002§§
Influent (2-5-76)	1.60	30.6	<0.001	0.0047	0.09	<0.0002§§
Influent (2-6-76)	2.78	39.2	<0.001	0.0078	0.11	<0.0002§§
Effluent (2-4-76)**	1.27	38.8	<0.001	0.0036	0.11	<0.0002§§
Effluent (2-5-76)	0.008	--	<0.001	<0.0002§§	--	<0.0002§§
Effluent (2-6-76)	0.141	37.0	<0.001	0.0008	0.22	<0.0002§§

** Effluent was obtained from discharge pipes near influent pipe; approximately 300 yd of overland flow and little ponds.

†† Dry weight.

§§ Cold vapor method.

Table B16

Physical and Chemical Composition of Disposal Area Influents and Effluents and Duwamish Waterway Background Water, Seattle, Washington

Sample Identification	Total Solids %	Non-filterable Solids, %	Settleable Solids, mℓ/ℓ	Area of Dredging Activity	Water Sample Depth, Metres (River Water Versus Salt Wedge)
Predredge background water (2-27-76)	--	--	--	--	<1 >8
Background water (3-6-76)	--	--	--	--	<1 >8
Background water (3-8-76)	--	--	--	--	<1 >8
Background water (3-18-76)	--	--	--	--	<1 >8
Background water (3-22-76)	--	--	--	--	<1 >8
Background water (3-23-76)	--	--	--	--	<1 >8
Postdredge background water (4-20-76)	--	--	--	--	<1 >8
Influent (3-16-76)	10.5	--	300	5, 6	--
Influent (3-19-76)	3.2	--	300	3	--
Influent (1)(3-22-76)	3.8	--	220	3	--
Influent (2)(3-22-76)	12.4	--	800	1, 2	--
Influent (3-23-76)	3.5	--	140	1	--
Effluent (Pond 1)(4-3-76)	0.01224	0.01	<0.01	Postdredge	--
Effluent (Pond 1)(4-4-76)	0.01104	--	0.6	Postdredge	--
Effluent (Pond 1)(4-6-76)	0.03424	0.03	1.2	Postdredge	--

	Total Organic Carbon		Oil and Grease			Total PCB's (1242, 1254, 1260) mg/ℓ	PCB's (Centrifuged Solids) mg/kg*
	Total mg/ℓ	<0.45 µm mg/ℓ	Total mg/ℓ	Solids mg/kg*	<0.45 µm mg/ℓ		
Predredge background water (2-27-76)	5 3	--	0.35 0.15	--	--	0.000021 0.000014	--
Background water (3-6-76)	3 3	--	0.2 0.1	--	--	0.000022 0.000014	--
Background water (3-8-76)	3 2	--	<0.3 0.1	--	--	0.000011 0.000024	--
Background water (3-18-76)	5 4	--	0.1 <0.1	--	--	0.000021 0.000007	--
Background water (3-22-76)	4 3	--	0.1 0.1	--	--	0.000014 0.000013	--
Background water (3-23-76)	5 3	--	0.1 <0.1	--	--	0.000016 0.000010	--
Postdredge background water (4-20-76)	4 3	--	0.2 <0.1	--	--	0.000009 0.000007	--
Influent (3-16-76)	--	11	795	6319	41.5	1.47	13.7
Influent (3-19-76)	--	19	183	8492	48	0.51	15.9
Influent (1)(3-22-76)	--	6	147	5255	2.8	3.75	98.5
Influent (2)(3-22-76)	--	14	1497	7305	12	11.2	89.8
Influent (3-23-76)	--	6	288	2928	2.0	9.2	263.2
Effluent (Pond 1)(4-3-76)	--	6	3	--	6	0.0012	6.0
Effluent (Pond 1)(4-4-76)	--	6	7	--	5	0.006	51
Effluent (Pond 1)(4-6-76)	--	12	256	--	4	0.016	41

(Continued)

Note: dashes (--) indicate missing data.
* Dry weight.

Table B16 (Continued)

Sample Identification	Soluble PCB's (Centrifuged) mg/ℓ	Organic-N Total mg/ℓ	Organic-N Solids mg/kg*	Organic-N <0.45 µm mg/ℓ	Ammonium-N Total mg/ℓ	Ammonium-N Exchangeable mg/kg*	Ammonium-N <0.45 µm mg/ℓ
Predredge background water (2-27-76)	--	0.11 0.46	-- --	-- --	0.38 0.04	-- --	-- --
Background water (3-6-76)	--	0.01 <0.01	-- --	-- --	0.48 0.04	-- --	-- --
Background water (3-8-76)	--	0.06 0.09	-- --	-- --	0.46 0.05	-- --	-- --
Background water (3-18-76)	--	0.03 0.05	-- --	-- --	0.42 0.27	-- --	-- --
Background water (3-22-76)	--	0.06 0.04	-- --	-- --	0.37 0.03	-- --	-- --
Background water (3-23-76)	--	0.06 0.02	-- --	-- --	0.34 0.04	-- --	-- --
Postdredge background water (4-20-76)	--	0.10 0.06	-- --	-- --	0.38 0.04	-- --	-- --
Influent (3-16-76)	0.037	95	905	0.4	--	--	7.8
Influent (3-19-76)	0.0041	81	2525	<0.1	--	--	16.0
Influent (1)(3-22-76)	0.0106	25.2	625	1.4	--	--	3.4
Influent (2)(3-22-76)	0.054	112	803	13.0	--	--	14.0
Influent (3-23-76)	0.013	25.5	722	0.2	--	10	3.6
Effluent (Pond 1)(4-3-76)	0.00048	--	--	4.2	--	--	4.1
Effluent (Pond 1)(4-4-76)	0.00039	--	--	4.1	--	--	4.2
Effluent (Pond 1)(4-6-76)	0.0019	--	--	7.1	--	--	7.1

Sample Identification	$NO_3 + NO_2$ -N <0.45 µm mg/ℓ	Orthophosphate-P Total mg/ℓ	Orthophosphate-P <0.45 µm mg/ℓ	Total-P Total mg/ℓ	Total-P Solids mg/kg*	Total-P <0.45 µm mg/ℓ
Predredge background water (2-27-76)	0.51 0.42	0.08 0.08	-- --	0.15 0.08	-- --	-- --
Background water (3-6-76)	0.52 0.37	0.04 0.04	-- --	0.11 0.09	-- --	-- --
Background water (3-8-76)	0.53 0.41	0.09 0.07	-- --	0.17 0.09	-- --	-- --
Background water (3-18-76)	0.46 0.43	0.08 0.08	-- --	0.15 0.13	-- --	-- --
Background water (3-22-76)	0.40 0.40	0.08 0.06	-- --	0.17 0.10	-- --	-- --
Background water (3-23-76)	0.41 0.40	0.08 0.06	-- --	0.15 0.09	-- --	-- --
Postdredge background water (4-20-76)	0.34 0.34	0.09 0.06	-- --	0.16 0.09	-- --	-- --
Influent (3-16-76)	0.37	--	0.39	160	1520	0.43
Influent (3-19-76)	0.33	--	0.40	52	1635	0.49
Influent (1)(3-22-76)	0.34	--	0.45	52	1365	0.44
Influent (2)(3-22-76)	0.13	--	0.31	162	1280	0.31
Influent (3-23-76)	0.17	--	0.31	45.5	1290	0.34
Effluent (Pond 1)(4-3-76)	0.38	--	0.30	0.35	--	0.30
Effluent (Pond 1)(4-4-76)	0.36	--	0.30	0.39	--	0.31
Effluent (Pond 1)(4-6-76)	0.35	--	0.27	1.1	--	0.28

(Continued)

Table B16 (Continued)

Sample Identification	Alkalinity as $CaCO_3$ mg/ℓ	Chloride mg/ℓ	Sulfate mg/ℓ	Sulfide			Cation Exchange Capacity meq/100 g*Soil
				Total mg/ℓ	Solids mg/kg*	<0.45 µm mg/ℓ	
Predredge background water (2-27-76)	--	--	--	<0.02	--	--	--
	--	--	--	<0.02	--	--	--
Background water (3-6-76)	--	--	--	<0.02	--	--	--
	--	--	--	<0.02	--	--	--
Background water (3-8-76)	--	--	--	<0.02	--	--	--
	--	--	--	<0.02	--	--	--
Background water (3-18-76)	--	--	--	<0.02	--	--	--
	--	--	--	<0.02	--	--	--
Background water (3-22-76)	--	--	--	--	--	--	--
	--	--	--	--	--	--	--
Background water (3-23-76)	--	--	--	--	--	--	--
	--	--	--	--	--	--	--
Postdredge background water (4-20-76)	--	--	--	--	--	--	--
	--	--	--	--	--	--	--
Influent (3-16-76)	367	15,800	2000	71	676	<0.02	--
Influent (3-19-76)	552	16,000	1800	99	3091	0.08	--
Influent (1)(3-22-76)	197	16,200	2100	27	711	<0.02	--
Influent (2)(3-22-76)	466	16,300	1950	45	363	0.02	--
Influent (3-23-76)	158	16,200	1930	28	800	<0.02	70
Effluent (Pond 1)(4-3-76)	177	15,700	2130	--	--	--	--
Effluent (Pond 1)(4-4-76)	179	15,700	2150	--	--	<0.02	--
Effluent (Pond 1)(4-6-76)	193	15,500	1900	--	--	--	--

	C:N:P Ratio	Calcium Solids mg/kg*	Magnesium Solids mg/kg*	Potassium Solids mg/kg*	Sodium Solids mg/kg*
Predredge background water (2-27-76)	--	--	--	--	--
Background water (3-6-76)	--	--	--	--	--
Background water (3-8-76)	--	--	--	--	--
Background water (3-18-76)	--	--	--	--	--
Background water (3-22-76)	--	--	--	--	--
Background water (3-23-76)	--	--	--	--	--
Postdredge background water (4-20-76)	--	--	--	--	--
Influent (3-16-76)	--	26,200	27,500	3400	13,100
Influent (3-19-76)	--	29,100	37,800	3100	12,800
Influent (1)(3-22-76)	--	14,700	16,400	3400	10,000
Influent (2)(3-22-76)	--	10,200	12,300	--	11,400
Influent (3-23-76)	--	10,700	10,700	--	10,200
Effluent (Pond 1)(4-3-76)	--	--	--	--	--
Effluent (Pond 1)(4-4-76)	--	--	--	--	--
Effluent (Pond 1)(4-6-76)	--	--	--	--	--

(Continued)

Table B16 (Continued)

Sample Identification	Iron			Manganese		
	Total mg/ℓ	Solids mg/kg*	<0.45 µm mg/ℓ	Total mg/ℓ	Solids mg/kg*	<0.45 µm mg/ℓ
Predredge background water (2-27-76)	0.67 0.30	-- --	-- --	0.047 0.048	-- --	-- --
Background water (3-6-76)	0.52 0.48	-- --	-- --	0.072 0.072	-- --	-- --
Background water (3-8-76)	0.46 0.42	-- --	-- --	0.084 0.062	-- --	-- --
Background water (3-18-76)	0.45 0.38	-- --	-- --	0.062 0.060	-- --	-- --
Background water (3-22-76)	0.44 0.32	-- --	-- --	0.062 0.064	-- --	-- --
Background water (3-23-76)	0.54 0.40	-- --	-- --	0.054 0.054	-- --	-- --
Postdredge background water (4-20-76)	0.40 0.36	-- --	-- --	0.052 0.036	-- --	-- --
Influent (3-16-76)	4945	47,090	0.25	54.0	513	0.100
Influent (3-19-76)	1600	50,000	0.24	8.1	250	0.078
Influent (1)(3-22-76)	1875	49,340	0.25	19.9	518	0.260
Influent (2)(3-22-76)	4845	39,100	0.27	50.4	405	0.208
Influent (3-23-76)	1640	46,840	0.36	15.9	447	0.340
Effluent (Pond 1)(4-3-76)	0.46	--	0.20	0.166	--	0.162
Effluent (Pond 1)(4-4-76)	0.54	--	0.20	0.184	--	0.176
Effluent (Pond 1)(4-6-76)	4.90	--	0.175	0.660	--	0.430

