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Strait of Canso, Nova Scotia, Canada

Environment Canada Atlantic Region Disposal at Sea Monitoring Program Report (2004/2005)

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ABSTRACT

The Canso common user dredged material disposal site monitoring program was conducted in 2004 to assess the impacts of disposal at the site and to investigate the concerns of elevated PCBs reported by past studies in sediments collected from the disposal site and the surrounding areas.

Multibeam and geophysical surveys conducted at the disposal site showed dredged materials characterized by rough surfaces appeared near the center of the disposal site in the deepest water depths. An accumulation of about 1 meter of recent sediments on the seafloor of the disposal site was detected in the sub-bottom profiler image. Seafloor photographs at the disposal site showed a seafloor covered with fine-grained sediment overlying coarser material in places.

Sediment samples collected at the disposal and reference sites were high in silt and clay content. The mean concentrations of heavy metals in all sediment samples were lower than the probable effect levels (PEL) of the Canadian Council of Ministers of the Environment (CCME) Interim Marine Sediment Quality Guideline. Two of the metals, Zn and Hg, were lower than the threshold effect levels (TEL). Cd concentrations exceeding the Disposal at Sea (DAS) regulated limit were only detected in sampling stations outside of the disposal site near the eastern shore of the Strait with a high density of industrial sites. Mean concentrations of Cu, Pb, Zn and Cd in the disposal site sediment samples from the present study were lower than those reported in the 1983 pre-dump survey and the 1985 and 1987 monitoring programs.

PAHs in one of the disposal site sediment samples and several samples outside of the disposal site exceeded the DAS regulated limit. However, no statistically significant difference of PAH concentrations was detected in samples collected at any of the studied areas. PCBs in the disposal site sediments were the lowest among all the samples. The highest concentration of PCBs, many times higher than the DAS regulated limit, was found outside of the disposal site near the shoreline closest to the industrial areas within the Strait of Canso. Concentration of PCBs at the disposal site did not change significantly since 1985.

None of the sediments was toxic to the test organisms in the amphipod and Microtox® (Microbics Inc, Carlsbad, CA) solid phase tests. Two sediments from the disposal sites were toxic as measured by the polychaete survival and growth test while all except one sample were toxic according to the sea urchin fertilization test criteria. However, no statistically significant differences in sediment toxicity were detected between the disposal and reference sites. For all the toxic endpoints observed in this study, none of them were statistically correlated to sediment contaminants. While the polychaete test results may be confounded by the effects of particle size, ammonia and sulfide, it is not known what caused the toxicity to the sea urchin.

Benthic communities at the disposal and reference sites were dominated in terms of numbers by polychaete worms. The dominant polychaete species were quite similar to those found in the 1987 monitoring study, with the exception that several species of polychaetes found in the 1987 study were absent at both the disposal and reference sites in the present study. No significant differences of all univariate measures, such as the species abundance, number of species and biomass, were detected between the disposal and reference sites. Cluster analysis showed the communities were very similar between the two sites while ABC curves showed the abundance and biomass of the communities living at these two sites were similar to communities living in an "undisturbed" environment.

The dominant polychaete species found at the disposal and reference sites were second stage colonizers showing that the communities may still remain in the second stage of recolonization process since 1999. Because of the similarities of species composition, diversity and other community measures between the disposal and reference sites, it is also possible that the community at the disposal site may have already fully recovered from the last disposal related impacts. The domination of second stage colonizers in these communities may be caused by the unfavorable condition of the local marine environment created by the construction of the Canso Causeway in 1954.

The results of this study showed that the Canso disposal site still has the capacity to receive more than 300,000 cu.m. of dredged materials before significant water depth changes would occur.

The site is located in a very stable environment within a low current regime with colder temperature and no water stratification. Recovery of the community at the site may be slow, but it has demonstrated that it can sustain and recover from the impacts of multiple disposals of dredged materials. Use of the site for future dredging projects within the Canso Strait is recommended.

It is also recommended that, because of the detection of hot spots in the Strait sediments, especially along the eastern shoreline, special attention should be paid to sediment chemical characterization of any dredging projects within the Strait to ensure that potential risk of sediment contamination can be minimized when dredging and disposing of dredged materials at the disposal site.

RÉSUMÉ

Le programme de surveillance du site d'immersion en mer à usage partagé pour déblais de dragage du détroit de Canso a été mis en œuvre en 2004 afin d'évaluer les impacts de l'immersion des déblais au site en question ainsi que pour faire la lumière au sujet des fortes concentrations de biphényles polychlorés (BPC) rapportées dans des études antérieures portant sur des échantillons de sédiments prélevés au site d'immersion et dans les zones environnantes.

Des levés multifaisceaux et géophysiques effectués au site d'immersion ont montré qu'il y avait des déblais de dragage caractérisés par des surfaces rugueuses près du centre du site, où la profondeur de l'eau est maximale. Le sondeur de sédiments a permis de détecter l'accumulation d'environ 1 mètre de sédiments récents au site d'immersion. Des photographies du fond marin prises au site d'immersion ont montré la présence de sédiments fins recouvrant par endroits des matériaux plus grossiers.

Des échantillons de sédiments recueillis au site d'immersion et au site de référence ont révélé la présence de fortes proportions de limon et d'argile. Dans tous les échantillons de sédiments, les concentrations moyennes de métaux lourds étaient inférieures à la concentration produisant un effet probable (CEP) indiquée dans la Recommandation provisoire sur la qualité des sédiments du Conseil canadien des ministres de l'environnement (CCME). Dans le cas de deux métaux, le zinc et le mercure, les concentrations étaient inférieures aux concentrations seuil produisant un effet (CSE). Des concentrations de cadmium supérieure à la limite établie pour l'immersion en mer ont été détectées uniquement aux stations d'échantillonnage situées en dehors du site d'immersion, près de la côte est du détroit, où il y a une forte densité de sites industriels. Les concentrations moyennes de Cu, Pb, Zn et Cd dans les échantillons de sédiments de site d'immersion utilisés dans la présente étude étaient inférieures à celles rapportées dans l'étude de 1983 précédant l'immersion et dans les programmes de surveillance de 1985 et 1987.

Dans un des échantillons de sédiments prélevé au site d'immersion ainsi que dans plusieurs des échantillons prélevés en dehors de ce site, les concentrations d'hydrocarbures poly-aromatiques (HPA) excédaient la limite établie pour l'immersion en mer. Toutefois, aucune différence

statistiquement significative entre les concentrations de HPA n'a été détectée dans les échantillons recueillis dans toutes les zones étudiées. Dans les sédiments du site d'immersion, les concentrations de BPC étaient les plus faibles parmi tous échantillons. La plus forte concentration de BPC, de beaucoup supérieure à la limite établie pour l'immersion en mer, a été détectée à l'extérieur du site d'immersion, près de la région côtière du détroit de Canso où se situent les zones industrielles. La concentration de BPC au site d'immersion n'a pas changé de façon significative depuis 1985.

Aucun des échantillons de sédiments n'était toxique pour les organismes utilisés pour les essais de toxicité sur les amphipodes et pour les essais Microtox® (Microbics Inc, Carlsbad, CA) en phase solide. Deux échantillons de sédiments des sites d'immersion étaient toxiques tel que measuré par essais de survie et de croissance des vers polychètes, tandis que tous les échantillons sauf un étaient toxiques selons les critères de les essais de fécondité des oursins. Toutefois, aucune différence statistiquement significative, en ce qui a trait à la toxicité des sédiments, n'a été décelée entre le site d'immersion et le site de référence. Aucun des effets toxiques observés dans la présente étude n'était statistiquement corrélé avec des contaminants sédimentaires. Bien qu'il puisse y avoir de la confusion associée aux résultats des essais sur les vers polychètes, attribuable aux effets de la taille des particules et à la présence d'ammoniac et de sulfures, on ignore quelle était la cause de la toxicité dans le cas des oursins.

Tant au site d'immersion qu'au site de référence, les communautés benthiques étaient dominées par les vers polychètes en ce qui a trait au nombre. Les espèces de vers polychètes dominantes étaient très semblables à celles observées dans l'étude de surveillance de 1987, sauf que plusieurs de ces espèces absentes dans la présente étude, tant au site d'immersion qu'au site de référence. On n'a décelé aucune différence significative, dans le cas de toutes les mesures unidimensionnelles, comme l'abondance des espèces et la biomasse, entre le site d'immersion et le site de référence. L'analyse typologique a montré qu'il y avait une grande similitude entre les communautés observées aux deux sites, tandis que les courbes de comparaison abondance-biomasse ont révélé que l'abondance et la biomasse des communautés vivant dans ces sites étaient semblables à celles des communautés vivant dans un environnement « intact ».

Les espèces dominantes de vers polychètes observées au site d'immersion et au site de référence étaient des colonisateurs de deuxième stade, ce qui montre qu'il peut y avoir encore des communautés au deuxième stade du processus de recolonisation depuis 1999. En raison des similitudes au niveau de la composition et de la diversité des espèces, et au niveau d'autres mesures relatives aux communautés benthiques, il est également possible que la communauté au site d'immersion se soit entièrement remise des impacts connexes de la dernière immersion de déblais. La prédominance de colonisateurs de deuxième stade au sein de ces communautés pourrait être attribuable aux conditions défavorables engendrées par la construction de la Chaussée de Canso en 1954.

Les résultats de la présente étude montrent que le site d'immersion de Canso a encore la capacité de recevoir plus de 300 000 m^3 de déblais de dragage avant qu'il n'y ait des changements importants au niveau de la profondeur de l'eau. Le site se trouve dans un environnement très stable, dans un régime de courant faible caractérisé par une température froide et l'absence de stratification de l'eau. La récupération de la communauté benthique au site d'immersion est peut-être lente, mais il a été montré que cette communauté est capable de subir les impacts causés par des immersions multiples de déblais de dragage et de s'en remettre. L'utilisation du site pour de futurs projets de dragage dans le détroit de Canso est recommandée.

En raison de la détection de zones plus polluées dans les sédiments, notamment le long de la côte est, on recommande également d'accorder une attention particulière à la détermination des caractéristiques chimiques des sédiments dans le cas de tout projet de dragage prévu dans le détroit de Canso, afin de pouvoir réduire le plus possible le risque de contamination lié au dragage et à l'immersion de déblais de dragage au site d'immersion.

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ACKNOWLEDGEMENTS

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The sections related to geophysical monitoring in this report were compiled by the authors based on scientific papers and reports written by Geological Survey of Canada. The sections on the bioassay tests were based on a report submitted by the Environment Canada Moncton Toxicological Laboratory and some sections of the benthic community study were compiled using two reports submitted by Envirosphere Consultants Ltd.

INTRODUCTION

In 1983, a common user ocean disposal site (45o35.65'N; 61o22.67'W) was designated by the Atlantic Regional Ocean Dumping Advisory Committee (RODAC) for the disposal of uncontaminated dredged materials in the Strait of Canso, Nova Scotia (Figure 1). The site designation was based on the results of a disposal site selection survey carried out on behalf of Environment Canada (EC) by Martec Ltd., Halifax, Nova Scotia (Martec, 1984), which concluded that the designated site has the least benthic activity and the highest concentration of sediment contaminants of the four sites selected for the survey. The disposal site is a 650 metres long and 200 metres wide trough with water depth ranging from 55 to 64 metres. It was determined that it has a capacity for holding 500,000 to 600,000 cu.m. of dredged materials before significant water depth changes would occur.

Between 1983 and 1984, approximately 177,000 cu. m. of dredged materials were disposed of at the site, most of these materials (137,000 cu. m.) came from Mulgrave Harbour, while the remainder came from the Port Hawkesbury Public Wharf. Sediment polychlorinated biphenyls (PCBs) concentrations in two grab samples collected from Mulgrave Harbour prior to dredging exceeded the EC Disposal at Sea (DAS) regulatory limit of 100 μg/kg while one cadmium value (Cd DAS limit is 0.6 mg/kg) and one PCB value were higher than the DAS regulatory limits in the pre-dredged sediments of Port Hawkesbury Public Wharf (OceanChem, 1987).

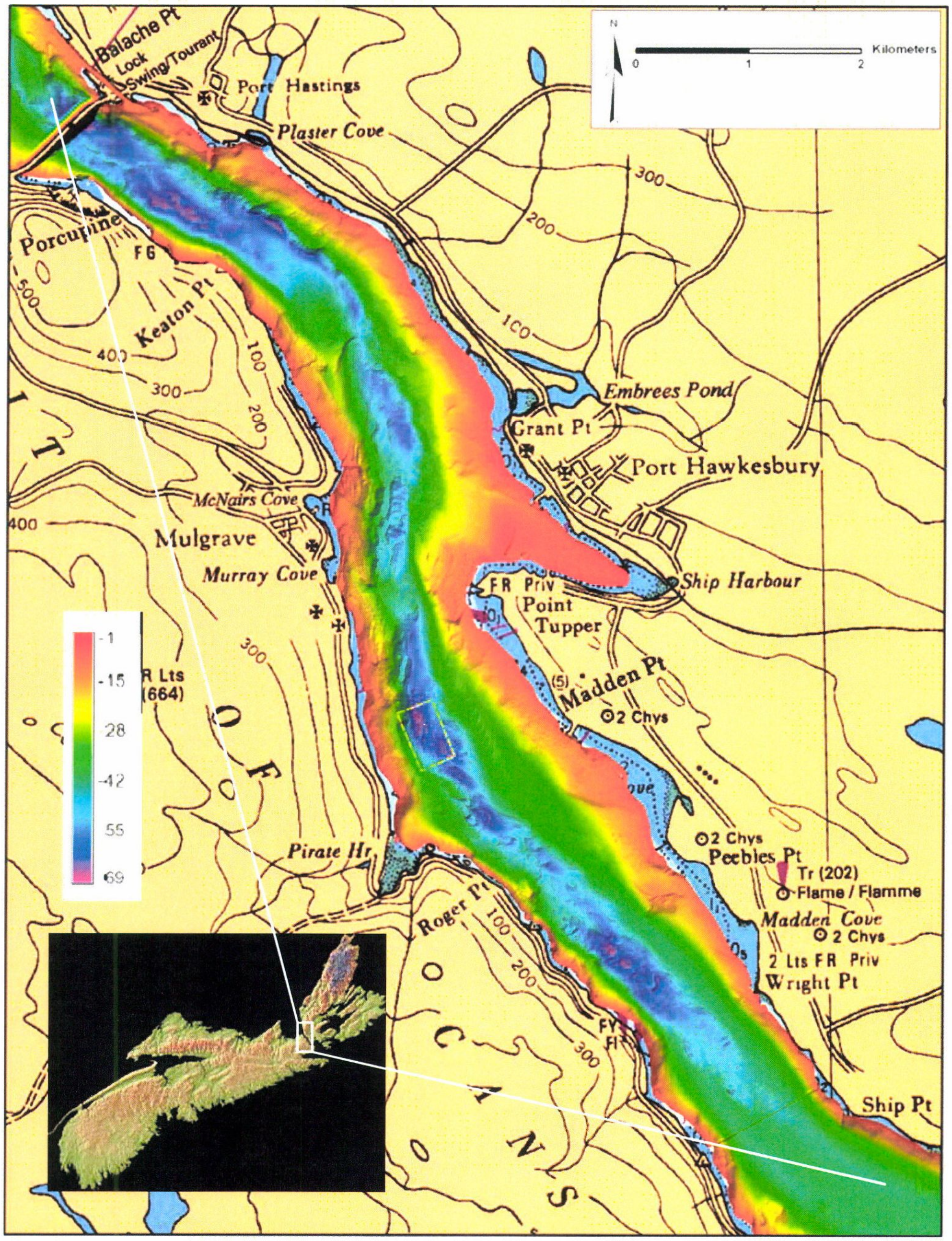


Figure 1. Colour shaded-relief image created from multibeam bathymetry data collected in the Strait of Canso in 2004. The location of the designated disposal site is shown by the yellow rectangle. The location of the survey area is shown on the inset map of Nova Scotia. The colour bar used for the bathymetry data is shown on the left side of the image.