	Zinc			Cadmium		
	Total mg/ℓ	Solids mg/kg*	<0.45 µm mg/ℓ	Total mg/ℓ	Solids mg/kg*	<0.45 µm mg/ℓ
Predredge background water (2-27-76)	0.015 0.002	-- --	-- --	<0.002 0.002	-- --	-- --
Background water (3-6-76)	0.010 <0.002	-- --	-- --	<0.002 <0.002	-- --	-- --
Background water (3-8-76)	0.010 <0.002	-- --	-- --	<0.002 <0.002	-- --	-- --
Background water (3-18-76)	0.010 0.004	-- --	-- --	<0.002 <0.002	-- --	-- --
Background water (3-22-76)	0.022 0.006	-- --	-- --	<0.002 <0.002	-- --	-- --
Background water (3-23-76)	0.026 0.006	-- --	-- --	<0.002 <0.002	-- --	-- --
Postdredge background water (4-20-76)	0.016 0.006	-- --	-- --	<0.002 <0.002	-- --	-- --
Influent (3-16-76)	206	1958	0.006	0.91	8.7	<0.002
Influent (3-19-76)	31.7	992	0.006	0.23	7.2	<0.002
Influent (1)(3-22-76)	26.2	690	0.008	0.17	4.35	<0.002
Influent (2)(3-22-76)	59.8	482	<0.002	0.57	4.58	<0.002
Influent (3-23-76)	19.6	560	0.016	0.15	4.21	<0.002
Effluent (Pond 1)(4-3-76)	0.016	--	0.014	<0.002	--	<0.002
Effluent (Pond 1)(4-4-76)	0.024	--	0.016	<0.002	--	<0.002
Effluent (Pond 1)(4-6-76)	0.273	--	0.048	0.005	--	<0.002

(Continued)

Table B16 (Continued)

	Copper			Nickel		
Sample Identification	Total mg/ℓ	Solids mg/kg*	<0.45 µm mg/ℓ	Total mg/ℓ	Solids mg/kg*	<0.45 µm mg/ℓ
Predredge background water (2-27-76)	0.005	--	--	<0.01	--	--
	0.004	--	--	<0.01	--	--
Background water (3-6-76)	0.002	--	--	<0.01	--	--
	0.005	--	--	<0.01	--	--
Background water (3-8-76)	0.003	--	--	<0.01	--	--
	0.006	--	--	<0.01	--	--
Background water (3-18-76)	0.016	--	--	<0.01	--	--
	0.040	--	--	<0.01	--	--
Background water (3-22-76)	0.012	--	--	<0.01	--	--
	0.044	--	--	<0.01	--	--
Background water (3-23-76)	0.012	--	--	<0.01	--	--
	0.036	--	--	<0.01	--	--
Postdredge background water (4-20-76)	0.014	--	--	0.03	--	--
	0.060	--	--	--	--	--
Influent (3-16-76)	17.4	165	0.072	7.8	74.1	0.02
Influent (3-19-76)	4.9	151	0.048	3.2	101	<0.01
Influent (1)(3-22-76)	4.5	117	0.046	2.1	54.8	0.03
Influent (2)(3-22-76)	13.8	111	0.044	4.8	38.7	0.02
Influent (3-23-76)	4.6	130	0.052	1.4	40.3	0.02
Effluent (Pond 1)(4-3-76)	0.056	--	0.052	--	--	--
Effluent (Pond 1)(4-4-76)	0.054	--	0.060	--	--	--
Effluent (Pond 1)(4-6-76)	0.120	--	0.058	--	--	--

	Lead			Mercury		
	Total mg/ℓ	Solids mg/kg*	<0.45 µm mg/ℓ	Total mg/ℓ	Solids mg/kg*	<0.45 µm mg/ℓ
Predredge background water (2-27-76)	--	--	--	0.0001	--	--
	--	--	--	0.0002	--	--
Background water (3-6-76)	--	--	--	0.0001	--	--
	--	--	--	0.0003	--	--
Background water (3-8-76)	--	--	--	0.0002	--	--
	--	--	--	0.0002	--	--
Background water (3-18-76)	--	--	--	0.0002	--	--
	--	--	--	0.0002	--	--
Background water (3-22-76)	--	--	--	0.0002	--	--
	--	--	--	0.0002	--	--
Background water (3-23-76)	--	--	--	<0.0002	--	--
	--	--	--	<0.0002	--	--
Postdredge background water (4-20-76)	--	--	--	0.0006	--	--
	--	--	--	0.0010	--	--
Influent (3-16-76)	34.6	327	--	0.040	0.38	0.0002
Influent (3-19-76)	9.9	299	--	0.033	1.03	<0.0002
Influent (1)(3-22-76)	3.9	94.2	--	0.022	0.57	<0.0002
Influent (2)(3-22-76)	12.1	94.8	--	0.087	0.70	<0.0002
Influent (3-23-76)	7.0	191	--	0.018	0.53	<0.0002
Effluent (Pond 1)(4-3-76)	--	--	--	0.0002	--	0.0002
Effluent (Pond 1)(4-4-76)	--	--	--	0.0002	--	0.0002
Effluent (Pond 1)(4-6-76)	--	--	--	0.0011	--	0.0003

(Continued)

Table B 16 (Continued)

Sample Identification	Chromium			Arsenic		
	Total mg/ℓ	Solids mg/kg*	<0.45 μm mg/ℓ	Total mg/ℓ	Solids mg/kg*	<0.45 μm mg/ℓ
Predredge background water (2-27-76)	0.007 0.045	-- --	-- --	0.001 0.011	-- --	-- --
Background water (3-6-76)	0.007 0.037	-- --	-- --	0.001 0.001	-- --	-- --
Background water (3-8-76)	0.003 0.033	-- --	-- --	0.002 0.001	-- --	-- --
Background water (3-18-76)	0.010 0.008	-- --	-- --	0.002 0.002	-- --	-- --
Background water (3-22-76)	0.008 0.036	-- --	-- --	0.002 0.002	-- --	-- --
Background water (3-23-76)	0.008 0.034	-- --	-- --	0.003 0.003	-- --	-- --
Postdredge background water (4-20-76)	0.006 0.028	-- --	-- --	<0.001 0.002	-- --	-- --
Influent (3-16-76)	--	--	--	2.25	20.8	0.084
Influent (3-19-76)	--	--	--	0.70	18.5	0.117
Influent (1)(3-22-76)	--	--	--	0.73	18.9	0.019
Influent (2)(3-22-76)	--	--	--	1.82	14.0	0.088
Influent (3-23-76)	--	--	--	0.50	13.9	0.014
Effluent (Pond 1)(4-3-76)	0.028	--	0.026	0.016	--	0.016
Effluent (Pond 1)(4-4-76)	0.024	--	0.024	0.008	--	0.014
Effluent (Pond 1)(4-6-76)	0.056	--	0.025	0.006	--	0.006

	Chemical Oxygen Demand		Volatile Solids	
	Total mg/ℓ	Solids mg/kg*	Total %	Solids %*
Predredge background water (2-27-76)	--	--	--	--
Background water (3-6-76)	--	--	--	--
Background water (3-8-76)	--	--	--	--
Background water (3-18-76)	--	--	--	--
Background water (3-22-76)	--	--	--	--
Background water (3-23-76)	--	--	--	--
Postdredge background water (4-20-76)	--	--	--	--
Influent (3-16-76)	11,000	104,600	--	--
Influent (3-19-76)	3,900	122,100	--	--
Influent (1)(3-22-76)	3,500	91,500	--	--
Influent (2)(3-22-76)	12,200	98,500	--	--
Influent (3-23-76)	3,200	91,700	--	--
Effluent (Pond 1)(4-3-76)	--	--	--	--
Effluent (Pond 1)(4-4-76)	--	--	--	--
Effluent (Pond 1)(4-6-76)	--	--	--	--

(Continued)

Table B16 (Continued)

Sample Identification	Total Solids %	Non-filterable Solids, %	Settleable Solids mℓ/ℓ	Area of Dredging Activity	Total Organic Carbon	
					Total mg/ℓ	<0.45 µm mg/ℓ
Effluent (Pond 2)(3-16-76)	0.00635	<0.01	0.4	5, 6	--	16
Effluent (Pond 2)(3-19-76)	0.01000	<0.01	0.2	3	--	14
Effluent (1)(Pond 2)(3-22-76)	0.00640	<0.01	<0.1	3	--	12
Effluent (2)(Pond 2)(3-22-76)	0.00515	<0.01	<0.1	1, 2	--	11
Effluent (Pond 2)(3-23-76)	0.01050	0.01	<0.1	1	--	11
Effluent (Pond 2)(4-1-76)	0.01170	0.01	1.0	Postdredge	--	9
Effluent (Pond 2)(4-3-76)	0.00656	--	--	Postdredge	--	7
Effluent (Pond 2)(4-4-76)	0.01508	--	--	Postdredge	--	7
Effluent (Pond 2)(4-5-76)	--	--	--	Postdredge	--	6
Effluent (Pond 2)(4-6-76)	0.00536	--	--	Postdredge	--	9
Slip 1 sediment (Area 6)	46.5	--	--	6	--	46
Slip 1 sediment (Area 5)	47.7	--	--	5	--	54
Slip 1 sediment (Areas 4 and 3)	40.7	--	--	3, 4	--	64
Slip 1 sediment (Area 2)	44.1	--	--	2	--	79
Slip 1 sediment (Area 1-PCB spill site)	42.5	--	--	1	--	46
Slip 1 dredge site water (2-25-76) (used for elutriate test)	--	--	--	1, 2, 3, 4, 5, 6	3	--

	Oil and Grease			Total PCB's (1242, 1254, 1260) mg/ℓ	Solids PCB's (Sediments) mg/kg*	Soluble PCB's mg/ℓ
	Total mg/ℓ	Solids mg/kg*	<0.45 µm mg/ℓ			
Effluent (Pond 2)(3-16-76)	5.4	--	4.1	<0.00008	--	<0.00008
Effluent (Pond 2)(3-19-76)	4.4	--	3.6	0.0011	--	0.00025
Effluent (1)(Pond 2)(3-22-76)	3.9	--	3.5	<0.00005	--	<0.00005
Effluent (2)(Pond 2)(3-22-76)	3.6	--	4.0	<0.0001	--	<0.00008
Effluent (Pond 2)(3-23-76)	2.6	--	3.2	<0.0006	--	<0.0012
Effluent (Pond 2)(4-1-76)	--	--	--	0.0028	--	0.00019
Effluent (Pond 2)(4-3-76)	--	--	--	0.00052	--	0.00029
Effluent (Pond 2)(4-4-76)	--	--	--	0.00045	--	0.00022
Effluent (Pond 2)(4-5-76)	--	--	--	--	--	--
Effluent (Pond 2)(4-6-76)	122	--	13	0.00080	--	0.00047
Slip 1 sediment (Area 6)	361	776	--	1	2	0.051
Slip 1 sediment (Area 5)	622	1306	--	<1	<2	0.085
Slip 1 sediment (Areas 4 and 3)	1120	2755	--	2	5	0.147
Slip 1 sediment (Area 2)	737	1673	--	8	18	0.143
Slip 1 sediment (Area 1-PCB spill site)	715	1680	--	71	167	1.70
Slip 1 dredge site water (2-25-76) (used for elutriate test)	<1	--	--	<0.0001	--	--

(Continued)

Table B16 (Continued)

Sample Identification	Organic-N Total mg/ℓ	Organic-N Solids mg/kg*	Organic-N <0.45 μm mg/ℓ	Ammonium-N Total mg/ℓ	Ammonium-N Exchangeable mg/kg*	Ammonium-N <0.45 μm mg/ℓ	$NO_3 + NO_2$ -N <0.45 μm mg/ℓ
Effluent (Pond 2)(3-16-76)	--	--	0.3	--	--	7.2	0.38
Effluent (Pond 2)(3-19-76)	--	--	0.4	--	--	7.4	0.36
Effluent (1)(Pond 2)(3-22-76)	--	--	0.6	--	--	7.6	0.36
Effluent (2)(Pond 2)(3-22-76)	--	--	0.5	--	--	7.7	0.36
Effluent (Pond 2)(3-23-76)	0.5	--	0.3	--	122	7.7	0.37
Effluent (Pond 2)(4-1-76)	--	--	<0.1	--	--	6.8	0.46
Effluent (Pond 2)(4-3-76)	--	--	1.1	--	--	5.4	0.31
Effluent (Pond 2)(4-4-76)	--	--	0.1	--	--	5.1	0.33
Effluent (Pond 2)(4-5-76)	<0.1	--	0.5	--	--	5.3	0.32
Effluent (Pond 2)(4-6-76)	0.1	--	0.1	--	--	5.4	0.28
Slip 1 sediment (Area 6)	411	884	3.8	--	--	8.2	0.85
Slip 1 sediment (Area 5)	557	1170	6.5	--	--	5.5	0.73
Slip 1 sediment (Areas 4 and 3)	445	1095	9.8	--	--	6.2	0.52
Slip 1 sediment (Area 2)	673	1528	6.0	--	--	11.0	0.38
Slip 1 sediment (Area 1-PCB spill site)	616	1448	3.0	--	47	9.0	0.39
Slip 1 dredge site water (2-25-76) (used for elutriate test)	0.13	--	--	0.04	--	--	0.41