On July 16 – 19, 1985, a field program was conducted by EC to assess the disposal site (Tay, 1987). A remote control underwater submersible, the "DART", was used with the assistance of the DND personnel to obtain a visual record of bottom characteristics of the site. Sediment samples and benthic infauna samples were collected from the disposal site from 13 sampling stations, including two reference stations located at the northeast end outside of the disposal site. The results of this survey showed that sediment particle size at the disposal site had shifted from fine silt and clay toward the coarse particles of sand and silt after the disposal operations. Concentrations of heavy metals in the disposal site sediment samples were higher in the post-dump samples (e.g. Cd – 0.65 ± 0.16 mg/kg, n = 24) than those collected in the pre-dump survey (e.g. Cd – 0.45 ± 0.17 mg/kg, n = 4). Elevated sediment PCB concentrations (104 to 1,015 ug/kg) were detected in samples collected from eight of the thirteen post-dump sampling stations. Polychaete and Nemertinea were the most abundant groups both before and after dumping.

On July 20 – 30, 1987, EC conducted another survey at the site to investigate the unusually high concentrations of Cd and PCBs detected in the 1985 survey and to focus more on the disposal impacts on the benthic community at the site. Sediment samples were collected at the disposal site from 17 sampling stations corresponded to stations sampled in the 1983 pre-dumping and 1985 post-dumping surveys (Tay, 1987). Seventeen additional sampling stations were randomly selected outside of the disposal site to investigate the spatial distribution of Cd and PCBs due to the concerns that some of the discharges from industrial sites along the eastern shoreline of the Strait may have been responsible for the elevated concentrations of Cd and PCBs at the disposal site in the 1985 survey.

Before the sediment collection, a geophysical survey was conducted by McElhanney Services Ltd. to obtain bathymetric and sub-bottom profile data at the disposal site. Geophysical data were collected by an O.R.E. sub-bottom profiler and a Ross Model 801 Echo Sounder over a rectangular survey grid comprised of 34 NNW – SSE oriented lines, spaced approximately 10 meters apart, and 12 EWE – WSW oriented lines with approximately 50 meters apart. Positioning and navigation were provided by a Del Norte Trisponder System integrated to McElhannery's computer navigation package "NAVPAK". The site specific bathymetry indicated that depth over the bottom of the disposal site varied between 45 m to 65 m. The bottom of the disposal site forms a depression in the center of the site with a ridge running north to south throughout the site. The sub-bottom profiling showed a homogenous sediment composition throughout the area. The exception to this is a surficial veneer (0 -2 m) of seismic reflections that is too inconsistent to be mapped properly using the available equipment (McElhanney, 1987).

Sediment particle size at the disposal site was, in general, quite similar between the samples collected from the 1985 and 1987 surveys. The mean concentration of Cd in the 1987 survey was below the DAS regulated limit while the mean PCBs was higher than the pre-dump survey and similar to those reported in 1985. High concentrations of PCBs (as high as 2,630 μg/kg) were detected in 15 samples collected from the 17 sampling station outside of the disposal site showing that sediment PCBs in the strait has a wider distribution and not just restricted within the disposal site. The highest concentrations of PCBs (1,395 - 2,630 μg/kg) were detected in the samples collected along the eastern shoreline of the strait in front of the pulp and paper mill and the Nova Scotia Power generating station. Comparison of the GC traces of PCBs measurements obtained from this survey with those obtained from a potential dredged site located at the Nova Scotia Power Corporation which is about 2 nM (3,800 metres) northeast of the disposal site indicated that the PCBs could have originated from the same source (OceanChem, 1987).

Both the number of species and individuals of polychaetes increased from 1985 to 1987, indicating that, although the original fauna was partially disturbed at the disposal site, rapid recolonization by a wider variety of organisms had taken place. *Prionospio steenstrupi*, a second stage recolonizer was the dominant species both in 1985 (44%) and 1987 (36%). Another spionid, *Spiophanes kroyeri*, also occurred regularly and in high numbers. Other second stage recolonizers collected in 1987 were members of the polychaete genera, *Lumbrineris, Eteone, Schistomeringos, Pholoe, and Nereis*. The increased number of species and individuals, especially the second stage recolonizers, indicated that the infaunal community at the disposal site had undergone a recolonization process after the initial disposal impact.

Results of cluster analyses conducted on the 1985 and 1987 infaunal data showed that there is a distinct group of infauna located at the center and south end of the disposal site where sediment samples are mostly sand and gravel. This related to the area where most dumping took place. The cluster patterns are also consistent with the depth contour of the 1987 sounding survey at the site.

In 1999, the disposal site received an additional 39,529 cu.m. of dredged materials from the dredging of the Georgia Pacific Terminal Wharf in the Strait. The materials which consisted mainly of gravel (73%) and sand (26%) were generally quite clean with the exception of two samples which contained 107 μg/kg and 122 μg/kg of PCBs respectively (mean PCB concentration of the 8 samples collected was only 52.81 ± 13.15 μg/kg). Before disposal, Georgia Pacific Inc. found high levels of PCBs (145.67 ± 25.1 μg/kg) in three sediment samples collected from the disposal site. After the disposal, the company reported that the level of PCBs at the disposal site had reduced significantly to 38.69 ± 10.11 μg/kg. Sediment particle size analysis showed an increase of coarser material (49% mean gravel/sand) in the disposal site sediment when comparing with the pre-disposal samples (16% mean gravel/sand).

In 2004, the disposal site was selected by EC Atlantic Regional Monitoring Program to assess the concerns of the repeated occurrence of elevated PCB at the disposal site and its surrounding areas and to monitor the impact of the 1999 disposal on the benthic communities at the disposal site which were shown to be in recovery in the EC 1987 study (Tay, 1987). Advanced physical oceanographic equipment including multibeam, side scan and sub-bottom profilers were used to delineate the disposal site and to collect geophysical and geochemical information of the site and the surrounding areas within the Strait of Canso. Following the physical and geological survey, the geophysical data was used to design sampling program to study the sediment contaminants, toxicity and the health of the benthic community at the disposal site and its surrounding areas. This report will summarize and assess the results of the 2004 monitoring program.

OBJECTIVES

The Canso disposal site is selected for the 2004/05 Atlantic Regional disposal site monitoring program because EC's records indicated that elevated PCB had been repeatedly detected in the sediments of some of the dredged sites which had used the Canso disposal site for disposal since 1983. In addition, elevated PCB was also detected in the disposal site sediments in several past disposal site surveys conducted by EC. Thus, the site meets the first trigger of the Environment Canada Disposal Site Monitoring Guidelines "*A permit for the disposal of dredged material was issued under the Rapidly Rendered Harmless provisions of CEPA*" (Chevrier and Topping, 1998).

The monitoring plan was designed to address the following impact hypotheses:

- *The disposal activities did not significantly increase sediment contaminant levels and their toxicity at the disposal site;*
- *The benthic communities at the disposal site have fully recovered or in the processing of recovering after the initial impact of the disposal activities.*

To achieve the objectives and to address the impact hypotheses, a reference site with similar physical background within the Strait was selected for the study and the following endpoints were employed to compare pre and post disposal data and data collected from the past monitoring surveys at the disposal site:

1. Sediment contaminants;
2. Sediment toxicity;
3. Benthic communities.

MATERIALS AND METHODS

Physical Monitoring

From April 27 to May 14, 2004, physical monitoring including multibeam bathymetry, sidescan sonar and sub-bottom profiling was conducted by the Geological Survey of Canada (GSC) on a 20 m inshore research vessel, *J.L.Hart*, operated by the Canadian Coast Guard, in the Strait of Canso, with special focus on the areas in and near the disposal site.

Multibeam bathymetry

Multibeam bathymetry data were collected using a Simrad EM3000 multibeam bathymetry system mounted in the hydrographic survey launch Plover. The EM3000 system provides a depth resolution of 1 cm with an accuracy of 5 cm RMS. Each of the beams insonify an area of approximately 1.35 m^2 at 50 meters water depth. Survey lines were run throughout the area to provide 200 percent coverage of the seafloor. The processed data were used to generate shaded-colour relief images which were overlain on bathymetry charts of the area as shown in Figures 2 and 3 (Parrott *et al.*, 2005a,b).

Geophysical survey

Geophysical equipment employed in this study included a Simrad MS992 dual frequency (120 and 330 kHz) sidescan sonar system, and an IKB Seistec® sub-bottom profiler (Parrott *et al.*, 2005b). Sidescan sonar and sub-bottom profiler data were collected on a 75 meter grid at the disposal site and the areas close to Port Hawkesbury. Sidescan sonar data were processed and combined into a mosaic to provide an overview of the character of the seafloor sediments. These data also provides additional information on the distribution of sediments and the fine-scale features on the seabed, such as dredged spoils and anchor furrows, with a higher resolution than that obtained with multibeam bathymetry data. The two data sets complement each other. Sub-bottom profiler data provide information on the thickness of surficial sediments and insight into the nature and genesis of sediments below the seafloor.

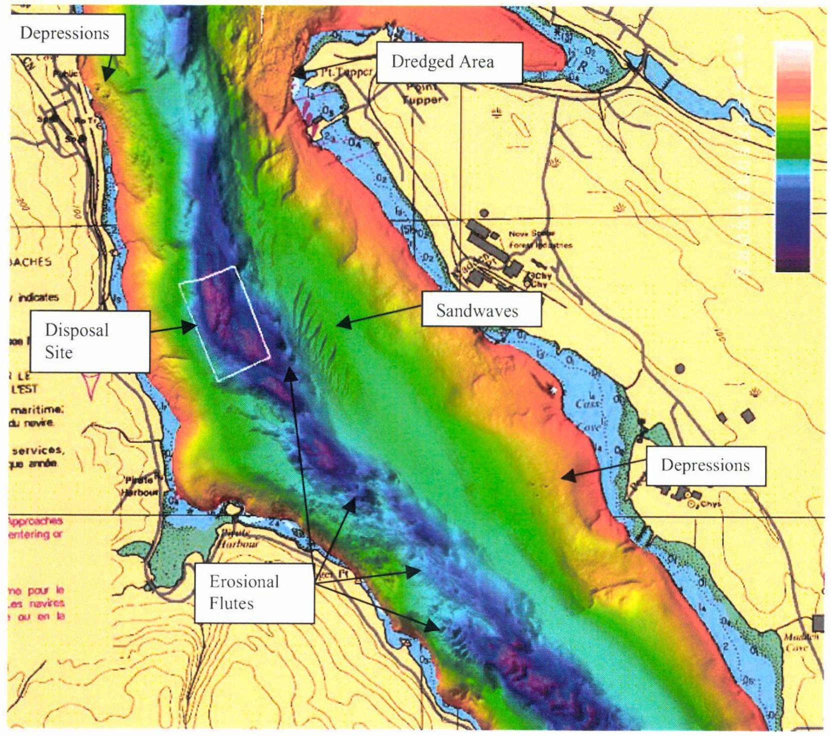


Figure 2. Multibeam bathymetry of the area adjacent to the Strait of Canso disposal site.

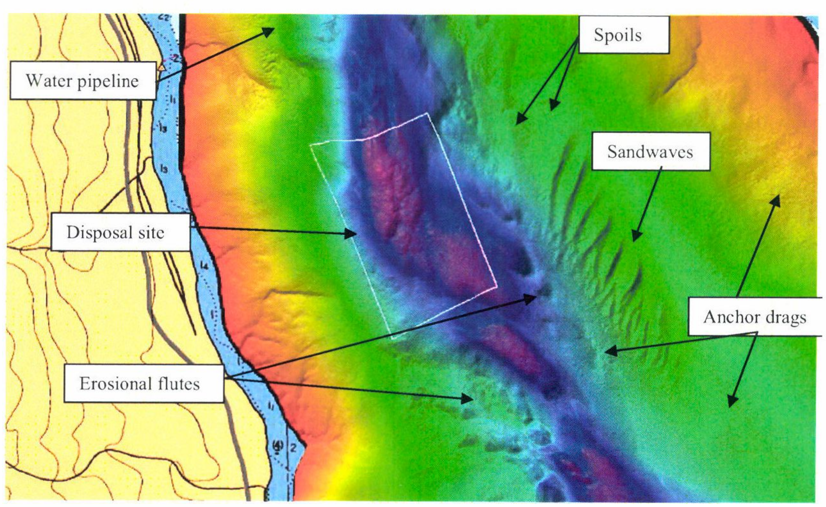


Figure 3. Multibeam bathymetry data showing the disposal site and some seafloor bottom features outside of the disposal site.

Sediment samples were collected to assist with interpretation of the geophysical and multibeam bathymetry data and to assist in the study of biological communities and sediment composition (Envirosphere Consultants Ltd., 2004a; Parrott *et al.*, 2005b). Historical sample data were recovered from the Geological Survey of Canada on-line database. The samples provide information on the nature of sediments throughout the Strait with emphasis on the area around the disposal site. All sediment samples were analyzed for grain size; many were also analyzed for carbon content and metal concentrations for geochemical purpose by the Geological Survey of Canada.

Photo images of the seafloor were obtained with a variety of systems including the Fisheries and Oceans (DFO) TOWCAM system (Gordon *et al.*, 2004); a 35 mm film camera; and an underwater video and digital still camera system (Figures 4). Physical habitat characteristics and the type and relative abundance of benthic fauna were interpreted from the photographs. Features such as counts of animals and burrow were classified. Physical habitat was described in terms of relative abundance of boulders, cobbles, pebbles, granules, sand and silt, large and small burrows, and siphons of infaunal invertebrates. All visible species of megabenthos were identified to the highest possible taxonomic resolution. Average abundance ranks and habitat characteristics were calculated for each seafloor photograph. The results of the physical surveys, seafloor photo images and the physical and biological community interpretation of the photos were summarized in two internal reports prepared by the Geological Survey of Canada (Parrott *et al.*, 2005b; Kostylev *et al.*, 2005).