Sample Identification	Orthophosphate-P Total mg/ℓ	Orthophosphate-P <0.45 μm mg/ℓ	Total-P Total mg/ℓ	Total-P Solids mg/kg*	Total-P <0.45 μm mg/ℓ	Alkalinity as $CaCO_3$ mg/ℓ	Chloride mg/ℓ	Sulfate mg/ℓ
Effluent (Pond 2)(3-16-76)	--	<0.01	0.19	--	0.01	206	8,800	1200
Effluent (Pond 2)(3-19-76)	--	0.02	0.15	--	0.03	209	10,600	1500
Effluent (1)(Pond 2)(3-22-76)	--	0.06	0.17	--	0.06	220	11,800	1500
Effluent (2)(Pond 2)(3-22-76)	--	0.10	0.21	--	0.11	237	12,400	1700
Effluent (Pond 2)(3-23-76)	--	0.15	0.25	--	0.15	249	13,100	1650
Effluent (Pond 2)(4-1-76)	--	0.03	0.21	--	0.04	188	--	1930
Effluent (Pond 2)(4-3-76)	--	<0.01	0.43	--	0.01	154	12,700	1680
Effluent (Pond 2)(4-4-76)	--	0.02	0.41	--	0.03	172	14,300	1830
Effluent (Pond 2)(4-5-76)	--	0.03	0.26	--	0.04	175	14,400	2000
Effluent (Pond 2)(4-6-76)	--	0.05	0.21	--	0.06	184	14,600	1850
Slip 1 sediment (Area 6)	--	--	510	1095	0.20	--	--	--
Slip 1 sediment (Area 5)	--	--	540	1135	0.26	--	--	--
Slip 1 sediment (Areas 4 and 3)	--	--	520	1280	1.36	--	--	--
Slip 1 sediment (Area 2)	--	--	530	1205	1.76	--	--	--
Slip 1 sediment (Area 1-PCB spill site)	--	--	590	1390	0.18	--	--	--
Slip 1 dredge site water (2-25-76) (used for elutriate test)	--	--	0.10	--	--	--	--	--

(Continued)

Table B16 (Continued)

Sample Identification	Sulfide			Cation Exchange Capacity	C:N:P	Calcium Solids	Magnesium Solids
	Total mg/ℓ	Solids mg/kg*	<0.45 µm mg/ℓ	meq/100 g Soil	Ratio	mg/kg	mg/kg
Effluent (Pond 2)(3-16-76)	<0.02	--	<0.02	--	--	--	--
Effluent (Pond 2)(3-19-76)	<0.02	--	<0.02	--	--	--	--
Effluent (1)(Pond 2)(3-22-76)	0.02	--	<0.02	--	--	--	--
Effluent (2)(Pond 2)(3-22-76)	<0.02	--	<0.02	--	--	--	--
Effluent (Pond 2)(3-23-76)	<0.02	--	<0.02	88.0	--	--	--
Effluent (Pond 2)(4-1-76)	--	--	--	--	--	--	--
Effluent (Pond 2)(4-3-76)	--	--	<0.02	--	--	--	--
Effluent (Pond 2)(4-4-76)	--	--	<0.02	--	--	--	--
Effluent (Pond 2)(4-5-76)	--	--	<0.02	--	--	--	--
Effluent (Pond 2)(4-6-76)	--	--	--	--	--	--	--
Slip 1 sediment (Area 6)	53	114	--	--	--	--	--
Slip 1 sediment (Area 5)	99	208	--	--	--	--	--
Slip 1 sediment (Areas 4 and 3)	86	212	--	--	--	--	--
Slip 1 sediment (Area 2)	42	95	--	--	--	--	--
Slip 1 sediment (Area 1-PCB spill site)	42	99	--	70.9	--	--	--
Slip 1 dredge site water (2-25-76) (used for elutriate test)	--	--	--	--	--	--	--

Sample Identification	Potassium Solids	Sodium Solids	Iron			Manganese		
	mg/kg	mg/kg	Total mg/ℓ	Solids mg/kg*	<0.45 µm mg/ℓ	Total mg/ℓ	Solids mg/kg*	<0.45 µm mg/ℓ
Effluent (Pond 2)(3-16-76)	--	--	4.80	--	0.74	1.52	--	1.40
Effluent (Pond 2)(3-19-76)	--	--	1.80	--	0.20	1.32	--	1.28
Effluent (1)(Pond 2)(3-22-76)	--	--	1.56	--	0.14	1.12	--	1.06
Effluent (2)(Pond 2)(3-22-76)	--	--	1.30	--	0.18	0.90	--	0.84
Effluent (Pond 2)(3-23-76)	--	--	1.14	--	0.28	0.84	--	0.75
Effluent (Pond 2)(4-1-76)	--	--	3.60	--	0.20	0.74	--	0.76
Effluent (Pond 2)(4-3-76)	--	--	14.0	--	0.18	1.12	--	0.104
Effluent (Pond 2)(4-4-76)	--	--	8.4	--	0.17	0.64	--	0.63
Effluent (Pond 2)(4-5-76)	--	--	4.00	--	0.14	0.73	--	0.60
Effluent (Pond 2)(4-6-76)	--	--	1.89	--	0.20	0.68	--	0.64
Slip 1 sediment (Area 6)	--	--	18,300	39,200	40.0	180	369	9.76
Slip 1 sediment (Area 5)	--	--	24,500	51,500	8.4	240	494	5.28
Slip 1 sediment (Areas 4 and 3)	--	--	21,000	51,700	0.20	220	541	0.22
Slip 1 sediment (Area 2)	--	--	21,800	49,400	0.41	250	565	1.92
Slip 1 sediment (Area 1-PCB spill site)	--	--	25,100	59,100	4.0	250	585	1.64
Slip 1 dredge site water (2-25-76) (used for elutriate test)	--	--	1.30	--	--	0.080	--	--

(Continued)

Table B16 (Continued)

	Zinc			Cadmium		
Sample Identification	Total mg/ℓ	Solids mg/kg*	<0.45 µm mg/ℓ	Total mg/ℓ	Solids mg/kg*	<0.45 µm mg/ℓ
Effluent (Pond 2)(3-16-76)	0.252	--	0.228	0.008	--	0.004
Effluent (Pond 2)(3-19-76)	0.480	--	0.216	0.006	--	0.004
Effluent (1)(Pond 2)(3-22-76)	0.400	--	0.148	0.008	--	0.004
Effluent (2)(Pond 2)(3-22-76)	0.224	--	0.100	0.008	--	<0.002
Effluent (Pond 2)(3-23-76)	0.174	--	0.052	0.004	--	<0.002
Effluent (Pond 2)(4-1-76)	0.152	--	0.070	0.002	--	<0.002
Effluent (Pond 2)(4-3-76)	--	--	--	<0.002	--	<0.002
Effluent (Pond 2)(4-4-76)	0.214	--	0.055	<0.002	--	<0.002
Effluent (Pond 2)(4-5-76)	0.134	--	0.044	<0.002	--	<0.002
Effluent (Pond 2)(4-6-76)	0.105	--	0.060	0.003	--	0.003
Slip 1 sediment (Area 6)	120	258	0.010	0.6	1.3	0.004
Slip 1 sediment (Area 5)	610	1280	0.074	2.8	5.9	0.006
Slip 1 sediment (Areas 4 and 3)	1000	2460	<0.002	5.0	12.3	0.004
Slip 1 sediment (Area 2)	310	704	0.010	1.4	3.2	0.004
Slip 1 sediment (Area 1-PCB spill site)	110	260	0.038	0.5	1.2	0.006
Slip 1 dredge site water (2-25-76) (used for elutriate test)	0.020	--	--	0.008	--	--

	Copper			Nickel		
	Total mg/ℓ	Solids mg/kg*	<0.45 µm mg/ℓ	Total mg/ℓ	Solids mg/kg*	<0.45 µm mg/ℓ
Effluent (Pond 2)(3-16-76)	0.036	--	0.034	0.01	--	0.01
Effluent (Pond 2)(3-19-76)	0.048	--	0.036	<0.01	--	<0.01
Effluent (1)(Pond 2)(3-22-76)	0.036	--	0.032	0.03	--	0.03
Effluent (2)(Pond 2)(3-22-76)	0.042	--	0.028	0.02	--	0.02
Effluent (Pond 2)(3-23-76)	0.048	--	0.048	0.02	--	0.02
Effluent (Pond 2)(4-1-76)	0.060	--	0.052	--	--	--
Effluent (Pond 2)(4-3-76)	0.070	--	0.046	--	--	--
Effluent (Pond 2)(4-4-76)	0.065	--	0.053	--	--	--
Effluent (Pond 2)(4-5-76)	0.065	--	0.042	--	--	--
Effluent (Pond 2)(4-6-76)	0.058	--	0.050	--	--	--
Slip 1 sediment (Area 6)	32	69	0.0090	--	--	<0.01
Slip 1 sediment (Area 5)	52	109	0.0096	--	--	<0.01
Slip 1 sediment (Areas 4 and 3)	59	145	0.0048	--	--	<0.01
Slip 1 sediment (Area 2)	42	95	0.0072	--	--	<0.01
Slip 1 sediment (Area 1-PCB spill site)	39	92	0.0060	--	--	<0.01
Slip 1 dredge site water (2-25-76) (used for elutriate test)	0.0072	--	--	<0.01	--	--

(Continued)

Table B16 (Continued)

Sample Identification	Lead			Mercury		
	Total mg/ℓ	Solids mg/kg*	<0.45 µm mg/ℓ	Total mg/ℓ	Solids mg/kg*	<0.45 µm mg/ℓ
Effluent (Pond 2)(3-16-76)	--	--	--	0.0001	--	0.0001
Effluent (Pond 2)(3-19-76)	--	--	--	<0.0002	--	<0.0002
Effluent (1)(Pond 2)(3-22-76)	--	--	--	<0.0002	--	<0.0002
Effluent (2)(Pond 2)(3-22-76)	--	--	--	<0.0002	--	<0.0002
Effluent (Pond 2)(3-23-76)	--	--	--	<0.0002	--	<0.0002
Effluent (Pond 2)(4-1-76)	--	--	--	0.0002	--	<0.0002
Effluent (Pond 2)(4-3-76)	--	--	--	0.0002	--	0.0002
Effluent (Pond 2)(4-4-76)	--	--	--	0.0003	--	0.0002
Effluent (Pond 2)(4-5-76)	--	--	--	0.0002	--	0.0003
Effluent (Pond 2)(4-6-76)	--	--	--	0.0004	--	0.0003
Slip 1 sediment (Area 6)	44	95	--	0.1	0.25	0.0001
Slip 1 sediment (Area 5)	67	140	--	0.1	0.25	0.0010
Slip 1 sediment (Areas 4 and 3)	84	207	--	0.1	0.25	0.0003
Slip 1 sediment (Area 2)	235	533	--	0.1	0.2	0.0001
Slip 1 sediment (Area 1-PCB spill site)	44	103	--	<0.1	<0.2	0.0004
Slip 1 dredge site water (2-25-76) (used for elutriate test)	--	--	--	0.0004	--	--

	Chromium			Arsenic			Chemical Oxygen Demand		Volatile Solids	
	Total mg/ℓ	Solids mg/kg*	<0.45 µm mg/ℓ	Total mg/ℓ	Solids mg/kg*	<0.45 µm mg/ℓ	Total mg/ℓ	Solids mg/kg*	Total %	Solids %
Effluent (Pond 2)(3-16-76)	--	--	--	0.009	--	0.003	--	--	--	--
Effluent (Pond 2)(3-19-76)	--	--	--	0.005	--	0.003	--	--	--	--
Effluent (1)(Pond 2) (3-22-76)	--	--	--	0.012	--	0.021	--	--	--	--
Effluent (2)(Pond 2) (3-22-76)	--	--	--	0.013	--	0.011	--	--	--	--
Effluent (Pond 2)(3-23-76)	--	--	--	0.019	--	0.016	--	--	--	--
Effluent (Pond 2)(4-1-76)	0.024	--	0.024	0.004	--	0.002	--	--	--	--
Effluent (Pond 2)(4-3-76)	0.028	--	0.020	0.006	--	0.0005	--	--	--	--
Effluent (Pond 2)(4-4-76)	0.029	--	0.024	0.013	--	0.0005	--	--	--	--
Effluent (Pond 2)(4-5-76)	0.025	--	0.024	0.008	--	0.001	--	--	--	--
Effluent (Pond 2)(4-6-76)	0.036	--	0.033	0.003	--	0.0005	--	--	--	--
Slip 1 sediment (Area 6)	15	32	0.048	6	13	0.0265	26,200	56,300	7.1	15.3
Slip 1 sediment (Area 5)	22	46	0.044	5	11	0.0204	20,900	43,900	7.5	15.8
Slip 1 sediment (Areas 4 and 3)	20	49	0.043	8	20	0.0215	28,700	70,600	10.4	25.6
Slip 1 sediment (Area 2)	37	84	0.034	7	16	0.0323	28,400	64,500	9.3	21.1
Slip 1 sediment (Area 1-PCB spill site)	21	49	0.015	8	19	0.0212	28,200	66,400	8.9	20.9
Slip 1 dredge site water (2-25-76)(used for elutriate test)	0.016	--	--	0.002	--	--	--	--	--	--

(Continued)

Table B16 (Continued)

Sample Identification	Total Solids %	Non-filterable Solids, %	Settleable Solids mℓ/ℓ	Area of Dredging Activity	Total Organic Carbon Total mg/ℓ	Total Organic Carbon <0.45 μm mg/ℓ	Oil and Grease Total mg/ℓ	Oil and Grease Solids mg/kg*	Oil and Grease <0.45 μm mg/ℓ
Slip 1 standard elutriate (Area 6)	--	--	--	6	--	15	--	--	1.2
Slip 1 standard elutriate (Area 5)	--	--	--	5	--	15	--	--	3.0
Slip 1 standard elutriate (Areas 4 and 3)	--	--	--	3, 4	--	42	--	--	13
Slip 1 standard elutriate (Area 2)	--	--	--	2	--	24	--	--	7.6
Slip 1 standard elutriate (Area 1-PCB spill site)	--	--	--	1	--	17	--	--	1.9

Sample Identification	Total PCB's (1242, 1254, 1260) mg/ℓ	Solids PCB's (Centrifuged Solids) mg/kg*	Soluble PCB's (Centrifuged) mg/ℓ	Organic-N Total mg/ℓ	Organic-N Solids mg/kg*	Organic-N <0.45 μm mg/ℓ
Slip 1 standard elutriate (Area 6)	--	--	0.008	--	--	2.0
Slip 1 standard elutriate (Area 5)	--	--	0.013	--	--	0.8
Slip 1 standard elutriate (Areas 4 and 3)	--	--	0.030	--	--	2.2
Slip 1 standard elutriate (Area 2)	--	--	0.029	--	--	2.0
Slip 1 standard elutriate (Area 1-PCB spill site)	--	--	0.158	--	--	1.2

Sample Identification	Ammonium-N Total mg/ℓ	Ammonium-N Exchangeable mg/kg*	Ammonium-N <0.45 μm mg/ℓ	$NO_3 + NO_2$ -N <0.45 μm mg/ℓ	Ortho-PO_4-P Total mg/ℓ	Ortho-PO_4-P <0.45 μm mg/ℓ	Total-P Total mg/ℓ	Total-P Solids mg/kg*	Total-P <0.45 μm mg/ℓ
Slip 1 standard elutriate (Area 6)	--	--	3.0	0.31	--	--	--	--	0.07
Slip 1 standard elutriate (Area 5)	--	--	2.2	0.29	--	--	--	--	0.19
Slip 1 standard elutriate (Areas 4 and 3)	--	--	2.6	0.30	--	--	--	--	0.52
Slip 1 standard elutriate (Area 2)	--	--	3.8	0.20	--	--	--	--	0.39
Slip 1 standard elutriate (Area 1-PCB spill site)	--	--	3.3	1.40	--	--	--	--	0.11

Sample Identification	Alkalinity as $CaCO_3$ mg/ℓ	Chloride mg/ℓ	Sulfate mg/ℓ	Sulfide Total mg/ℓ	Sulfide Solids mg/kg*	Sulfide <0.45 μm mg/ℓ	Cation Exchange Capacity meq/100 g Soil*	C:N:P Ratio
Slip 1 standard elutriate (Area 6)	--	--	--	--	--	--	--	--
Slip 1 standard elutriate (Area 5)	--	--	--	--	--	--	--	--
Slip 1 standard elutriate (Areas 4 and 3)	--	--	--	--	--	--	--	--
Slip 1 standard elutriate (Area 2)	--	--	--	--	--	--	--	--
Slip 1 standard elutriate (Area 1-PCB spill site)	--	--	--	--	--	--	--	--

(Continued)

(Sheet 12 of 13)

Table B16 (Concluded)

Sample Identification	Calcium Solids mg/kg	Magnesium Solids mg/kg	Potassium Solids mg/kg	Sodium Solids mg/kg	Iron		
					Total mg/ℓ	Solids mg/kg	<0.45 µm mg/ℓ
Slip 1 standard elutriate (Area 6)	--	--	--	--	--	--	0.54
Slip 1 standard elutriate (Area 5)	--	--	--	--	--	--	0.26
Slip 1 standard elutriate (Areas 4 and 3)	--	--	--	--	--	--	0.24
Slip 1 standard elutriate (Area 2)	--	--	--	--	--	--	0.30
Slip 1 standard elutriate (Area 1-PCB spill site)	--	--	--	--	--	--	0.56

	Manganese			Zinc			Cadmium		
	Total mg/ℓ	Solids mg/kg	<0.45 µm mg/ℓ	Total mg/ℓ	Solids mg/kg	<0.45 µm mg/ℓ	Total mg/ℓ	Solids mg/kg	<0.45 µm mg/ℓ
Slip 1 standard elutriate (Area 6)	--	--	3.36	--	--	0.004	--	--	0.004
Slip 1 standard elutriate (Area 5)	--	--	1.92	--	--	0.008	--	--	0.004
Slip 1 standard elutriate (Areas 4 and 3)	--	--	0.224	--	--	<0.002	--	--	0.004
Slip 1 standard elutriate (Area 2)	--	--	1.32	--	--	0.004	--	--	0.008
Slip 1 standard elutriate (Area 1-PCB spill site)	--	--	2.88	--	--	0.012	--	--	0.004

	Copper			Nickel			Lead		
	Total mg/ℓ	Solids mg/kg	<0.45 µm mg/ℓ	Total mg/ℓ	Solids mg/kg	<0.45 µm mg/ℓ	Total mg/ℓ	Solids mg/kg	<0.45 µm mg/ℓ
Slip 1 standard elutriate (Area 6)	--	--	0.0090	--	--	<0.01	--	--	--
Slip 1 standard elutriate (Area 5)	--	--	0.0180	--	--	<0.01	--	--	--
Slip 1 standard elutriate (Areas 4 and 3)	--	--	0.0036	--	--	<0.01	--	--	--
Slip 1 standard elutriate (Area 2)	--	--	0.0072	--	--	<0.01	--	--	--
Slip 1 standard elutriate (Area 1-PCB spill site)	--	--	0.0060	--	--	<0.01	--	--	--

	Mercury			Chromium			Arsenic		
	Total mg/ℓ	Solids mg/kg	<0.45 µm mg/ℓ	Total mg/ℓ	Solids mg/kg	<0.45 µm mg/ℓ	Total mg/ℓ	Solids mg/kg	<0.45 µm mg/ℓ
Slip 1 standard elutriate (Area 6)	--	--	0.0001	--	--	0.047	--	--	0.0117
Slip 1 standard elutriate (Area 5)	--	--	0.0006	--	--	0.047	--	--	0.0069
Slip 1 standard elutriate (Areas 4 and 3)	--	--	0.0002	--	--	0.043	--	--	0.0159
Slip 1 standard elutriate (Area 2)	--	--	0.0001	--	--	0.043	--	--	0.0122
Slip 1 standard elutriate (Area 1-PCB spill site)	--	--	0.0001	--	--	0.045	--	--	0.0162

(Continued)

Table B17

Physical and Chemical Composition of the Waterways Experiment Station Disposal Area Influents and Effluents and Brown Lake Background Water, Vicksburg, Mississippi

Sample Identification	Particle Size (Mechanical) % <2 μm	% 2-50 μm	% >50 μm	Coulter Counter* Particle Size μm >50%	>80%	Total Solids %	Nonfilterable Solids, %	Settleable Solids mℓ/ℓ
Background water (3-2-76) (surface water)	--	--	--	5.5	2.85	0.0428	0.00204	<0.1
Influent (3-17-76)	23	67	10	0.86	0.55	32.00	32.00	999
Influent (4-15-76)	22	78	0	0.92	0.56	15.00	14.60	900
Effluent (3-17-76)	--	--	--	9.6	3.7	0.570	0.520	32
Effluent (4-12-76)	--	--	--	10.5	3.8	0.220	0.150	4
Effluent (4-15-76)	--	--	--	6.6	2.8	0.100	0.0726	4.5

(Continued)

Note: -- indicates missing data.

* Influents: determined for <2-μm fraction from mechanical particle-size analysis, sodium hexametaphosphate added for dispersion; effluents determined for total effluent, no dispersive agent added; dispersed by sonication.

(Sheet 1 of 9)

Table B17 (Continued)

Sample Identification	Total Organic Carbon Total mg/ℓ	 Solids mg/kg**	 <0.45 µm mg/ℓ	Oil and Grease Total mg/ℓ	 Solids mg/kg**	 <0.45 µm mg/ℓ	Chlorinated Pesticides† op' DDE mg/ℓ	 pp' DDE mg/ℓ	Total PCB (1242, 1254, 1260) mg/ℓ
Background water (3-2-76)	10	--	6	--	--	--	--	--	--
Influent (3-17-76)	2667	8300	16	--	--	--	0.17	0.92	8.4
Influent (4-15-76)	974	6600	12	--	--	5.0	--	--	--
Effluent (3-17-76)	14	--	8	--	--	--	<0.01	0.02	<0.1
Effluent (4-12-76)	11	--	8	3.8	--	--	--	--	--
Effluent (4-15-76)	22	--	8	4.0	--	--	--	--	--

(Continued)

** Dry weight.

† op' DDD, pp' DDD, op' DDT, pp' DDT, dieldrin, aldrin, lindane, heptachlor, heptachlor epoxide, and chlordane are below detection limits.

(Sheet 2 of 9)

Table B17 (Continued)

	Organic-N††			Ammonium-N		
Sample Identification	Total mg/ℓ	Solids mg/kg**	<0.45 µm mg/ℓ	Total mg/ℓ	Exchangeable‡ mg/kg** (mg/ℓ)‡‡	<0.45 µm mg/ℓ
Background water (3-2-76)	0.45	--	0.45	0.36	--	0.54
Influent (3-17-76)	219	680	1.45	--	118.0 (37.8)	8.55
Influent (4-15-76)	95.5	640	2.35	--	98.0 (14.7)	5.35
Effluent (3-17-76)	4.0	--	2.8	7.59	--	4.85
Effluent (4-12-76)	2.65	--	2.05	9.05	--	3.29
Effluent (4-15-76)	3.1	--	3.1	6.58	--	4.10

(Continued)

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†† TKN - (<0.45 µm) ammonium-N.

‡ Sodium acetate extractable.

‡‡ Values in parentheses as (mg/ℓ) exchangeable ammonium-N in total sample.