Chemical and Biological Monitoring

Sediment Sampling

The sampling program was carried out from July 27 to 29, 2004 by staff of Envirosphere Consultants Limited on the fishing vessel *High Voltage,* operated by Al Richard of the Port Hawkesbury area (Envirosphere Consultants Ltd., 2004 a, b). Sediment samples were collected at six locations within the disposal site, and six respectively in the immediate vicinity of the disposal site and at a reference site for sediment contaminants, toxicity tests and benthic community studies (Figure 5). The locations of the sampling stations within the disposal site and the immediate vicinity were selected to match sampling stations used for the 1985 and 1987 studies. Additional samples were collected from fifteen stations on five transects of the Strait of

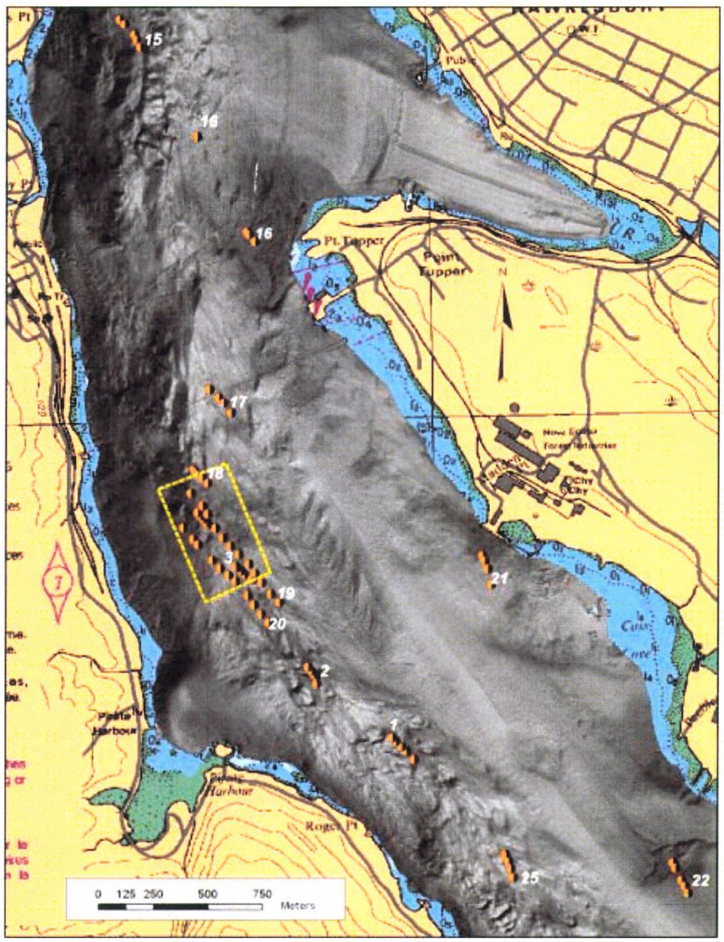


Figure 4a. Sea floor photograph stations near the disposal site overlaid on an image of acoustic backscatter. The disposal site is outlined by the yellow rectangle.

Canso from Ship Harbour to near the Nova Scotia Power Inc. thermal generating plant for sediment contaminants. Sediments collected from ten of the fifteen stations were used for toxicity tests while two locations where high concentrations of PCBs were detected in 1987 (OceanChem, 1987; Tay, 1987) were selected for benthic community studies. For data comparison and discussion purposes, these fifteen stations will be called "the outside area" in this report. All sampling stations were located by DGPS, NAD83 (Table 1).

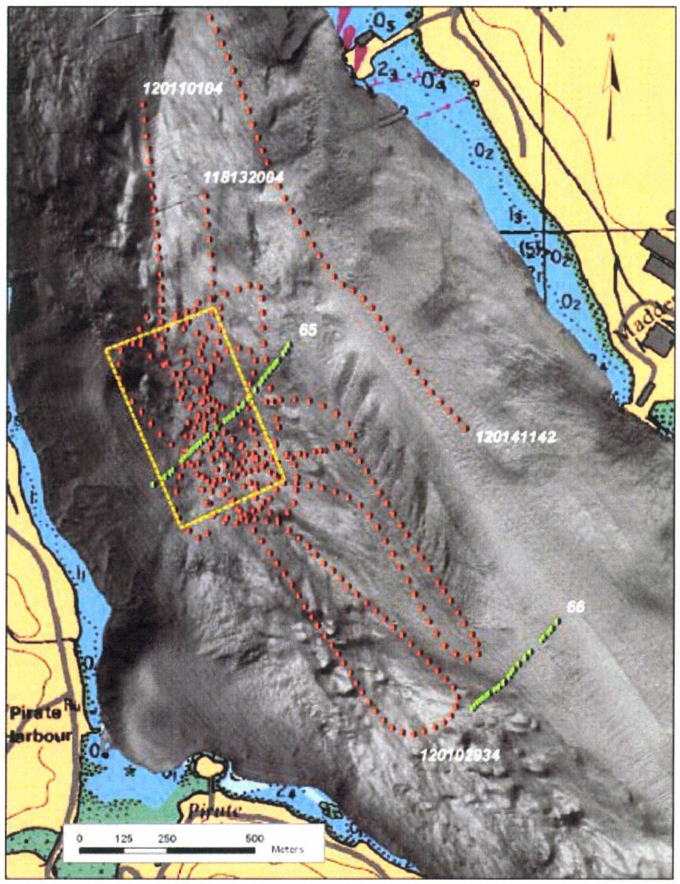


Figure 4b. TOWCAM and Scorpio video transects near the disposal site overlaid on an image of acoustic backscatter. Individual still image locations from the TOWCAM transects are shown by the red dots, and from the Scoprio system by the green dots. The disposal site is outlined by the yellow rectangle.

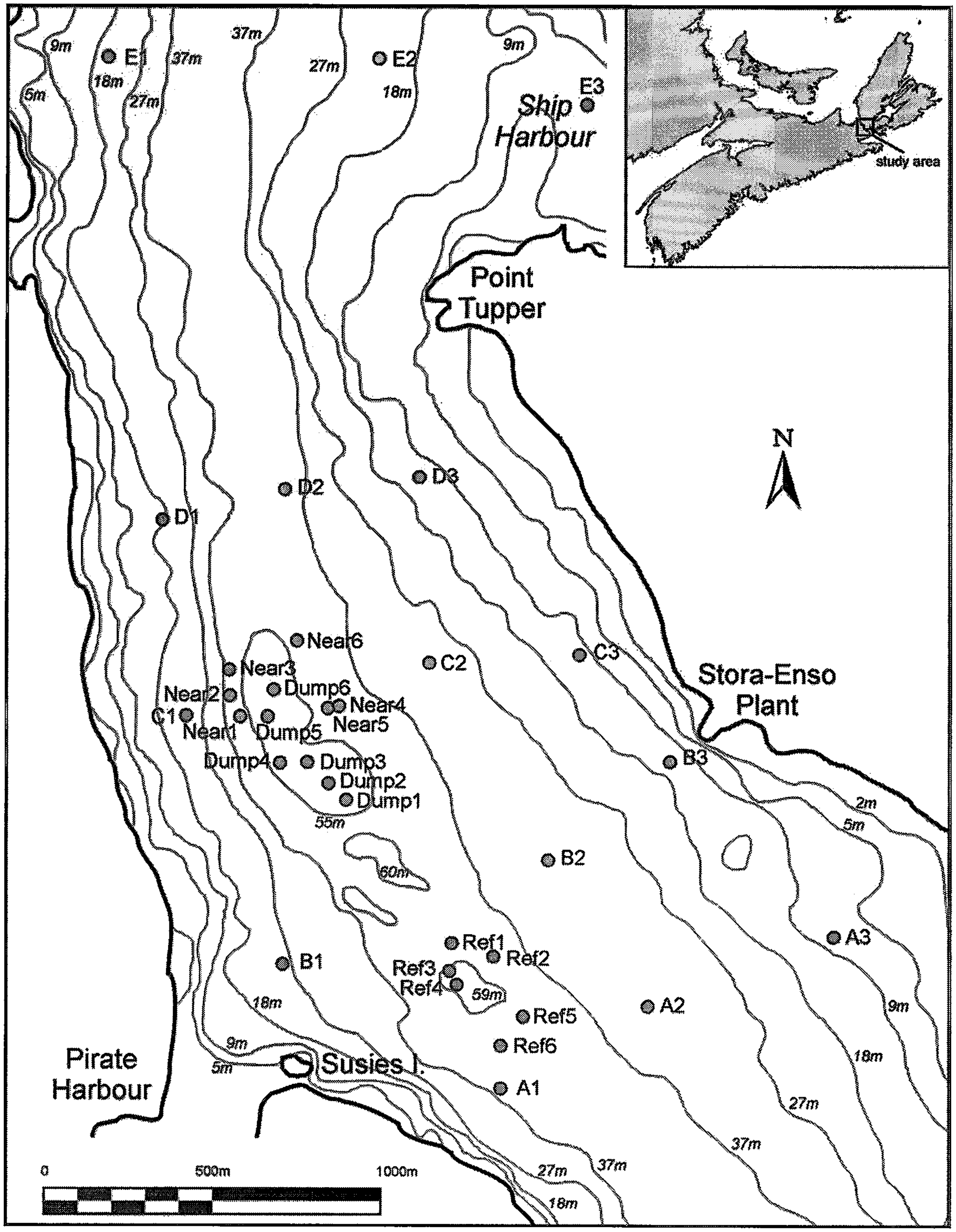


Figure 5. 2004 sampling stations.

A Van Veen grab (0.2 m^2) was used to collect sediment samples. Samples for contaminants (organics and metals), grain size, and total organic carbon were removed from the upper 5-10 cm of the sediment using appropriate precautions to avoid contamination (e.g. plastic utensils for metals; metal utensils for organic contaminants); and approximately 4-5 L transferred to clean, polyethylene buckets for toxicity testing. Samples for organic contaminants were placed in solvent-rinsed mason jars, while the remaining physical/chemical samples were placed in plastic whirlpak bags. At Reference Site 3, there was a layer of black, highly organic sediment below about 10 cm. This was sampled separately for PAH/PCB analysis (serial number 224448 and Station 'Ref3X'). Samples for animal community study were taken from each of typically two grabs taken at the site, using a square corer, 0.05 m^2 in area (Figure 6). A second grab sample was then taken to complete the required volume of 4-5 L for the toxicity testing, and to complete the required set of three samples for animal community from each station. The approach repeated as closely as possible to that used in the 1987 sampling (OceanChem, 1987). Physical/chemical samples were placed on ice, and sediments for animal communities were transferred to buckets. Sieving for animal communities were carried out on board using a 0.5 mm sieve and the animals and any residual sediment retained on the sieve were placed in mason jars and preserved in 10% formalin buffered with borax.

Sediment samples for particle size and chemical analyses were sent to Bedford Oceanographic Institute in Dartmouth, Nova Scotia, where PCBs and PAHs analysis was undertaken by Dr. Kenneth Lee's organic chemical laboratory of DFO and the grain size and total organic carbon analyses were performed by the GSC laboratory at the Atlantic Geoscience Centre. The remaining samples were forwarded by GSC laboratory to the ACME Analytical Laboratories Ltd. in North Vancouver, BC, for heavy metal analysis. Sediment samples for toxicity tests were sent to the Environment Canada Atlantic Laboratory for Environmental Testing Toxicology Laboratory in Moncton for testing while the benthic community samples were sent to Envirosphere Consultants Limited in Windsor, Nova Scotia, for sorting and identification.

Table 1. Station and sample data for survey of Strait of Canso disposal site, July 27-29, 2004.

Station	Date	Sequential Number	Latitude	Longitude	Depth (m)
A1	29/07/2004	224491	45 35.151	61 22.280	40.8
A2	29/07/2004	224492	45 35.285	61 21.932	36.9
A3	28/07/2004	224488	45 35.400	61 21.507	13.6
B1	29/07/2004	224493	45 35.356	61 22.790	26.2
B2	29/07/2004	224451	45 35.526	61 22.170	34.4
B3	28/07/2004	224490	45 35.689	61 21.885	18.3
C1	28/07/2004	224476	45 35.765	61 23.013	22.6
C2	29/07/2004	224460	45 35.851	61 22.451	34.1
C3	28/07/2004	224489	45 35.862	61 22.092	15.5
D1	29/07/2004	224464	45 36.050	61 22.963	30.5
D2	29/07/2004	224449	45 36.134	61 22.781	55.2
D3	29/07/2004	224450	45 36.155	61 22.468	18.9
E1	29/07/2004	224455	45 36.856	61 23.073	29.0
E2	29/07/2004	224458	45 36.843	61 22.560	25.0
E3	29/07/2004	224459	45 36.773	61 22.073	7.7
DUMP1	27/07/2004	224470	45 35.625	61 22.663	57.9
DUMP2	27/07/2004	224471	45 35.656	61 22.691	60.4
DUMP3	27/07/2004	224472	45 35.689	61 22.731	63.7
DUMP4	28/07/2004	224473	45 35.686	61 22.783	57.9
DUMP5	28/07/2004	224474	45 35.763	61 22.822	58.8
DUMP6	28/07/2004	224475	45 35.806	61 22.808	63.4
NEAR1	27/07/2004	224440	45 35.761	61 22.828	~ 55
NEAR2	27/07/2004	224441	45 35.797	61 22.928	~ 37
NEAR3	27/07/2004	224442	45 35.837	61 22.917	33.5
NEAR4	27/07/2004	224443	45 35.775	61 22.671	57.6
NEAR5	27/07/2004	224444	45 35.779	61 22.648	57.3
NEAR6	27/07/2004	224445	45 35.887	61 22.751	61.6
REF1	28/07/2004	224477	45 35.389	61 22.390	55.2
REF2	28/07/2004	224478	45 35.369	61 22.295	53.6
REF3	28/07/2004	224479	45 35.342	61 22.395	58.8
REF3X	28/07/2004	224448	45 35.342	61 22.395	58.8
REF4	28/07/2004	224480	45 35.321	61 22.382	57.0
REF5	28/07/2004	224486	45 35.267	61 22.232	57.0
REF6	28/07/2004	224487	45 35.221	61 22.281	47.5

A second sample (REF3X) is an organic sediment at the base of the sediment column at the site.

Figure 6. Square corer ($0.05m^2$) used for obtaining samples of seabed animal community July 2004.

Chemical Analysis

Heavy metals and organic carbon

Heavy metals analysis was performed by the ACME Analytical Laboratories Ltd. in North Vancouver, BC, using inductively coupled plasma – mass spectrometry (ICP – MS) following digestion of 1.0 g of each sample in *aqua regia* (2 mL HCl + 2 mL HNO_3 + 2 mL H_2O_2)) at 95°C for one hour (ACME Method IF-MS) (EPA Method 3050B). Analysis of duplicate samples and certified reference materials (STSD-1, STSD-2, STSD-3 and STSD-4; Lynch, 1990) was used to monitor analytical accuracy and precision.

Analyses of total organic carbon (TOC) were carried out in the GSC Atlantic laboratory using a LECO WR 112 Carbon Determinator. Organic carbon was determined in 0.5 g of sediment using a combustion method following removal of the inorganic carbon using 1 M HCl. Precision and accuracy were estimated to be ±0.03 wt.% on the basis of replicate analyses of calibration standards (Macintosh *et al.*, 1977).