Table B17 (Continued)

Sample Identification	NO_3+NO_2 -N <0.45 µm mg/ℓ	Ortho-PO_4 - P <0.45 µm mg/ℓ	Total-P Total mg/ℓ	Total-P Solids mg/kg**	Total-P <0.45 µm mg/ℓ	Alkalinity as $CaCO_3$ mg/ℓ	Chloride mg/ℓ
Background water (3-2-76)	1.98	0.15	0.15	--	0.15	290	35
Influent (3-17-76)	0.015	0.07	325	1016	0.08	308	75
Influent (4-15-76)	0.70	0.10	369	2528	0.14	263	95
Effluent (3-17-76)	0.07	0.03	6.6	--	0.09	202	65
Effluent (4-12-76)	0.90	0.12	1.63	--	0.14	273	55
Effluent (4-15-76)	0.60	0.13	1.24	--	0.13	237	95

(Continued)

Table B17 (Continued)

Sample Identification	Total Sulfide Total mg/ℓ	 Solids mg/kg**	Cation Exchange Capacity meq/100 g**	Total C:N:P Ratio	Calcium Total mg/ℓ	 Solids mg/kg**	 <0.45 μm mg/ℓ
Background water (3-2-76)	1.4	--	--	67:6.6:1	59	--	68
Influent (3-17-76)	15.4	47.9	18.3	8.2:0.7:1	8916	27,700	64
Influent (4-15-76)	4.9	26.9	2.37	2.6:0.3:1	1045	6,800	60
Effluent (3-17-76)	1.8	--	--	2:1.3:1	180	25,400	48
Effluent (4-12-76)	0.7	--	--	7:3.7:1	100	26,100	61
Effluent (4-15-76)	1.1	--	--	18:5.8:1	53	--	54

(Continued)

** Dry weight.

Table B17 (Continued)

Sample Identification	Magnesium			Potassium			Sodium		
	Total mg/ℓ	Solids mg/kg**	<0.45 μm mg/ℓ	Total mg/ℓ	Solids mg/kg**	<0.45 μm mg/ℓ	Total mg/ℓ	Solids mg/kg**	<0.45 μm mg/ℓ
Background water (3-2-76)	31.4	--	33	5	--	9	--	--	32
Influent (3-17-76)	315	933	27	5875	18,350	4.6	3180	9,850	64
Influent (4-15-76)	3055	20,750	32	6360	43,500	7.6	1860	12,650	65
Effluent (3-17-76)	99.4	15,100	21	85.2	14,700	8.8	126	12,950	58
Effluent (4-12-76)	51.4	16,300	27	33.2	14,150	12	115	43,200	49
Effluent (4-15-76)	30.4	4,700	27	18.2	16,700	6.1	151	--	68

(Continued)

** Dry weight.

Table B17 (Continued)

Sample Identification	Iron			Manganese			Zinc		
	Total mg/ℓ	Solids mg/kg**	<0.45 µm mg/ℓ	Total mg/ℓ	Solids mg/kg**	<0.45 µm mg/ℓ	Total mg/ℓ	Solids mg/kg**	<0.45 µm mg/ℓ
Background water (3-2-76)	1.84	--	<0.001	0.403	--	0.016	0.74	--	0.010
Influent (3-17-76)	7,780	24,300	11.9	256.6	796	2.88	25.2	78.4	0.115
Influent (4-15-76)	11,900	81,650	9.0	310.1	2105	3.43	37.6	257	0.031
Effluent (3-17-76)	195	37,500	0.016	5.03	772	1.02	2.47	474	0.003
Effluent (4-12-76)	54.5	36,300	0.010	2.13	1360	0.09	5.49	3655	0.006
Effluent (4-15-76)	32.2	44,300	0.014	2.23	1270	1.31	0.026	31.7	0.003

(Continued)

** Dry weight.

Table B17 (Continued)

Sample Identification	Cadmium			Copper			Nickel		
	Total mg/ℓ	Solids mg/kg**	<0.45 µm mg/ℓ	Total mg/ℓ	Solids mg/kg**	<0.45 µm mg/ℓ	Total mg/ℓ	Solids mg/kg**	<0.45 µm mg/ℓ
Background water (3-2-76)	<0.0002	--	--	--	--	0.006	0.018	--	0.004
Influent (3-17-76)	0.040	7.6	0.0002	6.65	20.8	0.001	14.6	45	<0.003
Influent (4-15-76)	0.013	8.5	0.0002	4.68	32	0.002	18.2	124	<0.003
Effluent (3-17-76)	<0.002	--	0.0001	0.195	36.5	0.005	0.207	39.8	0.004
Effluent (4-12-76)	<0.002	--	0.0001	0.042	26	0.003	0.112	74.6	<0.003
Effluent (4-15-76)	<0.002	--	0.0003	0.038	48.2	0.003	0.043	59.2	<0.003

(Continued)

** Dry weight.

(Sheet 8 of 9)

Table B17 (Concluded)

	Lead			Mercury		
Sample Identification	Total mg/ℓ	Solids mg/kg**	<0.45 μm mg/ℓ	Total mg/ℓ	Solids mg/kg**	<0.45 μm mg/ℓ
Background water (3-2-76)	<0.001	--	<0.001	<0.0002§	--	<0.0002§
Influent (3-17-76)	5.89	18.4	<0.001	0.022	0.07	
Influent (4-15-76)	17.3	119	<0.001	0.243	1.66	0.0002§
Effluent (3-17-76)	0.17	32.7	<0.001	0.1594	--	0.0034
Effluent (4-12-76)	0.006	4	<0.001	0.0015	1	<0.0002§
Effluent (4-15-76)	<0.001	<1	<0.001	0.0023	3.2	<0.0002§

** Dry weight.

§ Cold vapor technique.

(Sheet 9 of 9)

Table B18a

Physical and Chemical Composition of the Oak Island Disposal Area, Influents and Effluents, and Atlantic Intracoastal Waterway Background Water, Southport, North Carolina, Initial Sample Collection

Sample Identification	Particle Size (Mechanical) % <2 µm	% 2-50 µm	% >50 µm	Nonfilterable Solids, %	Settleable Solids mℓ/ℓ	Total Organic Carbon Total mg/ℓ	Solids mg/kg*	<0.45 µm mg/ℓ
Background water (5-6-76) (surface water)	--	--	--	0.0094	<0.1	4	--	8
Influent (5-6-76)	39	19	42	32.20	600	6960	21,600	8
Influent (5-6-76)	43	28	29	8.30	450	3000	36,100	3
Influent (5-7-76)	65	32	3	15.00	450	5730	38,200	12
Effluent (5-6-76)	--	--	--	0.0134	<0.1	8	--	7
Effluent (5-7-76)	--	--	--	0.0077	<0.1	4	--	9
Effluent (5-7-76)	--	--	--	0.0246	<0.1	4	--	6

	Oil and Grease Total mg/ℓ	Solids mg/kg*	<0.45 µm mg/ℓ	Organic-N** Total** mg/ℓ	Solids mg/kg*	<0.45 µm mg/ℓ	Ammonium-N Total mg/ℓ	Exchangeable† mg/kg* (mg/ℓ)††	<0.45 µm mg/ℓ
Background water (5-6-76)	15.6	--	--	2.35	--	0.9	<0.01	--	<0.01
Influent (5-6-76)	--	--	1.8	630	1950	3.8	--	2.79 (0.9)	4.0
Influent (5-6-76)	--	--	3.0	178	2140	0.75	--	2.40 (0.2)	0.66
Influent (5-7-76)	--	--	--	442	2940	0.7	--	130.8 (19.6)	37.1
Effluent (5-6-76)	11.6	--	--	0.85	--	0.85	0.82	--	0.74
Effluent (5-7-76)	2.4	--	--	1.3	--	0.6	2.06	--	1.40
Effluent (5-7-76)	3.8	--	--	0.85	--	0.9	2.47	--	1.60

	$NO_3 + NO_2$ -N <0.45 µm mg/ℓ	Ortho-PO_4-P <0.45 µm mg/ℓ	Total-P Total mg/ℓ	Solids mg/kg*	<0.45 µm mg/ℓ	Alkalinity as $CaCO_3$ mg/ℓ	Chloride mg/ℓ
Background water (5-6-76)	<0.010	<0.03	0.07	--	<0.5	101	20,500
Influent (5-6-76)	0.042	0.20	496	1540	<0.5	396	16,100
Influent (5-6-76)	<0.010	0.26	89	1065	<0.5	171	15,000
Influent (5-7-76)	<0.010	0.78	108	715	0.8	970	17,000
Effluent (5-6-76)	0.100	0.09	0.29	--	--	182	15,100
Effluent (5-7-76)	0.043	0.22	0.44	--	--	177	13,000
Effluent (5-7-76)	0.020	0.20	0.46	--	--	175	15,600

	Total Sulfide Total mg/ℓ	Solids mg/kg*	Cation Exchange Capacity meq/100 g*	Total C:N:P Ratio	Calcium Total mg/ℓ	Solids mg/kg*	<0.45 µm mg/ℓ
Background water (5-6-76)	0.8	--	--	57:34:1	340	--	330
Influent (5-6-76)	81	247	43.4	14:1.3:1	10,460	31,680	380
Influent (5-6-76)	--	--	55.5	34:2.0:1	2,960	31,730	360
Influent (5-7-76)	123	811	66.1	53:4.4:1	5,390	33,910	360
Effluent (5-6-76)	2.4	--	--	28:5.5:1	390	--	390
Effluent (5-7-76)	1.4	--	--	9:6.1:1	380	--	360
Effluent (5-7-76)	0.9	--	--	9:5.3:1	380	--	380

(Continued)

Note: dashes (--) indicate missing data.
* Dry weight.
** TKN - (<0.45 µm) ammonium-N.
† Sodium acetate extractable.
†† Values in parentheses as (mg/ℓ) ammonium-N in total sample.

Table B18a (Concluded)

Sample Identification	Magnesium			Potassium			Sodium		
	Total mg/ℓ	Solids mg/kg*	<0.45 µm mg/ℓ	Total mg/ℓ	Solids mg/kg*	<0.45 µm mg/ℓ	Total mg/ℓ	Solids mg/kg*	<0.45 µm mg/ℓ
Background water (5-6-76)	1200	--	1100	320	--	370	8000	--	7600
Influent (5-6-76)	3080	6,820	1300	3525	10,000	450	--	--	9500
Influent (5-6-76)	1710	9,575	1000	1290	10,780	430	9850	--	8900
Influent (5-7-76)	2560	10,815	1100	1920	10,920	330	--	--	9300
Effluent (5-6-76)	1100	--	1100	340	--	440	8500	--	8100
Effluent (5-7-76)	1200	--	1100	360	--	440	9300	--	8900
Effluent (5-7-76)	1200	--	1100	360	--	410	8600	--	8400

	Iron			Manganese			Zinc		
	Total mg/ℓ	Solids mg/kg*	<0.45 µm mg/ℓ	Total mg/ℓ	Solids mg/kg*	<0.45 µm mg/ℓ	Total mg/ℓ	Solids mg/kg*	<0.45 µm mg/ℓ
Background water (5-6-76)	1.35	--	1.43	0.080	--	0.082	1.22	--	0.093
Influent (5-6-76)	10,500	32,615	1.39	174.3	539	1.07	18.3	55.8	0.496
Influent (5-6-76)	2,900	34,915	1.31	42.8	509	0.652	7.79	92.8	0.093
Influent (5-7-76)	6,125	40,795	6.63	77.6	484	5.93	17.1	113.4	0.098
Effluent (5-6-76)	--	--	1.18	0.489	--	0.483	1.50	--	0.098
Effluent (5-7-76)	1.62	--	1.26	0.670	--	0.661	1.03	--	0.099
Effluent (5-7-76)	1.27	--	1.40	0.722	--	0.711	1.29	--	0.101

	Cadmium			Copper			Nickel		
	Total mg/ℓ	Solids mg/kg*	<0.45 µm mg/ℓ	Total mg/ℓ	Solids mg/kg*	<0.45 µm mg/ℓ	Total mg/ℓ	Solids mg/kg*	<0.45 µm mg/ℓ
Background water (5-6-76)	0.0096	--	0.0098	0.026	--	0.026	--	--	0.033
Influent (5-6-76)	0.62	1.9	0.0150	1.94	6.0	0.004	6.6	20.4	0.035
Influent (5-6-76)	0.59	7.0	0.0098	1.01	12.2	0.002	3.9	47.1	0.030
Influent (5-7-76)	0.57	3.8	0.0081	2.82	18.6	0.036	7.7	51.3	0.033
Effluent (5-6-76)	0.0076	--	0.0100	0.024	--	0.005	0.15	--	0.034
Effluent (5-7-76)	0.0098	--	0.0082	0.027	--	0.007	0.11	--	0.043
Effluent (5-7-76)	0.0112	--	0.0115	0.023	--	0.026	0.19	--	0.035

	Lead			Mercury		
	Total mg/ℓ	Solids mg/kg*	<0.45 µm mg/ℓ	Total mg/ℓ	Solids mg/kg*	<0.45 µm mg/ℓ
Background water (5-6-76)	0.002	--	0.001	0.0086	--	0.0017
Influent (5-6-76)	7.6	23.6	<0.001	0.075	0.23	0.0047
Influent (5-6-76)	1.96	23.7	<0.001	0.06	0.72	0.0017
Influent (5-7-76)	3.99	26.6	<0.001	0.05	0.33	--
Effluent (5-6-76)	0.032	--	<0.001	0.0084	--	0.0048
Effluent (5-7-76)	0.056	--	<0.001	0.0089	--	0.0017
Effluent (5-7-76)	0.030	--	0.002	0.0088	--	0.0009

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* Dry weight.

Table B18b

Physical and Chemical Composition of the Oak Island Disposal Area, Influents and Effluents, and Atlantic Intracoastal Waterway Background Water, Southport, North Carolina, Final Sample Collection

Sample Identification	Particle Size (Mechanical) % <2 µm	% 2-50 µm	% >50 µm	Non-filterable Solids, %	Settleable Solids mℓ/ℓ	Total Organic Carbon Total mg/ℓ	Solids mg/kg*	<0.45 µm mg/ℓ
Background water (5-17-76) (surface water)	--	--	--	0.02697	<0.1	--	--	23
Influent (5-17-76)	48	47	5	30.70	980	11,450	37,300	19
Influent (5-19-76)	38	49.5	12.5	17.60	600	6,620	37,600	7
Influent (5-20-76)	39	53	8	3.00	500	114	38,000	8
Effluent (5-17-76)	--	--	--	0.0484	0.7	26	--	10
Effluent (5-17-76)	--	--	--	0.3660	<0.1	20	--	10
Effluent (5-20-76)	--	--	--	0.1000	5	46	--	14

Sample Identification	Oil and Grease Total mg/ℓ	Solids mg/kg*	<0.45 µm mg/ℓ	Total Chlorinated Pesticides** op'DDE mg/ℓ	pp'DDE mg/ℓ	op'DDD mg/ℓ	pp'DDD mg/ℓ	op'DDT mg/ℓ	pp'DDT mg/ℓ
Background water (5-17-76)	1.2	--	--	--	--	--	--	--	--
Influent (5-17-76)	--	--	--	0.048	0.057	0.51	0.625	4.08	2.10
Influent (5-19-76)	--	--	--	<0.01	<0.01	0.48	0.44	4.64	<0.01
Influent (5-20-76)	--	--	--	<0.01	<0.01	1.28	0.88	5.40	5.94
Effluent (5-17-76)	--	--	--	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01
Effluent (5-17-76)	--	--	--	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01
Effluent (5-20-76)	--	--	--						

Sample Identification	Total PCB's (1242, 1254, 1260) mg/ℓ	Organic-N† Total mg/ℓ	Solids mg/kg*	<0.45 µm mg/ℓ	Ammonium-N Total mg/ℓ	Exchangeable†† mg/kg* (mg/ℓ)‡	<0.45 µm mg/ℓ
Background water (5-17-76)	--	0.8	--	1.35	0.55	--	<0.01
Influent (5-17-76)	9.0	839	2670	27.6	--	130.5 (40.1)	27.7
Influent (5-19-76)	1.92	511	2850	11.8	--	115.8 (20.4)	36.2
Influent (5-20-76)	6.72	100	3170	4.9	--	110.6 (3.3)	24.1
Effluent (5-17-76)	<0.1	7.4	--	4.8	14.6	119.0 (0.06)	14.2
Effluent (5-17-76)	<0.1	1.9	--	4.4	18.1	--	13.6
Effluent (5-20-76)		--	--	--	20.3	--	14.6

Sample Identification	$NO_3 + NO_2$-N <0.45 µm mg/ℓ	Ortho-PO_4-P <0.45 µm mg/ℓ	Total-P Total mg/ℓ	Solids mg/kg*	<0.45 µm mg/ℓ	Alkalinity as $CaCO_3$ mg/ℓ	Chloride mg/ℓ
Background water (5-17-76)	--	0.03	0.18	--	0.10	112	17,600
Influent (5-17-76)	--	5.89	201	639	6.31	1523	15,800
Influent (5-19-76)	--	5.14	153	826	9.47	736	15,500
Influent (5-20-76)	--	1.91	32	990	2.57	813	14,200
Effluent (5-17-76)	--	0.97	1.83	--	1.06	670	16,400
Effluent (5-17-76)	--	1.04	1.69	--	1.46	654	16,000
Effluent (5-20-76)	--	0.99	2.04	--	1.39	568	16,300

(Continued)

Note: dashes (--) indicate missing data.

* Dry weight.

** Dieldrin, aldrin, lindane, heptachlor, heptachlor epoxide, and chlordane are below detection limits.

† TKN - (<0.45 µm) ammonium-N.

†† Sodium acetate extractable.

‡ Values in parentheses as (mg/ℓ) exchangeable ammonium-N in total sample.

Table B18b (Concluded)

Sample Identification	Total Sulfide Total mg/ℓ	Total Sulfide Solids mg/kg*	Cation Exchange Capacity meq/100 g*	Total C:N:P Ratio	Calcium Total mg/ℓ	Calcium Solids mg/kg*	Calcium <0.45 μm mg/ℓ
Background water (5-17-76)	0.2	--	--	128:4.4:1	390	--	390
Influent (5-17-76)	153	496	48.3	57:4.3:1	11,450	36,510	350
Influent (5-19-76)	189	1069	55.7	43:3.6:1	5,785	31,230	350
Influent (5-20-76)	25	782	69.8	4:3.9:1	1,445	37,850	320
Effluent (5-17-76)	2.7	--	--	14:11.8:1	440	--	370
Effluent (5-17-76)	3.2	--	--	12:9.2:1	410	--	380
Effluent (5-20-76)	2.1	--	--	23:9.3:1	420	--	360

Sample Identification	Magnesium Total mg/ℓ	Magnesium Solids mg/kg*	Magnesium <0.45 μm mg/ℓ	Potassium Total mg/ℓ	Potassium Solids mg/kg*	Potassium <0.45 μm mg/ℓ	Sodium Total mg/ℓ	Sodium Solids mg/kg*	Sodium <0.45 μm mg/ℓ
Background water (5-17-76)	1300	--	1300	320	--	380	9,800	--	9700
Influent (5-17-76)	3620	9315	1100	3810	11,695	320	8,705	--	9000
Influent (5-19-76)	2470	8885	1100	2150	10,710	320	7,065	--	8900
Influent (5-20-76)	1250	9250	1000	610	10,580	300	8,710	--	7600
Effluent (5-17-76)	950	--	1000	360	--	340	10,065	--	9400
Effluent (5-17-76)	800	--	1200	300	--	340	7,735	--	9200
Effluent (5-20-76)	850	--	1200	360	--	340	11,330	--	9200

Sample Identification	Iron Total mg/ℓ	Iron Solids mg/kg*	Iron <0.45 μm mg/ℓ	Manganese Total mg/ℓ	Manganese Solids mg/kg*	Manganese <0.45 μm mg/ℓ	Zinc Total mg/ℓ	Zinc Solids mg/kg*	Zinc <0.45 μm mg/ℓ
Background water (5-17-76)	1.12	--	0.881	0.074	--	0.054	1.28	--	0.121
Influent (5-17-76)	12,635	41,150	0.930	132.6	427	2.14	30.9	100.5	0.105
Influent (5-19-76)	6,400	36,350	2.14	78.7	434	2.80	18.1	102.2	0.119
Influent (5-20-76)	1,305	43,490	1.38	14.8	416	2.40	4.62	150.7	0.104
Effluent (5-17-76)	215	--	0.789	2.06	--	1.50	0.166	--	0.107
Effluent (5-17-76)	4.64	--	0.858	1.83	--	1.77	0.160	--	0.116
Effluent (5-20-76)	32.3	--	0.919	2.27	--	1.47	0.264	--	0.102

Sample Identification	Cadmium Total mg/ℓ	Cadmium Solids mg/kg*	Cadmium <0.45 μm mg/ℓ	Copper Total mg/ℓ	Copper Solids mg/kg*	Copper <0.45 μm mg/ℓ
Background water (5-17-76)	0.0100	--	0.0120	0.024	--	0.028
Influent (5-17-76)	4.00	4.2	0.0092	7.38	23.8	0.106
Influent (5-19-76)	4.04	12.1	0.0092	3.54	19.9	0.046
Influent (5-20-76)	1.50	45.3	0.0060	1.23	40.0	0.031
Effluent (5-17-76)	0.0051	--	0.0104	0.090	--	0.100
Effluent (5-17-76)	0.015	--	0.0104	0.061	--	0.035
Effluent (5-20-76)	0.023	--	0.0096	0.099	--	0.030

Sample Identification	Nickel Total mg/ℓ	Nickel Solids mg/kg*	Nickel <0.45 μm mg/ℓ	Lead Total mg/ℓ	Lead Solids mg/kg*	Lead <0.45 μm mg/ℓ	Mercury Total mg/ℓ	Mercury Solids mg/kg*	Mercury <0.45 μm mg/ℓ
Background water (5-17-76)	--	--	0.036	0.002	--	0.003		--	0.0009
Influent (5-17-76)	10.7	34.7	0.047	--	--	0.003	--	--	0.0049
Influent (5-19-76)	7.3	41.2	0.042	1.01	5.7	0.001	--	--	0.0020
Influent (5-20-76)	1.2	38.6	0.010	0.46	15.3	<0.001	--	--	--
Effluent (5-17-76)	0.074	--	0.036	0.021	--	<0.001	--	--	0.0047
Effluent (5-17-76)	0.075	--	0.034	0.008	--	0.002	--	--	0.0017
Effluent (5-20-76)	0.088	--	0.010	--	--	0.007	--	--	0.0009

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* Dry weight.

Table B19

Geochemical Partitioning of Influent and Effluent Solids from Four Confined Land Disposal Areas and of a Dredge Site Sediment

Sample Identification	Geochemical Partitioning Phase	Calcium		Magnesium		Potassium		Sodium	
		mg/kg*	%	mg/kg*	%	mg/kg*	%	mg/kg*	%
Wilmington, N. C. influent (1-22-76)	A. Exchangeable	2887	46.4	1588	24.0	557.0	5.4	514.0	11.1
	B. Carbonate	1276	20.5	376	5.7	73.6	0.7	98.8	2.1
	C. Easily reducible	469	7.5	262	4.0	40.8	0.4	5.03	0.1
	D. Acid digest (organic-moderately reducible-residual)	1596	25.6	4386	66.3	9,561.0	93.5	4005 **	86.8**
	Total	6228	100.0	6612	100.0	10,232	100.0	4625	100.0
Wilmington, N. C. effluent (1-23-76)	A. Exchangeable	3232	48.8	1794	30.1	703	6.7	1622	--
	B. Carbonate	1266	19.1	230	3.9	26.6	0.3	11.6	--
	C. Easily reducible	506	7.6	280	4.7	76.4	0.7	15.7	--
	D. Acid digest (organic-moderately reducible-residual)	1625	24.5	3661	61.3	9,732	92.3	--	--
	Total	6629	100.0	5965	100.0	10,538	100.0	--	--

Sample Identification	Geochemical Partitioning Phase	Iron		Manganese		Zinc		Lead	
		mg/kg*	%	mg/kg*	%	mg/kg*	%	mg/kg*	%
Wilminton, N. C. influent (1-22-76)	A. Exchangeable	32.1	0.1	82.6	9.9	5.46	2.9	†	<0.1
	B. Carbonate	1,488	2.9	312	37.6	41.0	21.7	†	<0.1
	C. Easily reducible	7,314	14.3	179	21.5	28.6	15.1	22.3	45.9
	D. Acid digest (organic-moderately reducible-residual)	42,342	82.7	257	31.0	114	60.3	26.3	54.1
	Total	51,176	100.0	831	100.0	189	100.0	48.6	100.0
Wilmington, N. C. effluent (1-23-76)	A. Exchangeable	57.9	0.1	303	25.5	6.18	2.3	1.7	2.4
	B. Carbonate	4,619	8.8	446	37.5	134	49.6	28.9	40.7
	C. Easily reducible	8,395	15.9	151	12.7	27.2	10.1	22.5	31.7
	D. Acid digest (organic-moderately reducible-residual)	39,616	75.2	289	24.3	103	38.0	17.9	25.2
	Total	52,688	100.0	1189	100.0	270	100.0	71.0	100.0