Polynuclear aromatic hydrocarbons (PAHs) and polychlorinated biphenyls (PCBs)

Sediment PAHs and PCBs analysis was performed following the methods described in King and Lee (2004, 2008). Sediment samples were dried and sieved (1 mm mesh sieve) and stored in cleaned glass vials prior to extraction. Dried sediments were mixed with anhydrous sodium sulphate and copper, placed in an cellulose extraction thimble, spiked with surrogate internal standards [predueterated PAHs d_{10}-phenanthrene, d_{10}-pyrene, d_{12}-benzo[*a*]pyrene, d_{12}-benzo[*b*]fluoranthene, d_{14}-dibenz[*a,h*]anthracene and ^{13}C PCB, ^{13}C CB 77, ^{13}C 153 and ^{13}C CB 194 (Cambridge Isotope Laboratories Andover, Massachusetts USA.)], and covered with clean glass wool. The sediments were soxhlet extracted (Modified version of EPA method 3540 C) with 300-mL dichloromethane for 18 hrs. After extraction, the extracts were concentrated on a Turbo Vap Concentrator to 1 mL. The extracts were purified using a Solid-Phase extraction column packed with silica gel (activated 200 °C for 17 hr and deactivated 5% w/v with HPLC grade water). The column was washed with hexane and the samples were applied to columns as a 1 ml extracts and PAHs are eluated with 10 mL hexane:dichloromethane (4:1 v/v). Additional copper was added if sulfur had not been completely removed.

Concentrated extracts (1.000 mL) of sediments were analysed using high resolution gas chromatography (Agilent 6890 GC) coupled to a mass selective detector (Agilent 5973N) (Willmington, DE, USA) in the selective ion monitoring mode (SIM). Separate SIM methods were used for PAHs and PCB using similar GC conditions. PAHs were obtained from Ultra Scientific, Kingstown, Rhode Island USA; Absolute Standards, Hemden, Conneticut, USA and PCBs from Supelco Canada. A duplicate, certified reference material (NIST 1944 New York-New Jersey Waterway) and operational blank was routinely performed with each batch of 10 samples.

Ammonia, sulfide and Redox analysis for sediment toxicity tests.

Samples were stored at $4 \pm 2°C$ prior to testing. Each of the sediments was homogenized prior to use in each assay.

Sediment samples were thoroughly homogenized and subsampled for analysis of sulfide, redox potential (Eh), and ammonia by specific ion electrodes according to the manufacturer's instructions (Hargrave *et al.,* 1995). A sub-sample of each of the sediments were placed in tared vessels and dried at 100°C for 24 hours. These weights were used to convert results to a dry weight basis. Results for sediments are expressed as μg S/g dry weight of sediment for sulfide, μg NH_3-N/g dry weight of sediment for ammonia and millivolts corrected for the normal hydrogen electrode for redox potential.

In addition to the analysis of the sediment samples, the level of ammonia was measured in the overlying water in the amphipod and polychaete assays at the start and end of the test and in the porewater used for the sea urchin fertilization assay by specific ion electrode. A calculation was performed to determine the amount of unionized ammonia based on the test temperature, salinity, and pH in the water samples (Bower and Bidwell, 1978).

Sediment Toxicity Tests

Amphipod lethality test

Eohaustorius estuarius were purchased from Northwest Aquatic Sciences in Newport, OR, USA. Animals were collected in Yaquina Bay, OR and shipped to the Environment Canada laboratory in Moncton, NB. The animals were acclimated to and maintained at $15 \pm 2°C$ and 30 ± 2 ppt salinity until used for testing. All samples plus a lab control sample of amphipod collection site sediment were tested.

The test was conducted according to the method recommended by Environment Canada (1998). On the day prior to starting the test, each container of test sediment was homogenized and 175 mL portions were added to a 1L glass mason jar (there were no lab replicates tested since there was sufficient field replication). The jars were then filled with 800 mL of clean seawater (salinity was 28 ppt), covered, then aerated overnight with oil free compressed air at a rate of approximately 150 mL /minute.

The following day amphipods were removed from their holding sediment by sieving the contents through a 500 μm sieve. Animals were double-counted and twenty animals were added to each of the test vessels. Aeration was stopped for thirty minutes to assist the animals in burrowing in the test sediment. Testing was performed at 15 ± 1°C with a 24 hour light photoperiod provided by overhead fluorescent fixtures at an intensity of 500 to 1000 lux. The tests were checked daily for observations, aeration and temperature. Three times a week, a field replicate of each sample area was measured for temperature, pH, salinity and dissolved oxygen. At the start and end of the test, five milliliter samples of overlying water were removed from each field replicate and combined. This combined sample was analyzed for concentration of ammonia using a specific ion electrode.

After 10 days, the contents of each jar were sieved through a 500 μm sieve. Any immobile animals were observed under a microscope to determine mortality, defined as lack of all movement when observed under a dissecting microscope for 5 - 10 seconds. Any animals missing were assumed to be dead. The mean survival and standard deviation of each treatment was calculated and compared for statistically different significance from the lab control and the field reference site (Systat Software Inc. 2004).

A reference toxicant test was conducted with cadmium chloride using a water only exposure for 96-hours duration. Using the mortality data at each test concentration, the 96-hour LC50 (concentration calculated to cause 50% mortality after 96-h exposure) was calculated using the methods of Stephan (1977) with the CETIS statistical software (Tidepool Scientific Software, 2002). The 96-hour LC50 for this analysis was entered into the Shewhart Control Chart to ensure that the test was within standard operating limits, and that the population of amphipods used in the test was of normal sensitivity.

Microtox solid phase assay

The test was conducted according to the method recommended by Environment Canada (2002) based on the assumption that, when bacteria are exposed to the sediment sample and if toxic materials are present, they will interfere with the cellular respiration of the bacteria. This interference is measured as a decrease in light output by the bacterium, *Vibrio fischeri* (previously *Photobacterium phosphoreum*).

An aliquot of the wet sediment was transferred to a beaker and stirred for 10 minutes with diluent. A dilution series of 12 concentrations and 3 controls were prepared from this mixture. Bacterial reagent was added to the dilutions and incubated at 15°C for 20 minutes. These dilutions were filtered and the filtrate transferred to the Microtox analyser. After 10 minutes in the analyser, bioluminescence was recorded. Statistics were performed on the data to calculate an EC50 on a wet weight basis (the concentration of test sediment at which light output by a population of the luminescent bacterium was reduced by 50 percent when compared to the untreated control population). Three aliquots of the sediment were dried at 100°C for 24 hours and the percentage moisture determined. The IC50 was corrected for moisture content and quoted on a dry weight basis as mg dry sediment/litre of diluent.

A reference toxicant test using a NRC Certified Reference Material (HS-5) was performed in the same manner as the sediments. The IC50 for this analysis was entered into the Shewhart Control Charts to ensure that the test was within standard operating limits, and that the population of bacteria used in the test was of normal sensitivity.

Sea urchin fertilization assay

White sea urchins, *Lytechinus pictus*, tested during the study were from the Environment Canada Laboratory stock (received from Marinus Inc. of Long Beach, California,).

A portion of each of the sediments was centrifuged at 10,000 g for 15 minutes at 4°C. The supernatant liquid was transferred to a new centrifuge tube and centrifuged for an additional 15 minutes at 10,000 g at 4°C. Porewater was measured for temperature, pH, salinity and dissolved oxygen. If dissolved oxygen was low (<40 % saturation), samples were aerated for a maximum of 20 minutes or until 40 % saturation was obtained. No lab replications of the undiluted porewater were prepared as there were sufficient field replicates. The control/dilution water was laboratory seawater supply filtered through a 0.45 μm filter prior to use, and stored for no more than 24 hours, five replicates of the control water were also prepared. A portion of the porewater was also tested for ammonia.

The test was conducted according to the method recommended by Environment Canada (1992). Sea urchins were injected with 0.5 mL of 0.5 M potassium chloride solution to induce spawning. Eggs produced from all females were pooled, and the concentration was adjusted to 2000 eggs/mL. Sperm were pooled from all males using the "dry" spawning technique, then stored in a vial on ice. A fixed "sperm to egg ratio" of 20,000:1 was used to produce approximately 90% fertilization in the controls. Sperm were activated with seawater immediately prior to the start of the test.

Test volume was 10 mL and test temperature was 20 ± 1°C. Sperm were exposed to the test solutions for 10 minutes, followed by an additional 10 minutes exposure of the sperm and eggs. The test was then terminated using 1 mL of 0.5% glutaraldehyde per replicate. One hundred eggs were examined from each field replicate at 100 times magnification to check for the presence of a fertilization membrane. The mean percent fertilized in undiluted porewater was calculated for each sample. A blank sample containing eggs but no sperm (four replicates) was also analysed to ensure no false fertilization occurred.

A reference toxicant test was performed simultaneously with the porewater toxicity tests using copper sulfate. The IC50 was calculated using log dose transformed data. The value was entered into the control chart to ensure that the test was within standard operating limits, and that the test gametes were of normal sensitivity to the reference toxicant.

Polychaete survival and growth test

The test organisms were juvenile polychaetes, *Polydora cornuta*, collected from laboratory cultures. The polychaetes were approximately 3 weeks old at the start of the test, the mean dry weight was between 0.06 and 0.5 mg.

The method used for this study was recommended by Environment Canada (2001). On the day prior to starting the test, each container of test sediment was homogenized and a 50 mL portion was added to a 300 mL tall form glass beaker (there were no lab replicates tested since there was sufficient field replication). The beakers were then filled with 200 mL of control/dilution water. The control/dilution water was natural seawater. A laboratory control was also started for QA/QC purposes and consisted of five replicated tall form glass beakers each containing 50 mL portions of collection site sediment. The test beakers were covered, then aerated overnight with oil free compressed air at a rate of 2 to 3 bubbles/ second.

The following day water quality was measured on the overlying water in the test chambers. One of the field replicates from each sample area was tested for temperature, dissolved oxygen, salinity and pH. A sample of the overlying water was taken from each replicate from each sample area and combined. This combined solution was measured for ammonia.

Five worms were counted into each of the transfer dishes. The animals were double counted checked to ensure they were complete and appeared healthy, then added to each of the test vessels. Forty-five worms were measured for length. Groups of fifteen were combined, rinsed with distilled water, and dried to determine the initial dry weight of test organisms. Testing was performed with 16 hour of light and eight hours of darkness with lighting provided by overhead fluorescent fixtures at an intensity of 500 to 1000 lux. Testing was performed at 23 ± 1°C. Tests were checked daily for observations, aeration and temperature. Three times a week, one field replicate of each sampling area was monitored for temperature, pH and dissolved oxygen. The test solutions were fed three times a week with 10 mg dry weight of a food mixture. The mixture was a slurry suspension of sea lettuce leaves and Nutrafin fish food. On day 7 of the experiment, the overlying seawater was renewed by siphoning off about 80% of the water and replacing with clean seawater.

After 14 days, the same chemical measurements were performed as those at the start of the test. The contents of each jar were sieved through a 0.5 mm sieve. All of the worm tubes were collected and transferred to a disposable dish. The worm tubes were gently prodded with a small artist's paint brush to check for presence of worms. Any animals missing were assumed to be dead. The mean survival and standard deviation of all replicates from each sample area were calculated. Mean percentage survival of polychaetes exposed to sediments from each sample area was statistically compared to mean percent survival of polychaetes exposed to the clean, field reference area and to the laboratory control sediment.

All surviving worms were rinsed with distilled water and transferred to a tared weighing dish. The worms from each test chamber were combined into a single weighing dish, dried at 60°C for approximately 24 hours, cooled in a dessicator for 15 minutes and the dry weight was determined. Organisms were dried to a constant weight (±0.05 mg or less for repeated weighing of the same sample). The average dry weight of test organism for all replicates from each of the sampling areas were compared to the clean, field reference area and to the laboratory control sediment using ANOVA statistics in SigmaStat statistical software (Systat Software Inc., 2004) to determine if exposure to the sediments from each sampling areas caused a significant decrease in organism growth.

Polychaete weight is the average weight of individual worm in milligrams. The growth factor is: the average weight of individual polychaete in test sediment from a sampling area at end of the test (in milligrams)/average weight of individual at start of test (in milligrams); so there are no units for the growth factor.

A reference toxicant test was conducted with cadmium chloride using water only exposures for 96-hours duration. Using the mortality data at each test concentration, the 96-hour LC50 (concentration calculated to cause 50% mortality after 96-h exposure) was calculated using the methods of Stephan (1977). The value was entered into the control chart to ensure normal operating conditions were maintained, and that the population of polychaetes used in the test was of normal sensitivity. No organism dry weights were determined in this reference toxicant test protocol.

Benthic Biota Sorting and Identification

In the laboratory, sediment samples were transferred from formalin to 70% isopropanol one week after sampling, and animals were later removed by stereomicroscope at 6.4x magnification with a subsequent final check at 16x magnification. Sorting efficiency was checked on 10% of samples, and was greater than 95%. Organisms were collectively blotted dry and weighed to the nearest milligram to determine wet weight (Rowe and Menzel, 1971). After they had been identified, animals were further weighed to provide wet weight of each species. Animals other than polychaetes were identified by Envirosphere Consultants Limited while polychaetes were identified by Arenicola Marine, Wolfville, Nova Scotia.

Benthic Biota Data Analysis

For each sample, number of species, number of individuals/m^2, biomass, and several indices of community structure commonly used in benthic studies, such as the Shannon-Wiener Index (H'), Pielou's Evenness (J'), Simpsons, were determined. H' is widely used in ecology (Clark and Warwick, 2001) and represents both the number of species and its distribution among individuals, while J' and Simpsons Index are measures of distribution among individuals (equitability) (Pielou, 1974) (Legendre and Legendre, 1983). Indices were used here principally in interpreting patterns of abundance and dominance in benthic communities. Several other measures were also estimated and presented (e.g. Margalef's Index, McIntosh's index).

Multivariate analysis including cluster and coordination were performed on the benthic community data to study the similarity and occurrence of dominated species at the disposal and reference sites. The Abundance Biomass Comparison (ABC) Curve were plotted to asses the degree of disturbance on the communities at the disposal, nearfield and reference sites.

Statistical Methods

The TOXSTAT® statistical computer programs (WEST, Inc., 2003 Central Avenue, Cheyenne, WY, USA) were used to test for normality of the data before ANOVA or the Kruskai-Wallis ranking test was performed to compare the chemical and biological endpoints for disposal site, nearfield, reference and the outside areas. For the benthic community endpoints, the stations at the shoreline area were excluded from the ANOVA analysis because of the small sample size. Pearson Correlation analysis and scatterplot were performed by SYSTAT® (SYSTAT Software, Inc., Richmond, CA) and Statgraphics® (STSCm Ubc,m Rockville, MD) computer programs to assess the relationships between the concentrations of sediment contaminants and toxicity endpoints, benthic indices, and also to detect the potential impacts of confounding factors, such as grain size, sulfide and ammonia, on the toxicity endpoints and the abundance of individual benthic species.

The Primer® (PRIMER-E Ltd., Plymouth, UK), an ecological research software package, was used to calculate univariate diversity indices, distribution of benthic communities, clustering and ABC plots (Clarke and Gorley, 2001).

RESULTS AND DISCUSSION

Physical Monitoring

Multibeam bathymetry data showed the disposal site is a deep trough located in 55 – 64 meters of water in the Strait (Figures 2 and 3). Dredged materials characterized by rough surfaces appeared near the center of the disposal site in the deepest water depths (Figure 7). Most of the dredged material was located within the boundaries of the disposal site. In the area east of the disposal site outside the disposal site boundaries, dredged material, appearing as faint circles on the multibeam bathymetry data, was detected. The subdued relief present on these features indicates that they may predate the initiation of controlled disposal in the Strait (Figure 8). Figures 8 and 9 shows the acoustic backscatter data extracted from the multibeam bathymetry

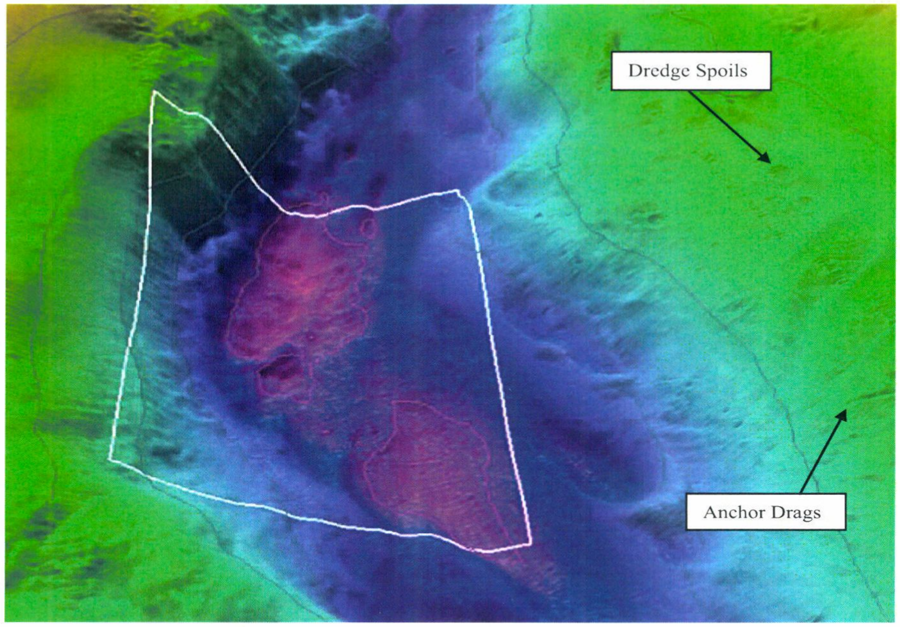


Figure 7. Dredged materials characterized by rough surfaces appear near the center of the disposal site in the deepest water depths with contours shown at 10 metre intervals.

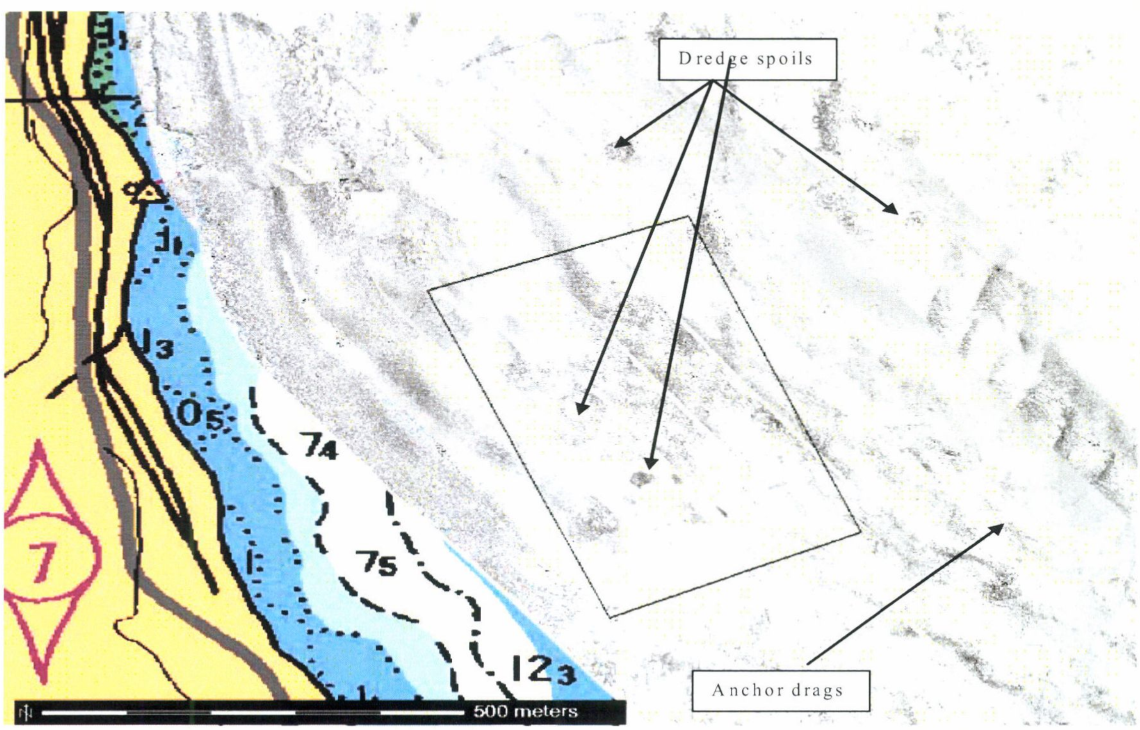


Figure 8. Sidescan sonar data showing seafloor character at the disposal site, (black rectangle). Dredge spoils and anchor drags are evident.

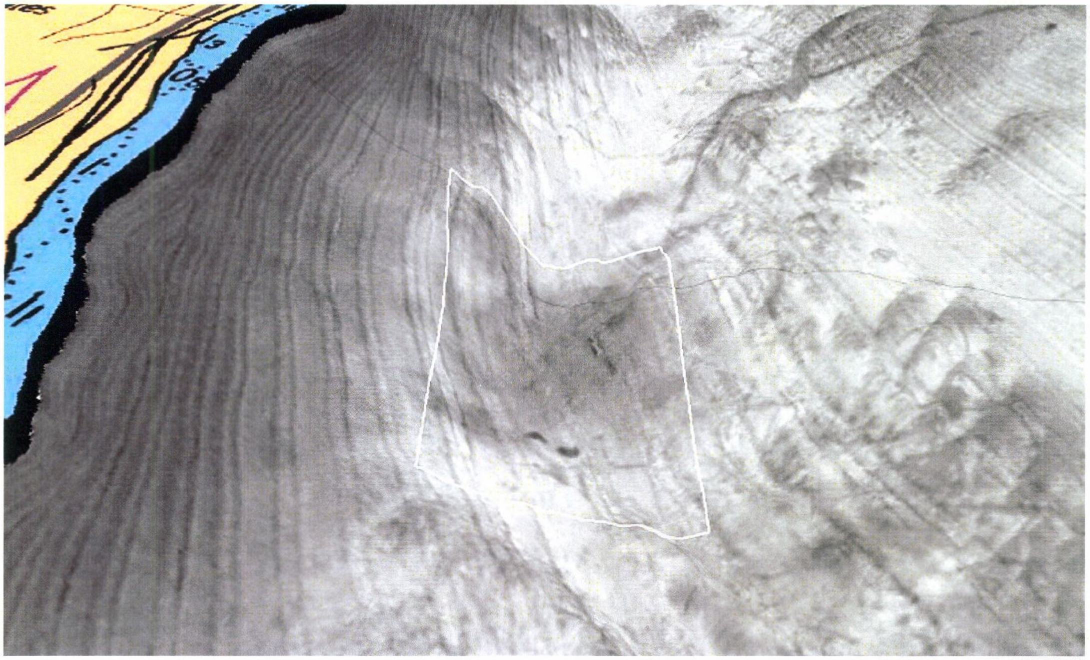
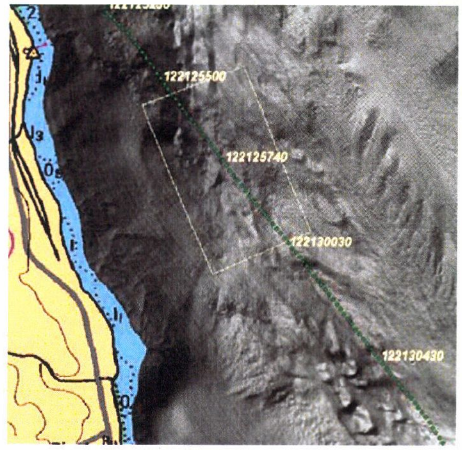


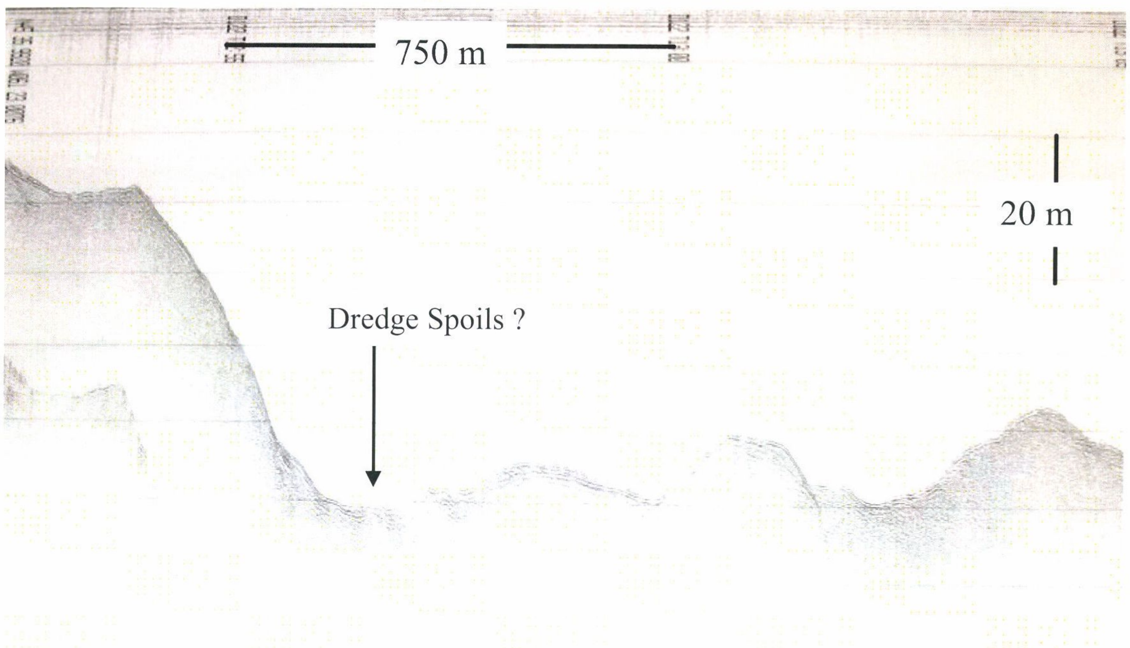
Figure 9. Backscatter intensity of the disposal site shown in oblique views with the backscatter draped over the bathymetry.

data. Two deposits with high backscatter intensity are evident in the southern portion of the disposal site indicating that this may be the result of the disposal of coarse material at the site. Very little relief is associated with the deposits.

Sub-bottom profiler data showed that the seafloor at the disposal site is composed of coarse sediments, probably glacial till, overlying bedrock. An accumulation of about 1 meter of recent sediments in this area is probably related to the most recent disposal of dredged materials at the site (Figure 10).

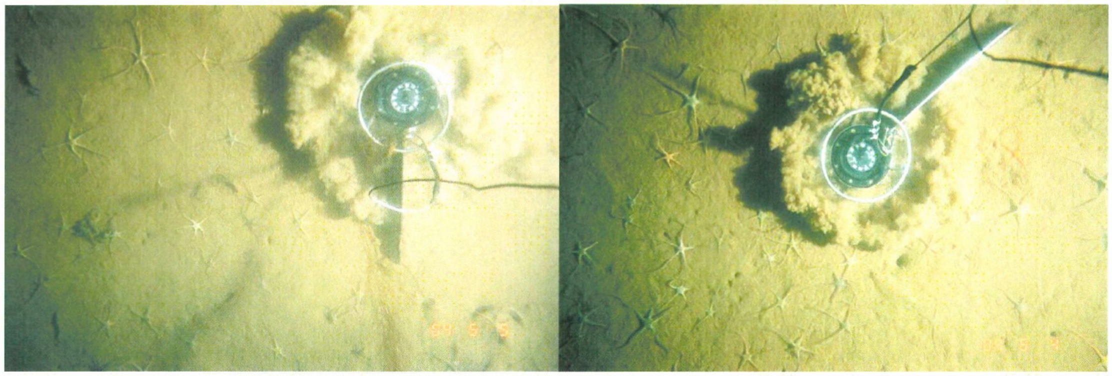
Seafloor photographs at the disposal site and other deeper portions of the Strait show a seafloor covering with fine-grained sediment which overlies coarser material in places. The trigger weight for the camera disturbed the seafloor sediments, and caused suspension of the fine material at the seafloor (Figure 11). Large numbers of the brittle star *Ophiura sarsi* and burrows





b)

Figure 10. Location a) and Seistec sub-bottom profiler data b) through the disposal area. The Seistec data is shown from Day 122 12:53 to 13:05.



(a) (b)

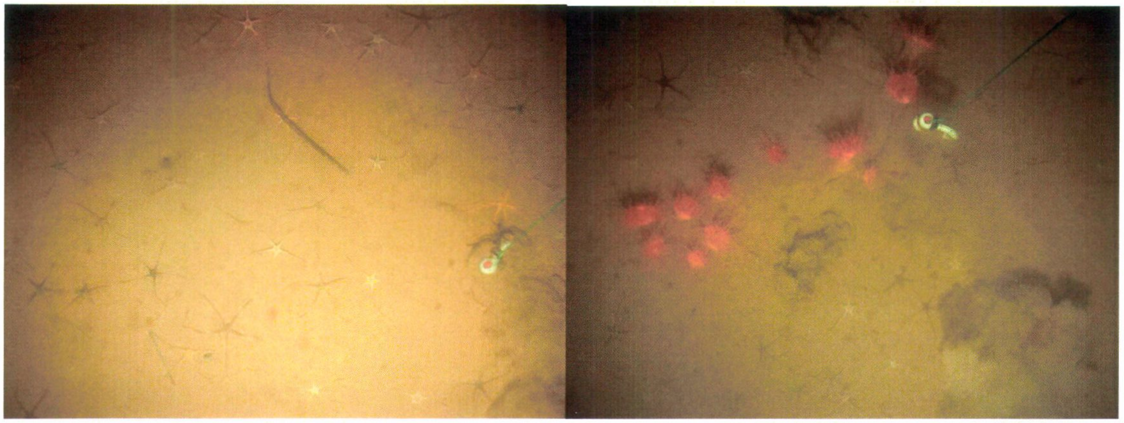
Figure 11. Seafloor photographs from within the disposal site from a) Hart 2004010 Station 19 and b) Station 3. (Russell et al., 2005b).

of unknown origin were present throughout the area. Sea anemone *Hormathia nodosa*, were visible in many images as shown in figure 12. Coarse sediment and pebbles were visible in the disposal site. Photographic transects through the disposal site also showed the presence of boulders, cobbles and gravel, along with assorted debris including tires, trees, wood and a trailer (Russell *et al.*, 2005b). It seems that the site may have been used in the past for dumping of debris before it was designated by Environment Canada for dredged material disposal.