Sample Identification	Geochemical Partitioning Phase	Cadmium		Copper		Nickel		Mercury	
		mg/kg*	%	mg/kg*	%	mg/kg*	%	mg/kg*	%
Wilmington, N. C. influent (1-22-76)	A. Exchangeable	0.067	28.5	0.21	0.5	3.84	8.2	0.0046	0.5
	B. Carbonate	0.112	47.7	3.16	8.0	5.8	12.4	†	<0.1
	C. Easily reducible	0.056	23.8	7.32	18.4	2.57	5.5	†	<0.1
	D. Acid digest (organic-moderately reducible-residual)	**,††	††	29.0	73.1	34.6	73.9	0.991	99.5
	Total	0.235	100.0	39.7	100.0	46.8	100.0	0.996	100.0
Wilmington, N. C. effluent (1-23-76)	A. Exchangeable	0.022	1.2	†	<0.1	4.4	6.6	†	<0.1
	B. Carbonate	1.73	94.5	9.77	21.8	6.07	9.2	†	<0.1
	C. Easily reducible	0.079	4.3	8.09	18.1	†	<0.1	0.225	17.2
	D. Acid digest (organic-moderately reducible-residual)	**,††	††	26.9	60.1	55.8	84.2	1.08	82.8
	Total	1.83	100.0	44.8	100.0	66.3	100.0	1.305	100.0

(Continued)

Note: dashes (--) indicate missing data.
A. Exchangeable = ammonium acetate extractable.
B. Carbonate = 1 M acetic acid extractable.
C. Easily reducible = 0.1 M hydroxylamine hydrochloride in 0.01 M nitric acid extractable.
D. Acid digest = hydrofluoric-nitric-fuming nitric acid digestion.
E. Moderately reducible-residual = hydrofluoric-nitric-fuming nitric acid digestion.
* Dry weight.
** Value derived indirectly from total acid digest data.
† Below detection limit.
†† Total acid digest data indicate very low levels in this partitioning phase.

Table B19 (Continued)

Sample Identification	Geochemical Partitioning Phase	Calcium		Magnesium		Potassium		Sodium	
		mg/kg	%	mg/kg	%	mg/kg	%	mg/kg	%
Richmond, Va. influent (2-21-76)	A. Exchangeable	1756	20.9	181	3.9	105	0.6	20.7	0.9
	B. Carbonate	1225	14.6	375	8.1	58.0	0.3	87.7	3.6
	C. Easily reducible	927	11.1	133	2.9	30.2	0.2	35.8	1.5
	D. Acid digest (organic-moderately reducible-residual)	4482	53.4	3929	85.1	16,875	98.9	2250 **	94.0**
	Total	8390	100.0	4618	100.0	17,068	100.0	2395	100.0
Richmond, Va. effluent (2-21-76)	A. Exchangeable	2653	58.0	372	10.9	202	1.7	68.7	28.6
	B. Carbonate	694	15.2	75.2	2.2	26.2	0.2	11.1	4.6
	C. Easily reducible	379	8.3	52.9	1.5	1.96	<0.1	4.58	1.9
	D. Acid digest (organic-moderately reducible-residual)	847	18.5	2917	85.4	11,736	98.1	156	64.9**
	Total	4573	100.0	3417	100.0	11,966	100.0	240	100.0

Sample Identification	Geochemical Partitioning Phase	Iron		Manganese		Zinc		Lead	
		mg/kg	%	mg/kg	%	mg/kg	%	mg/kg	%
Richmond, Va. influent (2-21-76)	A. Exchangeable	1,428	3.2	272	22.8	12.1	5.2	4.7	7.2
	B. Carbonate	1,297	2.9	204	17.1	34.0	14.7	†	<0.1
	C. Easily reducible	2,292	5.2	60.1	5.0	34.2	14.7	33.5	51.5
	D. Acid digest (organic-moderately reducible-residual)	39,170	88.7	659	55.1	152	65.4	26.8	41.3
	Total	44,187	100.0	1195	100.0	232	100.0	65.0	100.0
Richmond, Va. effluent (2-21-76)	A. Exchangeable	76	0.1	44.3	5.2	24.4	4.1	7.3	4.3
	B. Carbonate	1,313	2.3	170	20.0	247	41.1	17.5	10.4
	C. Easily reducible	3,680	6.3	188	22.1	54.6	9.1	39.2	23.3
	D. Acid digest (organic-moderately reducible-residual)	53,382	91.3	450	52.7	275	45.7	104.2	62.0
	Total	58,451	100.0	852	100.0	601	100.0	168	100.0

Sample Identification	Geochemical Partitioning Phase	Cadmium		Copper		Nickel		Mercury	
		mg/kg	%	mg/kg	%	mg/kg	%	mg/kg	%
Richmond, Va. influent (2-21-76)	A. Exchangeable	0.017	2.3	†	<0.1	3.4	4.5	†	<0.1
	B. Carbonate	0.096	12.3	1.47	2.8	4.3	5.8	†	<0.1
	C. Easily reducible	0.145	18.6	9.94	19.2	3.8	5.8	†	<0.1
	D. Acid digest (organic-moderately reducible-residual)	0.52**	66.8	40.5	78.0	63.8	84.7	0.518	100.0
	Total	0.78**	100.0	51.9	100.0	75.3	100.0	0.518	100.0
Richmond, Va. effluent (2-21-76)	A. Exchangeable	1.32	27.2	4.59	3.9	3.57	4.1	--	--
	B. Carbonate	1.16	23.8	19.9	16.9	4.08	4.7	†	<0.1
	C. Easily reducible	†	<0.1	15.7	13.3	†	<0.1	†	<0.1
	D. Acid digest (organic-moderately reducible-residual)	2.38**	49.0	77.6	65.9	79.2	91.2	0.285	100.0
	Total	4.86**	100.0	117.8	100.0	86.9	100.0	0.285	100.0

(Continued)

** Value derived indirectly from total acid digest data.

† Below detection limit.

Table B19 (Continued)

Sample Identification	Geochemical Partitioning Phase	Calcium mg/kg	Calcium %	Magnesium mg/kg	Magnesium %	Potassium mg/kg	Potassium %	Sodium mg/kg	Sodium %
Lake Charles, La. influent (2-4-76)	A. Exchangeable	1550	33.7	1464	22.4	1,255	6.5	2785	89.0
	B. Carbonate	579	12.6	268	4.1	153	0.8	328	10.5
	C. Easily reducible	843	18.3	351	5.4	54.9	0.3	14.7	0.5
	D. Acid digest (organic-moderately reducible-residual)	1625	35.4	4464	68.1	17,768	92.4	**,††	**,††
	Total	4597	100.0	6547	100.0	19,231	100.0	3130	100.0
Lake Charles, La. effluent (2-4-76)	A. Exchangeable	1748	25.3	1129	20.1	1,393	7.0	2640	78.0
	B. Carbonate	902	13.1	443	7.9	271	1.4	705	21.0
	C. Easily reducible	984	14.3	474	8.4	87.8	0.4	31.0	1.0
	D. Acid digest (organic-moderately reducible-residual)	3267	47.3	3583	63.6	18,250	91.2	**,††	**,††
	Total	6901	100.0	5629	100.0	20,001	100.0	3375	100.0

Sample Identification	Geochemical Partitioning Phase	Iron mg/kg	Iron %	Manganese mg/kg	Manganese %	Zinc mg/kg	Zinc %	Lead mg/kg	Lead %
Lake Charles, La. influent (2-4-76)	A. Exchangeable	418	1.2	333	41.3	3.36	2.4	2.7	13.1
	B. Carbonate	718	2.1	78.2	9.7	9.13	6.5	†	<0.1
	C. Easily reducible	1,931	5.8	105	13.0	15.0	10.8	†	<0.1
	D. Acid digest (organic-moderately reducible-residual)	30,420	90.9	291	36.0	112.0	80.3	17.9	86.9
	Total	33,487	100.0	807	100.0	139.5	100.0	20.6	100.0
Lake Charles, La. effluent (2-4-76)	A. Exchangeable	579	1.3	377	32.1	3.17	1.4	2.8	7.8
	B. Carbonate	151	0.3	203	17.3	23.5	10.3	†	<0.1
	C. Easily reducible	1,901	4.4	154	13.1	14.9	6.6	†	<0.1
	D. Acid digest (organic-moderately reducible-residual)	40,642	94.0	442	37.5	186	81.7	33.3	92.2
	Total	43,273	100.0	1176	100.0	228	100.0	36.1	100.0

Sample Identification	Geochemical Partitioning Phase	Cadmium mg/kg	Cadmium %	Copper mg/kg	Copper %	Nickel mg/kg	Nickel %	Mercury mg/kg	Mercury %
Lake Charles, La. influent (2-4-76)	A. Exchangeable	0.002	1.7	†	<0.1	2.92	4.4	†	<0.1
	B. Carbonate	0.018	15.0	2.03	7.8	2.93	4.4	†	<0.1
	C. Easily reducible	†	<0.1	3.04	11.7	3.43	5.2	†	<0.1
	D. Acid digest (organic-moderately reducible-residual)	0.10**	83.3	20.8	80.5	56.6	86.0	0.518	100.0
	Total	0.12**	100.0	25.9	100.0	65.9	100.0	0.518	100.0
Lake Charles, La. effluent (2-4-76)	A. Exchangeable	0.003	0.1	†	--	3.04	2.8	†	<0.1
	B. Carbonate	0.05	99.1	†	--	1.96	1.8	†	<0.1
	C. Easily reducible	†	<0.1	1.52	4.4	3.56	3.3	†	<0.1
	D. Acid digest (organic-moderately reducible-residual)	**,††	††	32.8	95.6	100.1	92.1	0.958	100.0
	Total	0.05	100.0	34.3	100.0	108.7	100.0	0.958	100.0

(Continued)

** Value derived indirectly from total acid digest data.
† Below detection limit.
†† Total acid digest data indicate very low levels in this partitioning phase.

Table B19 (Continued)

Sample Identification	Geochemical Partitioning Phase	Calcium		Magnesium		Potassium		Sodium	
		mg/kg	%	mg/kg	%	mg/kg	%	mg/kg	%
Lake Charles, La. influent (2-5-76)	A. Exchangeable	1607	41.9	1,358	21.6	1,290	8.2	--	--
	B. Carbonate	259	6.8	292	4.6	186	1.2	450	--
	C. Easily reducible	756	19.7	302	4.8	66.0	0.4	21.7	--
	D. Acid digest (organic-moderately reducible-residual)	1209	31.6	4,328	69.0	14,105	90.2	--	--
	Total	3831	100.0	6,280	100.0	15,647	100.0	472	--
Lake Charles, La. effluent (2-6-76)	A. Exchangeable	2009	46.1	1,706	15.2	1,394	7.1	3850	89.5
	B. Carbonate	362	8.3	176	1.6	115	0.6	398	9.5
	C. Easily reducible	777	17.8	315	2.8	72.8	0.4	53.2	1.0
	D. Acid digest (organic-moderately reducible-residual)	1206	27.8	9,048	80.4	18,175	91.9	**,††	**,††
	Total	4354	100.0	11,245	100.0	19,757	100.0	4301	100.0

Sample Identification	Geochemical Partitioning Phase	Iron		Manganese		Zinc		Lead	
		mg/kg	%	mg/kg	%	mg/kg	%	mg/kg	%
Lake Charles, La. influent (2-5-76)	A. Exchangeable	55.3	0.2	315	36.0	6.88	5.6	2.0	8.2
	B. Carbonate	58.5	0.2	172	19.7	9.64	7.8	†	<0.1
	C. Easily reducible	1,501	4.2	149	17.0	9.47	7.7	†	<0.1
	D. Acid digest (organic-moderately reducible-residual)	33,933	95.4	239	27.3	97.5	78.9	22.4	91.8
	Total	35,548	100.0	875	100.0	123.5	100.0	24.4	100.0
Lake Charles, La. effluent (2-6-76)	A. Exchangeable	18.9	<0.1	392	17.5	7.7	4.7	†	<0.1
	B. Carbonate	493	1.0	322	14.4	4.82	2.9	†	<0.1
	C. Easily reducible	2,335	4.7	777	34.7	13.6	8.2	8.2	20.6
	D. Acid digest (organic-moderately reducible-residual)	46,643	94.3	751	33.4	139	84.2	31.7	79.4
	Total	49,490	100.0	2242	100.0	165	100.0	39.9	100.0

Sample Identification	Geochemical Partitioning Phase	Cadmium		Copper		Nickel		Mercury	
		mg/kg	%	mg/kg	%	mg/kg	%	mg/kg	%
Lake Charles, La. influent (2-5-76)	A. Exchangeable	0.017	68.0	0.22	0.8	3.53	9.6	†	<0.1
	B. Carbonate	0.008	32.0	2.06	7.5	2.33	6.3	†	<0.1
	C. Easily reducible	†	<0.1	2.22	8.0	2.42	6.6	†	<0.1
	D. Acid digest (organic-moderately reducible-residual)	**,††	††	23.1	83.7	28.4	77.5	0.843	100.0
	Total	0.025	100.0	27.6	100.0	36.7	100.0	0.843	100.0
Lake Charles, La. effluent (2-6-76)	A. Exchangeable	0.063	61.2	0.41	1.2	4.12	8.4	0.023	3.8
	B. Carbonate	0.007	6.8	1.83	5.5	1.43	2.9	†	<0.1
	C. Easily reducible	0.033	32.0	3.68	11.2	3.60	7.3	†	<0.1
	D. Acid digest (organic-moderately reducible-residual)	**,††	††	27.1	82.1	40.0	81.4	0.587	96.2
	Total	0.103	100.0	33.0	100.0	49.2	100.0	0.610	100.0

(Continued)

** Value derived indirectly from total acid digest data.
† Below detection limit.
†† Total acid digest data indicate very low levels in this partitioning phase.