Sediment Characterization

Sediment particle size

Sediment particle size data are summarized in Table 2 and Figure 13. In general, sediments samples collected for this study are high in silt (59.55% - 76.57%) and clay (10.25% - 21.31%), especially those samples collected from the disposal (76.57% silt; 16.19% clay) and reference sites (76.3% silt; 21.31% clay). Coarser samples were mostly found at the area near or outside of the disposal site. Samples from the stations closer to the shoreline in shallow water area contained more sand (up to 27.62% sand). At the disposal site, the amounts of silt/clay mixtures increased to 92% silt/clay from the 51% reported by Georgia Pacific Inc. after the 1999 disposal.



(a) (b)

Figure 12. Seafloor photographs from Canso Strait showing a) the soft sediments and high density of brittle stars present throughout much of the area. Sea anemones b) were present in many areas, indicating the presence of hard substrate under the fine sediments. Note how the range marker for the camera (a 5 cm shackle) caused suspension of the fine material. (Russell et al., 2005b)

The reference site has a significantly (Fisher LSD = 2.407 – 2.691, $p = <0.001$) higher percentage clay than all the other sites.

Buckley *et al.* (1974) and Lewis and Keen (1990) reported that, after construction of the causeway in 1954, the Strait of Canso exhibited conditions more closely related to a tidal inlet, or fjord, which is much more prone to deposition of fine-grained sediments. It has been shown that the quieter current regime has allowed fine-grained sediments to accumulate at rates in the range of 1 to 2 mm per year. That may explain that percent silt and clay are the highest in the deepest parts of the Strait where the disposal, nearfield and reference sites of this study are located. The presence of a veneer of fine mud overlaying most of the seafloor within the Strait was also reported by Parrott *et al.* (2005b) and Envirosphere Consultants Ltd. (2004b).

Table 2. Particle size, trace metals, TOC, PCBs and PAHs in sediment samples collected at the Canso disposal site and outside areas in surveys conducted in 1983, 1985, 1987 and 2004. (* TOC 1985 n = 46; 1987 n = 34; 1999 n = 3)

	1983	1985	1987	1987 Outside Surface	1987 Outside Bottom	1999	2004	2004 Outside
	n = 4	n = 24	n = 17	n = 17	n = 17	n = 8	n = 6	n = 15
Particle Size (%)								
Gravel	0	7.9 ± 10.8	1.6 ± 3.3			73.1 ± 1.4	0.65 ± 0.25	1.99 ± 0.74
Sand	4.01 ± 1.3	27.2 ± 12.9	35.3 ± 11.1			26.1 ± 4.0	6.59 ± 1.63	23.40 ± 4.17
Silt	51.9 ± 2.0	40.0 ± 11.4	44.1 ± 9.1			0.6 ± 0.1	76.57 ± 1.88	62.94 ± 4.09
Clay	44.1 ± 1.4	26.5 ± 9.4	19.0 ± 4.7			0.5 ± 0.01	16.29 ± 0.81	11.67 ± 0.70
TOC*		1.44 ± 0.05	6.74 ± 0.47			3.87 ± 0.04	3.22 ± 1.22	5.14 ± 1.23
Trace Metal (mg/kg)								
Cd	0.45 ± 0.17	0.65 ± 0.16	0.54 ± 0.11	0.478 ± 0.131	0.324 ± 0.105	0.04 ± 0.01	0.24 ± 0.03	0.56 ± 0.12
Zn	116.3 ± 3.4	139 ± 28	141.1 ± 12.6	127.235 ± 22.081	112.000 ± 36.258	91.75 ± 4.00	110.62 ± 2.12	112.96 ± 6.92
Pb	44.5 ± 0.5	58.9 ± 14.2	48.4 ± 21.1	40.371 ± 7.601	34.529 ± 8.092	21.00 ± 1.44	30.90 ± 32.27	29.34 ± 1.36
Cu	37 ± 5	32.0 ± 17.8	103.1 ± 113.1	29.306 ± 9.706	53.024 ± 94.062	22.75 ± 2.38	26.49 ± 0.86	25.03 ± 1.53
Organic Compounds								
∑ PCBs (µg/kg)	80 ± 4	167 ± 216.8	157.8 ± 101.1	329.235 ± 333.093	427.705 ± 610.356	52.81 ± 13.15	76.67 ± 5.10	229.93 ± 41.74
∑ PAHs (mg/kg)						0.67 ± 0.28	2.27 ± 0.18	1.73 ± 2.43

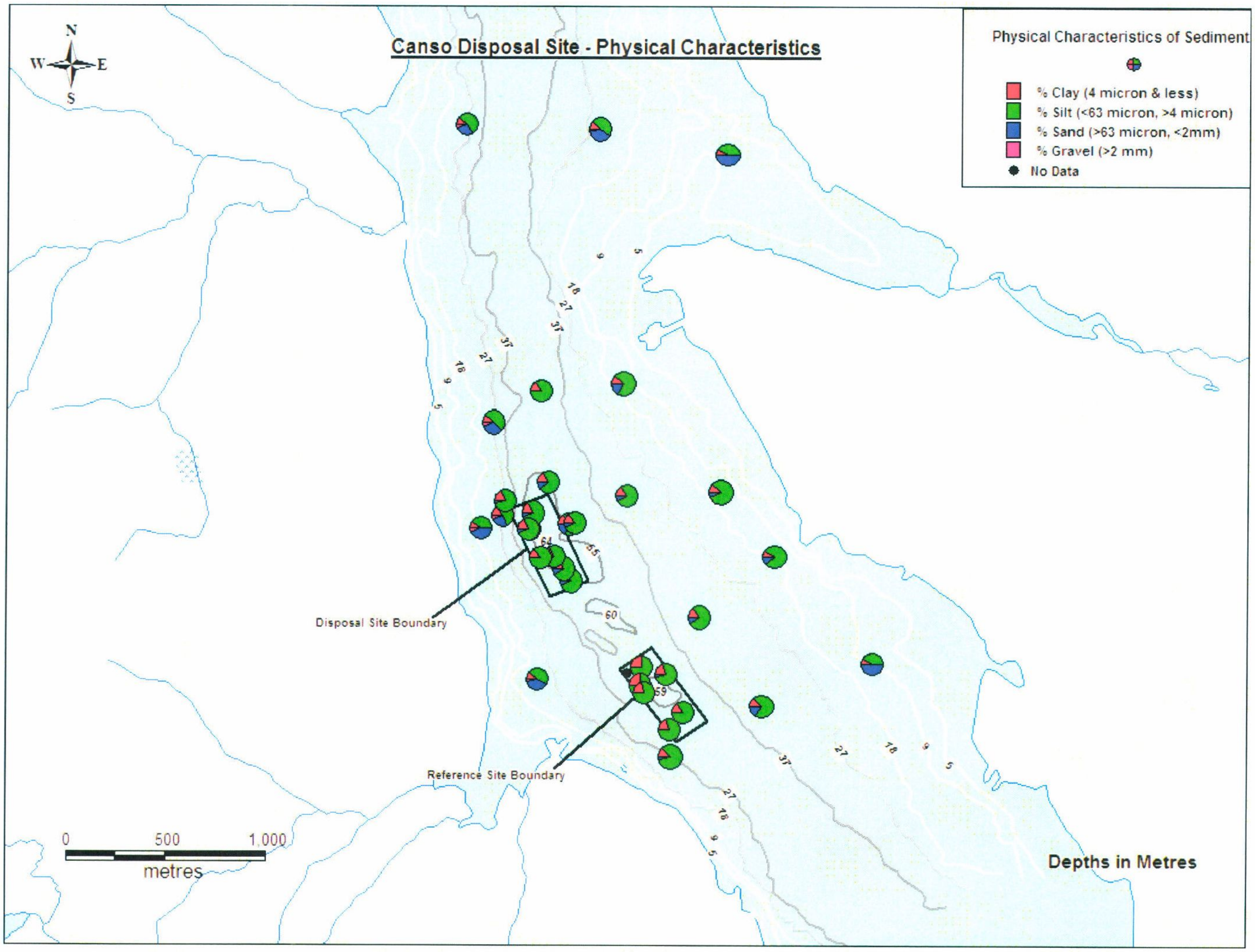


Figure 13. Sediment particle size in samples collected from the disposal site, nearfield site, reference site and the shoreline areas.

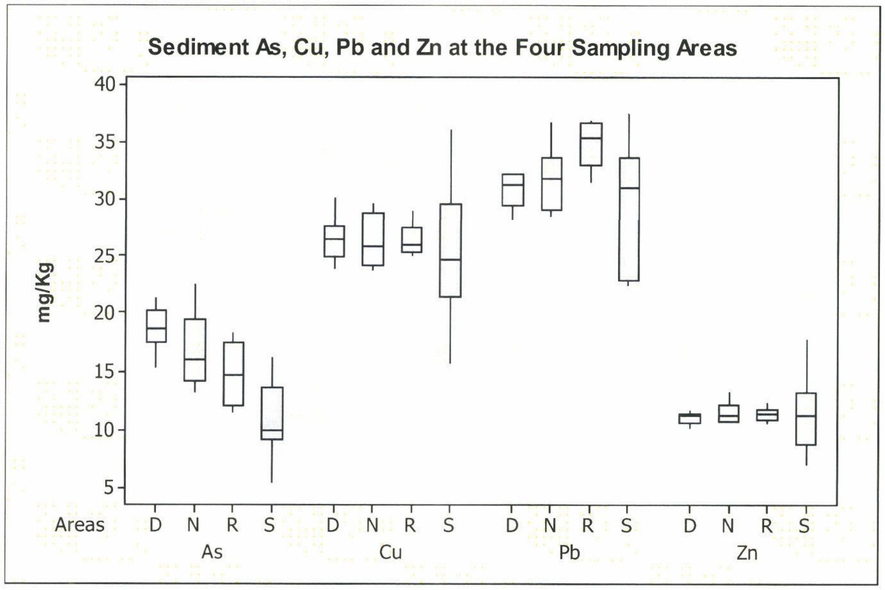
Heavy metals

Heavy metal data are summarized in Table 2 and Figure 14. The mean concentrations of heavy metals in all sediment samples collected at the disposal site, nearfield site, reference site and the outside area are lower than the probable effect levels (PEL) of the Canadian Council of Ministers of the Environment (CCME) Interim Marine Sediment Quality Guideline. Two of the metals, Zn and Hg, are lower than the threshold effect levels (TEL). Cd concentrations (0.8 – 1.82 mg/kg) exceeding the Disposal at Sea (DAS) regulated limit of 0.6 mg/kg are only detected in sampling stations outside the disposal site and along the eastern shore of the Strait near the area with high density of industrial sites. Elevated concentrations of heavy metal along this costal area have also been reported by Buckley (1974) and OceanChem (1987), especially the area near the pulp and paper mill where large quantities of wood fiber on the seafloor were observed. However, the mean concentrations of Cd between the four studied areas (disposal, nearfield, reference and outside area) are not significantly different.

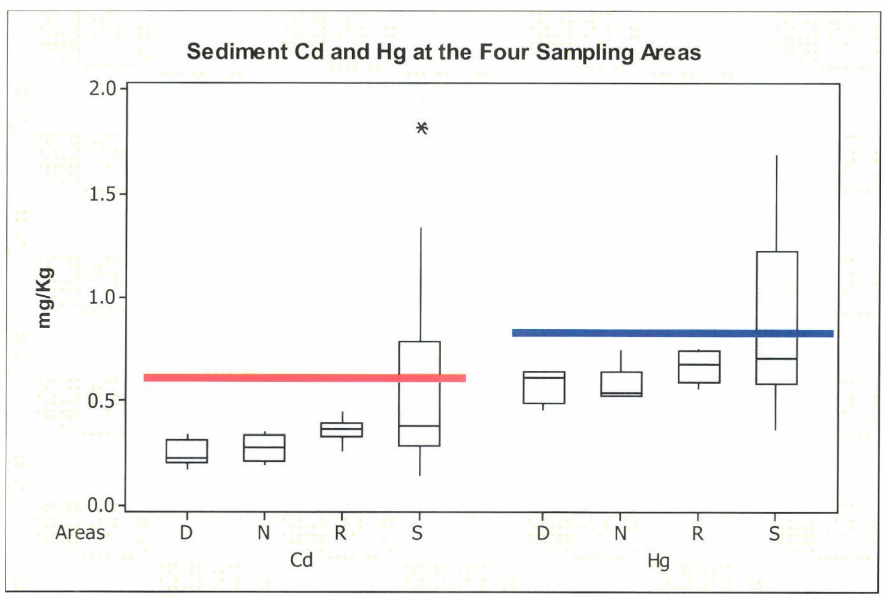
As concentration was significantly higher at the disposal site than the reference site (Fisher LSD = 3.003, $p < 0.001$) and the outside area (Fisher LSD = 3.357, $p = 0.025$) while Pb was significantly higher (Dunn's Q = 2.943, $p < 0.05$) at the reference site than the outside area.

Mean concentrations of Cu, Pb, Zn and Cd in the disposal site sediment samples from the present study were lower than those reported in the 1983 pre-dump survey and the 1985 and 1987 monitoring programs (Table 2; Figure 15).

AGC collected 56 surface sediment samples in the Strait during their bathymetric and side scan survey for geochemistry study. Their data showed a strong positive correlation between Li, which is a common constituent of clay minerals, and Pb (Figure 16) (Russell *et al.*, 2005b). Similar relationships were also found between Li and the other two heavy metals, Cr and Ni. Russell et al. (2005b) concluded that these relationships suggest that variations in sediment grain size are primarily responsible for controlling the distribution of these elements (Loring, 1991). Other parameters, e.g. Cd and Hg, do not correlate well with Li, suggesting that the concentrations of these elements are mainly controlled by factors other than grain size.



(a)



(b)

Figure 14. Sediment heavy metal concentrations at the four sampling areas. (a) As, Cu, Pb and Zn sediment concentrations (Zn = Zn/10); (b) Cd and Hg sediment concentrations (red and blue lines are the Cd and Hg DAS limits (Cd = 0.6 mg/kg; Hg = 0.75 mg/kg)). (D = Disposal site; N = Nearfield site; R = Reference site; S = Outside areas).

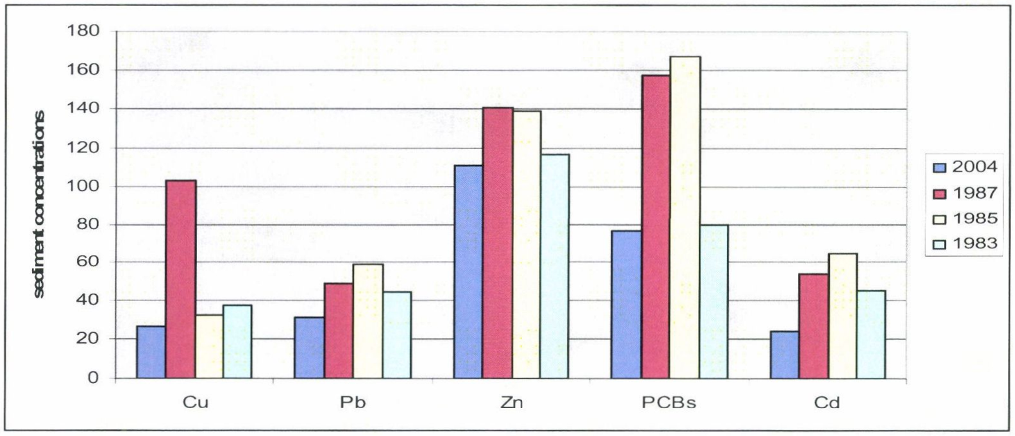


Figure 15. Concentrations of Cu, Pb, Zn, Cd and PCBs in disposal site sediments from 1983 to 2004. Cu, Pb and Zn - mg/kg; PCBs – µg/kg; Cd = Cd x 100.

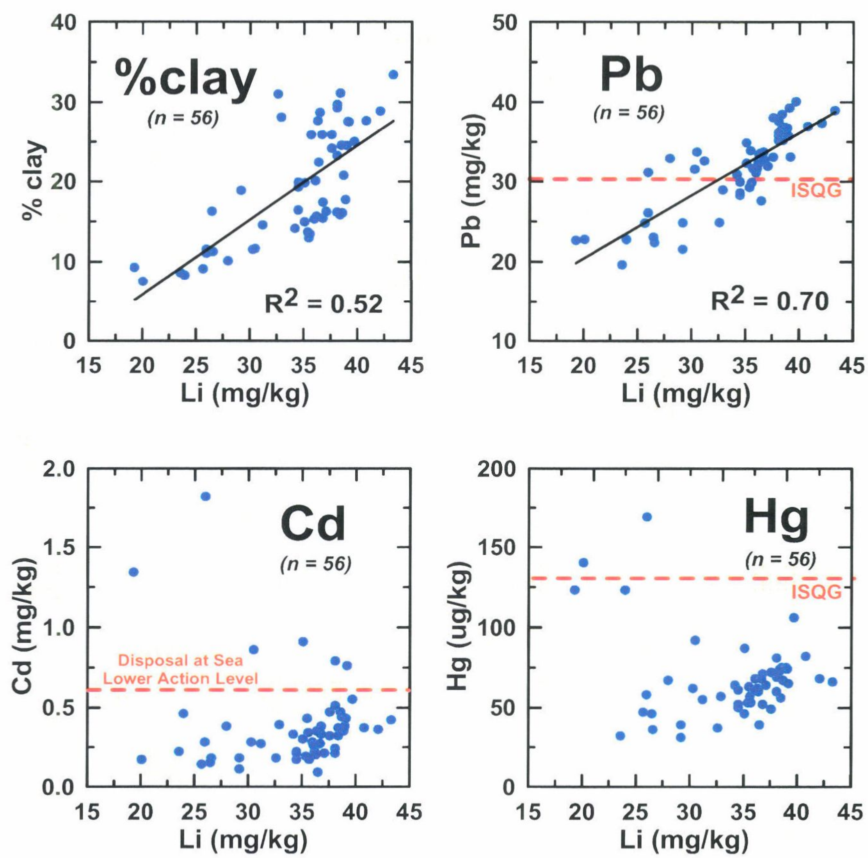


Figure 16. Lithium concentrations in surface sediments (0-5 cm) from the Strait of Canso plotted against %clay, and the concentrations of Pb, Cd, and Hg. The red horizontal line on the Pb and Hg plots is the CCME Interim Marine Sediment Quality Guideline. The red horizontal line on the Cd plot is EC DAS's Lower Action Level. (Russell et al., 2005b)

Total organic carbon (TOC), PAHs and PCBs

The concentrations of total organic carbon (TOC) are presented in Table 2 and Figure 17.

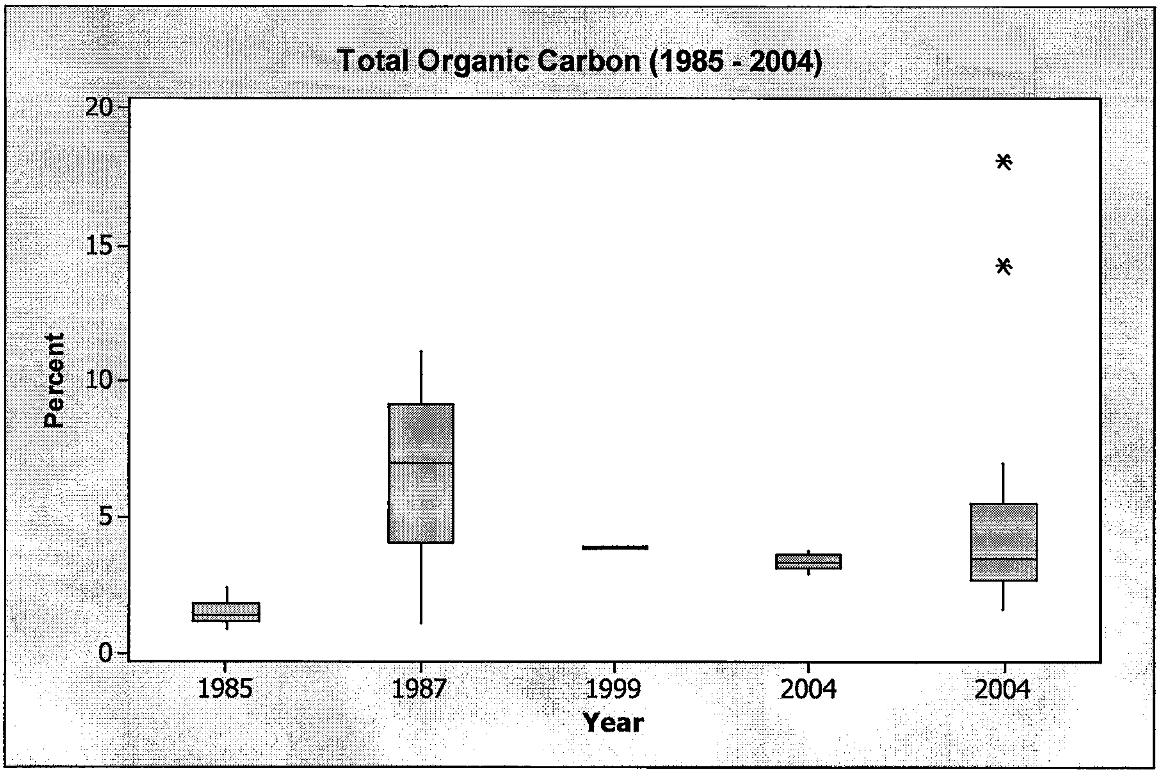


Figure 17. Total organic carbon in sediment samples collected from 1985 to 2004 at the disposal site. The last box plot on the right side of the graph is TOC in sediment samples collected from the outside area which contained two very high TOC.

In general, TOC in all sediment samples collected in this study are quite similar to those reported in 1987 and 1999 (OceanChem 1987; Tay 1987). TOC in sediments collected in 1985 are significantly lower than all other years.

The concentrations of total sediment PAHs are presented in Table 2 and Figure 18. One of the six sediment samples from the disposal site (3.1 mg/kg) exceeds the DAS regulated limit of 2.5 mg/kg. Similar elevated levels of PAH (3.2 – 3.3 mg/kg) were also found in areas outside of the disposal site. However, there is no significant difference between these PAH concentrations and those detected in sediments collected from all the other studied areas.

Table 2 and Figure 18 showed the concentrations of total sediment PCBs at all the studied sites. PCBs in the disposal site sediments are the lowest among all the study areas. The highest concentration of PCBs which is many times higher than the DAS regulated limit of 100 μg/kg was found outside of the disposal site, with the highest concentration of 535 μg/kg occurring near the shoreline closest to the industrial areas within the Strait. High PCBs in sediment samples collected from the Strait outside of the disposal site have been reported in the past studies (Tay 1987; OceanChem 1987). The concentrations of sediment PCB at the disposal site did not change significantly since 1985 (Figure 19).

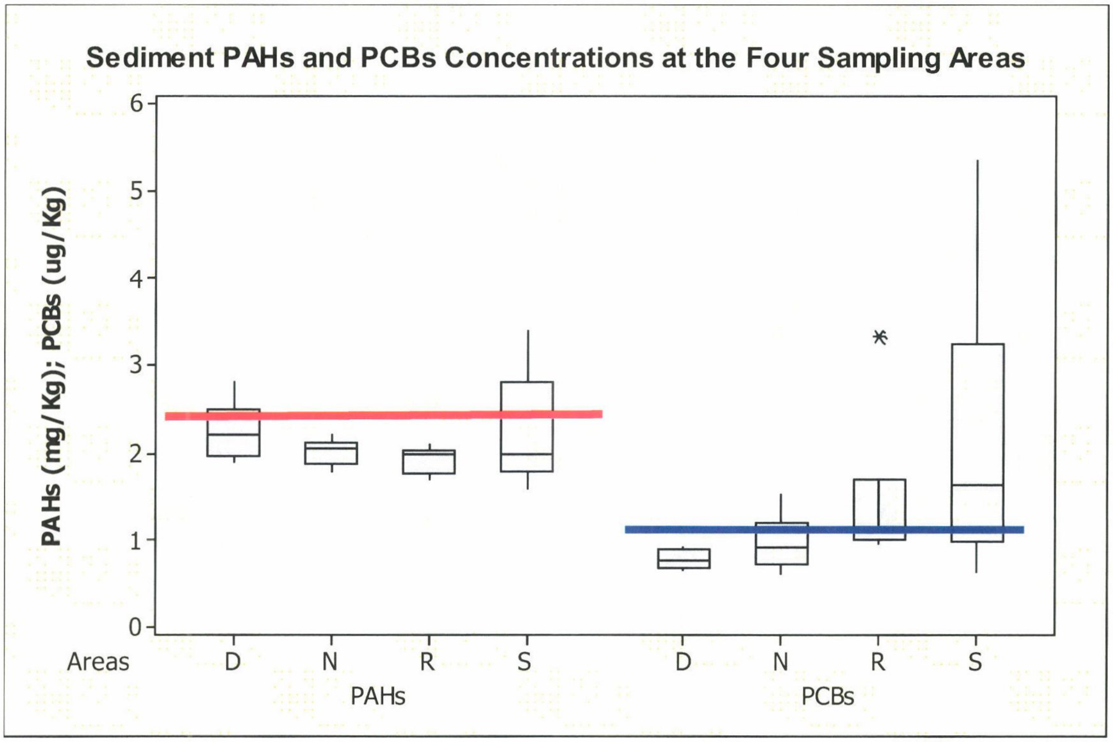


Figure 18. Sediment PAH and PCB concentrations at the four sampling areas. The red and blue lines are the PAH and PCB DAS limits (PAH = 2.5 mg/kg; PCB = 1 mg/kg). (D = Disposal site; N = Nearfield site; R = Reference site; S = Outside areas).

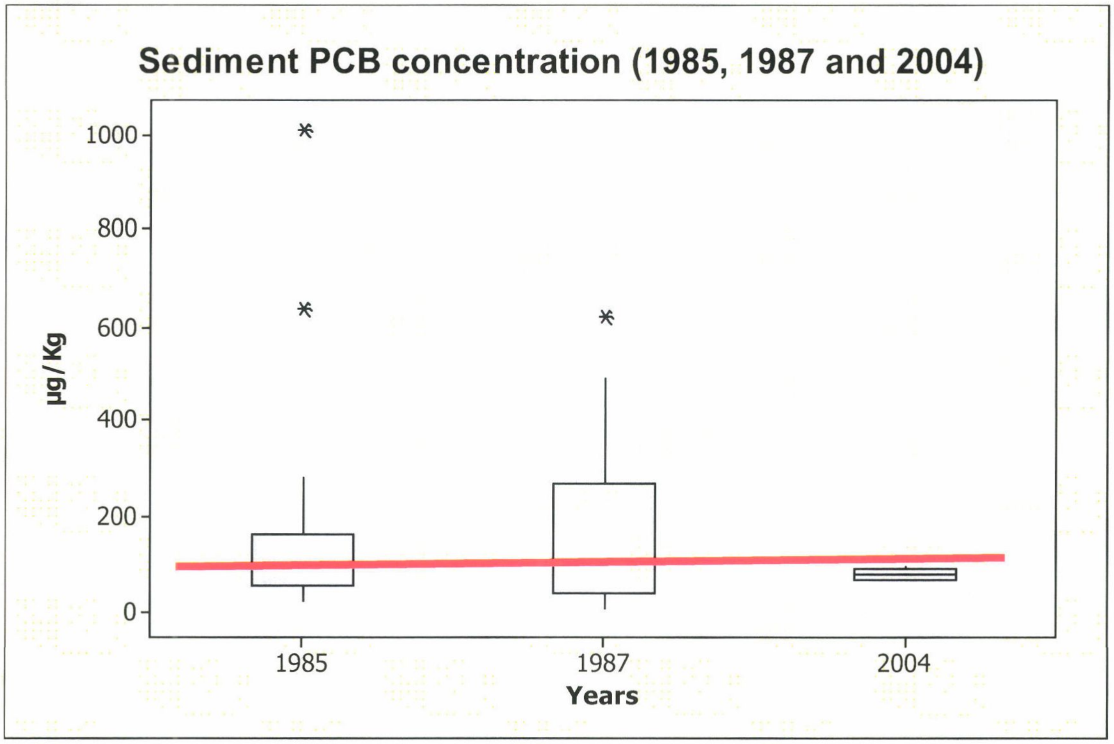


Figure 19. Total sediment PCBs in sediment samples collected from 1985 to 2004 at the disposal site. The red line is the DAS Regulated Limit (100 µg/kg).

Sediment Toxicity Tests

Four sediment toxicity tests, amphipod lethality, Microtox solid phase, sea urchin fertilization, and polychaete survival and growth were employed in this study.

The interpretation criteria for all the toxicity tests conducted in this study are listed in Table 3. These criteria were developed and recommended by EC Disposal at Sea Program, the EC Moncton Laboratory, and the EC Methods Development and Applications Section.

Table 3. Interpretation criteria for sediment toxicity tests.