Table B19 (Continued)

Sample Identification	Geochemical Partitioning Phase	Calcium		Magnesium		Potassium		Sodium	
		mg/kg	%	mg/kg	%	mg/kg	%	mg/kg	%
Seattle, Wash. slip 1 sediment (1-16-76)	A. Exchangeable	1,800	5.1	2,500	12.0	1,550	13.2	6,300	24.5
	B. Carbonate	6,800	19.3	780	3.8	205	1.7	380	1.5
	C. Easily reducible	1,700	4.8	130	0.6	94	0.8	48	0.2
	D. Organic-sulfide #1	7,100	19.7	7,600	30.6	275	2.4	510	2.0
	Organic-sulfide #2	6,800	‡	5,100	‡	287	‡	530	‡
	E. Moderately reducible-residual #1	16,000	51.1	11,000	53.0	8,900	81.9	20,000	71.8
	Moderately redicible-residual #2	20,000	‡	11,000	‡	10,400	‡	17,000	‡
	Total	35,250	100.0	20,760	100.0	11,780	100.0	25,750	100.0
Seattle, Wash. influent (3-23-76)	A. Exchangeable	2,800	9.2	3,600	16.9	1,570	13.5	9,000	26.4
	B. Carbonate	5,200	17.2	1,200	5.6	157	1.3	320	0.9
	C. Easily reducible	1,500	5.0	360	1.7	81	0.7	60	0.2
	D. Organic-sulfide #1	2,800	9.2	5,800	26.6	337	2.8	810	2.3
	Organic-sulfide #2	2,800	‡	5,500	‡	316	‡	730	‡
	E. Moderately reducible-residual #1	16,000	59.4	8,900	49.2	8,400	81.7	21,000	70.2
	Moderately reducible-residual #2	20,000	‡	12,000	‡	10,600	‡	27,000	‡
	Total	30,300	100.0	21,260	100.0	11,635	100.0	34,150	100.0
Seattle, Wash. effluent--Pond 2 (4-1-76 to 4-6-76)	A. Exchangeable	2,700	8.4	3,300	14.8	862	6.3	7,600	23.2
	B. Carbonate	2,700	8.4	2,000	8.9	96	0.7	300	0.9
	C. Easily reducible	560	1.7	260	1.2	31	0.2	25	0.1
	D. Organic-sulfide #1	2,500	8.5	2,200	10.3	176	1.5	780	2.4
	Organic-sulfide #2	3,000	‡	2,400	‡	232	‡	790	‡
	E. Moderately reducible-residual #1	21,000	73.0	15,000	64.8	13,000	91.3	28,000	73.4
	Moderately reducible-residual #2	26,000	‡	14,000	‡	12,000	‡	20,000	‡
	Total	32,210	100.0	22,360	100.0	13,695	100.0	32,710	100.0

Sample Identification	Geochemical Partitioning Phase	Iron		Manganese		Zinc		Lead	
		mg/kg	%	mg/kg	%	mg/kg	%	mg/kg	%
Seattle, Wash. slip 1 sediment (1-16-76)	A. Exchangeable	13.4	<0.1	28	4.5	0.7	0.3	0.7	0.6
	B. Carbonate	6,000	13.9	86	13.7	23.0	9.3	1.7	1.5
	C. Easily reducible	1,500	3.5	20	3.2	15.0	6.1	2.3	2.0
	D. Organic-sulfide #1	9,200	22.1	128	20.2	88.0	39.2	42.0	41.6
	Organic-sulfide #2	9,800	‡	125	‡	106	‡	55.0	‡
	E. Moderately reducible-residual #1	34,000	60.5	338	58.4	99.0	45.1	60.0	54.3
	Moderately reducible-residual #2	18,000	‡	396	‡	123	‡	67.0	‡
	Total	43,015	100.0	628	100.0	247	100.0	117	100.0
Seattle, Wash. influent (3-23-76)	A. Exchangeable	483	1.0	12	2.4	0.5	0.1	0.9	0.4
	B. Carbonate	8,100	17.1	70	13.9	14.0	3.3	1.7	0.8
	C. Easily reducible	2,400	5.1	18	3.6	29.0	6.8	1.9	0.9
	D. Organic-sulfide #1	12,000	24.2	138	27.0	238	54.3	149	71.6
	Organic-sulfide #2	11,000	‡	135	‡	228	‡	169	‡
	E. Moderately reducible-residual #1	23,000	52.6	235	53.1	130	35.5	59.0	26.3
	Moderately reducible-residual #2	27,000	‡	301	‡	174	‡	58.0	‡
	Total	47,485	100.0	505	100.0	429	100.0	222	100.0
Seattle, Wash. effluent--Pond 2 (4-1-76 to 4-6-76)	A. Exchangeable	3.3	<0.1	11	1.5	8.7	0.9	0.8	0.3
	B. Carbonate	6,800	9.7	91	12.4	585	61.5	17.0	5.7
	C. Easily reducible	4,000	5.7	133	18.1	105	11.0	25.0	8.4
	D. Organic-sulfide #1	15,000	25.1	54	7.8	129	14.8	83.0	34.5
	Organic-sulfide #2	20,000	‡	61	‡	152	‡	122	‡
	E. Moderately reducible-residual #1	41,000	59.5	424	60.2	108	11.8	149	51.1
	Moderately reducible-residual #2	42,000	‡	457	‡	116	‡	155	‡
	Total	69,805	100.0	733	100.0	951.0	100.0	297.0	100.0

(Continued)

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‡ Percent value is an average of the two treatments.

Table B19 (Concluded)

Sample Identification	Geochemical Partitioning Phase	Cadmium mg/kg	Cadmium %	Copper mg/kg	Copper %	Nickel mg/kg	Nickel %	Mercury mg/kg	Mercury %
Seattle, Wash. slip 1 sediment (1-16-76)	A. Exchangeable	<0.02	<1.0	0.3	0.3	0.8	1.0	--	--
	B. Carbonate	0.07	6.3	0.3	0.3	4.1	5.1	--	--
	C. Easily reducible	<0.07	<6.0	0.2	0.2	1.5	1.9	--	--
	D. Organic-sulfide #1	0.78	93.7	51.0	55.4	16.0	21.1	--	--
	Organic-sulfide #2	1.32	‡	72.0	‡	18.0	‡	--	--
	E. Moderately reducible-residual #1	<0.5	<30.0	48.0	43.8	56.0	70.9	--	--
	Moderately reducible-residual #2	<0.5	‡	49.0	‡	58.0	‡	--	--
	Total	1.12	100.0	111	100.0	80.5	100.0	0.29+	100
Seattle, Wash. influent (3-23-76)	A. Exchangeable	<0.02	<0.5	0.2	0.2	1.2	1.3	--	--
	B. Carbonate	<0.09	<2.5	0.4	0.3	1.7	8.3	--	--
	C. Easily reducible	<0.13	<4.0	0.5	0.4	4.4	4.7	--	--
	D. Organic-sulfide #1	3.01	100.0	96.0	79.2	32.0	29.6	--	--
	Organic-sulfide #2	3.38	‡	102.0	‡	23.0	‡	--	--
	E. Moderately reducible-residual #1	<0.5	<18.0	26.0	19.9	46.0	56.1	--	--
	Moderately reducible-residual #2	<0.5	‡	23.0	‡	58.0	‡	--	--
	Total	3.2	100.0	125	100.0	93.0	100.0	0.68+	100
Seattle, Wash. effluent--Pond 2 (4-1-76 to 4-6-76)	A. Exchangeable	0.05	0.5	0.6	0.4	6.0	4.8	--	--
	B. Carbonate	5.5	58.5	26.0	17.7	15.0	11.9	--	--
	C. Easily reducible	2.1	22.3	20.0	13.6	4.1	3.3	--	--
	D. Organic-sulfide #1	1.4	18.7	54.0	41.2	14.0	11.5	--	--
	Organic-sulfide #2	2.1	‡	67.0	‡	15.0	‡	--	--
	E. Moderately reducible-residual #1	<0.5	<5.0	42.0	27.1	83.0	68.5	--	--
	Moderately reducible-residual #2	<0.5	‡	37.0	‡	89.0	‡	--	--
	Total	9.4	100.0	147	100.0	126	100.0	1.02+	100

Sample Identification	Geochemical Partitioning Phase	Chromium mg/kg	Chromium %	Arsenic mg/kg	Arsenic %
Seattle, Wash. slip 1 sediment (1-16-76)	A. Exchangeable	0.10	0.2	0.15	1.7
	B. Carbonate	1.4	2.2	<0.15	<1.0
	C. Easily reducible	1.0	1.6	<0.15	<1.0
	D. Organic-sulfide #1	18.0	28.0	0.78	30.3
	Organic-sulfide #2	17.0	‡	4.5	‡
	E. Moderately reducible-residual #1	42.0	68.0	7.1	68.0
	Moderately reducible-residual #2	43.0	‡	4.8	‡
	Total	62.5	100.0	8.75	100.0
Seattle, Wash. influent (3-23-76)	A. Exchangeable	<0.05	<0.1	0.13	1.2
	B. Carbonate	3.2	3.3	<0.15	<1.0
	C. Easily reducible	4.4	4.5	<0.15	<1.0
	D. Organic-sulfide #1	51.0	46.4	<0.5	<4.0
	Organic-sulfide #2	40.0	‡	<0.5	‡
	E. Moderately reducible-residual #1	43.0	45.8	12.0	98.8
	Moderately reducible-residual #2	47.0	‡	10.0	‡
	Total	98.1	100.0	11.15	100.0
Seattle, Wash. effluent--Pond 2 (4-1-76 to 4-6-76)	A. Exchangeable	0.2	0.1	7.1	14.7
	B. Carbonate	2.5	1.3	<0.15	<0.3
	C Easily reducible	1.0	0.5	<0.15	<0.3
	D. Organic-sulfide #1	81.0	43.6	0.69	29.6
	Organic-sulfide #2	88.0	‡	28.0	‡
	E. Moderately reducible-residual #1	95.0	54.5	33.0	55.7
	Moderately reducible-residual #2	116	‡	21.0	‡
	Total	194	100.0	48.5	100.0

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Hoeppel, Ronald E

Physical and chemical characterization of dredged material influents and effluents in confined land disposal areas / by Ronald E. Hoeppel, Tommy E. Myers, Robert M. Engler. Vicksburg, Miss. : U. S. Waterways Experiment Station ; Springfield, Va. : available from National Technical Information Service, 1978.

195, [117] p. : ill. ; 27 cm. (Technical report - U. S. Army Engineer Waterways Experiment Station ; D-78-24)

Prepared for Office, Chief of Engineers, U. S. Army, Washington, D. C., under DMRP Work Unit No. 2D01.

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