BIOASSAY	CRITERIA	RESTRICTIONS
Amphipod test on whole Sediment	If, compared to the reference, the survival of the test organisms is statistically different, and at least 20% lower, the test sediment is considered toxic.	The control survival must be at lest 90%; 1. For *Eohaustorius estuarius*, the reference sediment must have a mean survival of ≥ 80% to be considered as acceptable. 2. If the reference survival is at least 20% lower than the control survival and is statistically different, the reference samples should be abandoned; 3. If reference samples are unavailable or abandoned, a sediment sample is considered toxic if control % survival is at least 30% higher than the treatment % survival and is statistically different; 4. The acceptability of the reference is judged by: (1) the chemical measurements of the contaminants and measured toxicity being low and (2) the grain size and organic carbon results being similar to those of the test sediments.
Microtox Solid Phase Test	Test sediments producing a 5 minute IC_{50} ≤ 1,000 mg/L dw are toxic regardless of any grain size characteristics.	Microtox solid phase test is sensitive to the percent fine in a sediment sample. To avoid the grain size effect, a second criterion is proposed for coarse grained sediments. A sediment with <20% fines and has an IC50 ≥ 1000 mg/L, the IC50 of the sediment must be compared against a clean reference or negative control sediment with a percent fines that does not differ by more than 30%. The sediment is judged to have failed the toxicity test if (1) the IC50 is more than 50% lower than that determined for the reference or control sediment and (2) the IC50's for the test and reference sediment differ significantly

BIOASSAY	CRITERIA	RESTRICTIONS
Sea Urchin Fertilization test on sediment porewater	If, compared to the control, the fertilization success in test sediment is statistically different and at least 25% lower, the test sediment is considered toxic, that is, if an IC_{25} can be calculated.	Uncontaminated sediments contain naturally occurring ammonia and sulfides can be toxic in the sea Urchin Fertilization test on sediment porewater. To correct for this interference, results from the test sediments should be compared to the results from the reference sediments to determine if that are statistically significantly different.
Polychaete growth and survival test on whole sediment	The test sediment is considered toxic, if compared to the reference: (1) the average mortality of the polychaetes in the test sediment is statistically different, and at least 20% lower; (2) the average dry weight of the polychaetes in the test sediment is statistically different, and at least 25% lower; (3) the average growth of the polychaetes in the test sediment is statistically different, and at least 25% lower.	If field reference sediments are unavailable or unacceptable, the control sediments will be used for comparison using the same criteria for the reference samples. The test is sensitive to large differences in sediment grain size, TOC, sulfide and ammonia. If sediment grain size is outside the range considered suitable for normal growth of the test organisms (Environment Canada 2001), it may not be possible to reach a conclusion whether a sediment sample is toxic or not.

Toxicity tests results and associated sediment quality data, such as ammonia, sulfide and Redox of the test sediments are presented in Table 4.

Amphipod lethality test

The percent survival of amphipod was 99 ± 2.2% in the control test. Reference toxicant test conducted with cadmium chloride using a water only exposure for 96-hours duration was 7.78 (6.58-9.19) mg/L Cd (warning limits for the EC laboratory is 4.33 – 16.9 mg/kg).

None of the test sediments was toxic to the amphipod (Table 4). The lowest percent survival of amphipods was 65% in a sample from the disposal site. No significant difference among the endpoints ($F_{3,24}$ = 2.504, p = 0.083) was detected between the sampling stations from the disposal, nearfield, reference sites and the outside areas, even in stations with elevated concentrations of ammonia (30.8 mg/kg) and sulfide (214 mg/kg).

Table 4. Results of the biological tests (polychaete, amphipod, Microtox and sea urchin) employed in this study and the concentrations of NH_3, S and Redox in the test sediments.

Station	Polydora Survival (%)	Polydora Weight (mg)	Polydora Growth Factor[1]	Amphipods (%)	Microtox (EC_{50} (mg/kg dw)	Sea Urchin % fertilization	NH_3 (μg/g)	S (μg/g)	Redox (MV)
A1	100	1.6960	8.6580	80	1930	32.8	17.60	25.7	34
A3	100	1.8940	9.6688	100	22300	35.5	8.40	54.2	84
B1	80	0.2925	1.4932	75	18300	41.0	9.10	3.5	56
B3	0	0.0000	0.0000	85	4110	55.3	23.00	214.0	147
C1	60	0.3117	1.5910	85	4850	48.0	6.14	7.2	38
C3	60	0.6033	3.0800	90	9740	31.3	9.99	98.9	55
D1	80	0.3075	1.5698	80	3860	49.8	10.20	15.3	0
D3	80	0.8583	4.3817	95	4580	43.5	14.50	44.7	90
E1	100	1.8430	9.4084	85	3650	41.8	9.01	17.4	27
E3	60	0.5367	2.7397	95	38500	58.3	15.90	14.3	47
DUMP1	80	1.6263	8.3019	75	1860	58.8	4.70	84.5	16
DUMP2	20	0.0000	0.0000	80	1250	44.0	3.30	86.1	23
DUMP3	60	0.1083	0.5530	65	1060	42.3	22.90	50.3	2
DUMP4	40	0.0000	0.0000	80	1540	43.0	29.10	52.4	26
DUMP5	60	0.7317	3.7351	90	1460	43.8	30.80	2.0	-1
DUMP6	100	0.3290	1.6795	80	1770	50.3	27.10	46.0	-22
NEAR1	100	1.4880	7.5961	90	1530	60.3	18.00	31.4	23
NEAR2	60	0.8633	4.4073	85	1350	73.0	9.10	24.4	18
NEAR3	60	0.8600	4.3902	95	1230	33.8	11.90	39.0	21
NEAR4	100	1.1420	5.8298	85	2610	60.5	17.10	22.6	12
NEAR5	100	1.1530	5.8860	80	3290	44.5	13.50	19.4	6
NEAR6	100	1.5240	7.7799	95	1410	39.8	18.00	36.0	23
REF1	80	0.8250	4.2116	90	1000	25.5	20.60	36.4	70
REF2	60	0.6933	3.5394	95	1740	39.0	16.00	26.4	25
REF3	80	0.2700	1.3783	90	1420	40.8	6.31	26.4	23
REF4	80	0.1263	0.6445	95	1870	32.3	6.73	74.2	30
REF5	80	1.5550	7.9382	75	640	43.3	6.79	67.6	20
REF6	100	0.1890	0.9648	85	2100	43.5	11.50	42.2	13

[1] Growth Factor is final weight divided by initial weight.

Microtox solid phase assay

The reference toxicant test showed an EC_{50} of 4,250 mg/kg for a NRC Certified Reference Material (HS-5), which is within the laboratory warning limits of 2,780 – 7,510 mg/kg HS-5.

With the exception of one sample from the reference site (EC_{50} = 640 mg/kg), all test samples were non-toxic (EC_{50} < 1,000 mg/kg) to the Microtox bacteria (Table 4). Pearson correlation analysis showed significant negative relationship ($r = -0.46$ to -0.66, $p = \leq 0.01$, $n = 28$) between the test results and several heavy metals (As, Cu, Pb, Zn), even though the concentrations of these metals were quite low. Microtox results were also significantly correlated with the grain size (from $r = 0.75$, $p = < 0.001$ for sand to $r = -0.57$, $p = 0.002$ for clay; $n = 28$), indicating that the observed Microtox results may be affected by the change of grain size. Grain size has been demonstrated by Benton *et al.* (1995) and Tay *et al.* (1998) as one of the many confounding factors that can affect the data interpretation of Microtox test. This grain size effect was most evident in the samples from the outside area which have the highest percent sand than samples from all other studied areas.

The Microtox endpoints for the outside sediment samples were significantly (Dunn's Q = 12.533 – 14,033, $p < 0.05$) less toxic than those from the disposal, nearfield and reference sites. These differences were most likely caused by the larger particle size in sediments from the outside area. There was no difference (Dunn's Q = 0 – 0.316, $p > 0.05$) between the endpoints of sediment samples from the disposal, nearfield and reference sites.

Sea urchin fertilization assay

An IC_{50} of 290 µg/L copper sulfate was reported for the reference toxicant test which is well within the EC laboratory warning limits of 69.6 – 688 µg/L.

With the exception of one sample from the nearfield site, the fertilization success of all test samples are 25% significantly lower than the fertilization success of the control sample.

However, none of the toxicity endpoints are correlated to grain size or concentrations of sediment contaminants, ammonia, sulfide, Redox or TOC in the test sediments.

No significant difference ($F_{3,24} = 2.329$, $p = 0.100$) between the endpoints was observed between disposal, nearfield, reference sites and the outside area.

Polychaete survival and growth test

The reference toxicant test 96-hour LC_{50} (concentration calculated to cause 50% mortality after 96-h exposure) conducted with cadmium chloride using water only exposures for this study is 6.71 mg/L (5.16 – 8.72 mg/L), well within the laboratory warning limits of 6.51 – 12.2 mg/L. The mean percent survival, weight and growth factor for the polychaetes in the control samples are 96 ± 8.9%, 1.756 ± 0.52 mg and 8.963 respectively, which meets the test validity criterion.

The polychaetes in one of the sediment samples (Station B3) collected near the eastern shoreline had 100% mortality (Table 4). This sampling station is located near the pulp and paper mill on Point Tupper where the seabed was extensively covered by wood fibre (Buckley *et al.*, 1974; Stewart and White, 2001). The sample also has high PCBs (325 µg/kg), Cd (1.34 mg/kg), sediment ammonia (23 µg/g) and sulfide (214 µg/g). The polychaetes in two sediment samples from the disposal site also had low survival rates of 20 and 40%. There was no observed increase of growth and weight of polychaetes exposed to these two sediment samples. The sample which caused the 20% survival contained relatively high sediment PAHs (3.1 mg/kg) and sulfide (86.1 µg/g), while the sample that caused the 40% survival had the second highest ammonia concentration of 29.10 mg/kg among all the samples. Past studies (Corbin *et al.*, 1998, 2000) have shown that *P. cornuta* are sensitive to grain size, TOC and sediment ammonia. However, no significant correlations were observed between these confounding factors and the three endpoints of polychaetes used in this study.

In a study comparing spiked sediment toxicity tests, bioaccumulation/bioavailability, and the national status and trends approach (co-occurring chemical data and biological effects data) to finalize a Canadian sediment quality guideline for PCBs, Lee *et al.* (2004) found a LC20 of

6,180 μg/kg and a LC50 of 10,100 μg/kg Aroclor 1254 PCB for *P. cornuta*. These values were well above the CCME Aroclor 1254 ISQG (63.3 μg/kg) and PEL (709 μg/kg) and the DAS screening level of 100 μg/kg Total PCBs. They were also higher than the maximum Total PCBs concentration (535 μg/kg) detected in this study. It should be noted that the main form of PCBs in the Strait of Canso sediments was Aroclor1260, not the Aroclor 1254 that was used by Lee *at al.* (2004). Lee *et al.* (2004) found an IC25 of 665 μg/kg Aroclor 1254 for the polychaete growth endpoint. This value is also higher than the maximum Total PCB value of this study.

In a test to measure sensitivity of four species to copper-induced toxicity, *P. cornuta* was ranked behind the most sensitive species, *Neanthes arenaceodentata* but had better sensitivity than two other species, *Boccardia proboscidea* and *Leptocheirus plumulosus* (a marine amphipod crustacean). We did not find any significant relationships between the three endpoints of *P. cornuta* and the sediment heavy metals in this study.

No significant difference was observed between the average percent survival ($H = 3.026$, $df = 3$, $p = 0.388$), dry weight ($H = 2.554$, $df = 3$, $p = 0.466$) and growth ($F_{3,24} = 1.624$; $p = 0.210$) of the polychaete between the disposal, nearfield, reference sites and the outside area.

Benthic Communities at All Studied Areas in the Strait

Benthic communities in sediment samples collected for this study were dominated in terms of numbers by polychaete worms. In decreasing order of abundance, these species were *P. steenstrupi, Cossura longocirrata, Nephtys neotena, Goniada maculata, Laonice cirrata and Ninoe nigripes* (Table 5) (Figures 20 - 21). They made up 95% of individuals collected at all sampling stations in this study. About 50% of these dominant polychaete worms were found at the nearfield and disposal site stations. Of this 50%, half of them were *P. steenstrupi* (Table 6).

P. steenstrupi and *C. longocierrata* were the two most dominant species in all samples. *P. steenstrupi* is a mud worm that primary found on soft sticky mud and can occur in very high numbers (Pocklington, 1989). This species is generally found in

Table 5. Abundance (number per m^2) of dominant species at Canso Strait sampling stations. Based on three 0.05 m^2 replicates per station.

Station	Cossura longocirrata		Goniada maculata		Laonice cirrata		Nephtys neotena		Ninoe nigripes		Prionospio steenstrupi		Ophiura sarsi	
	0	S.D.	0	S.D.	0	S.D.	0	S.D.	0	S.D.	0	S.D.	0	S.D.
Nearshore														
A3	0	0	0	0	6.7	11.6	1506.7	945	753.3	189	813.3	404.6	0	0
C3	0	0	0	0	0	0	333.3	473.9	66.7	46.2	453.3	404.6	0	0
Disposal Site														
Dump 1	506.7	427.7	73.3	23.1	20	20	80	80	0	0	960	503.2	13.3	23.1
Dump 2	426.7	271.5	86.7	41.6	13.3	11.5	453.3	551.8	0	0	1166.7	758	0	0
Dump 3	133.3	147.4	73.3	75.7	0	0	486.7	319	0	0	880	365	0	0
Dump 4	1080	940	80	34.6	6.7	11.5	593.3	560.5	6.7	11.5	1833.3	1501.6	20	20
Dump 5	226.7	147.4	100	69.3	13.3	11.5	286.7	292.8	0	0	586.7	560.5	20	20
Dump 6	190	240.4	80	0	0	0	260	339.4	10	14.1	870	1202.1	10	14.1
Nearfield														
Near 1	326.7	325.8	33.3	23.1	20	20	486.7	317.7	6.7	11.5	813.3	807.5	6.7	11.5
Near 2	33.3	30.6	20	0	13.3	11.5	753.3	41.6	33.3	11.5	566.7	133.2	6.7	11.5
Near 3	26.7	23.1	66.7	83.3	0	0	380	351.6	33.3	41.6	480	365	13.3	23.1
Near 4	46.7	80.8	100	20	6.7	11.5	40	34.6	0	0	80	34.6	6.7	11.5
Near 5	833.3	1218.3	126.7	75.7	20	20	200	163.7	0	0	713.3	241.9	0	0
Near 6	1186.7	889.1	86.7	70.2	13.3	11.5	153.3	122.2	0	0	1266.7	800.3	6.7	11.5
Reference														
Ref1	366.7	270.1	100	60	6.7	11.6	420	190.8	6.7	11.6	1013.3	578.4	6.7	11.6
Ree2	1673.3	372.2	60	34.6	0	0	620	163.7	0	0	1486.7	215.7	0	0
Ref3	1686.7	1313.7	40	40	0	0	240	365	0	0	993.3	837.7	0	0
Ref4	1086.7	1003.3	60	20	20	20	53.3	92.4	13.3	23.1	580	399.5	0	0
Ref5	393.3	127	66.7	23.1	6.7	11.6	186.7	136.1	13.3	23.1	1380	659.4	6.7	11.6
Ref6	860	190.8	60	69.3	6.7	11.6	346.7	246.9	6.7	11.6	1460	177.8	0	0

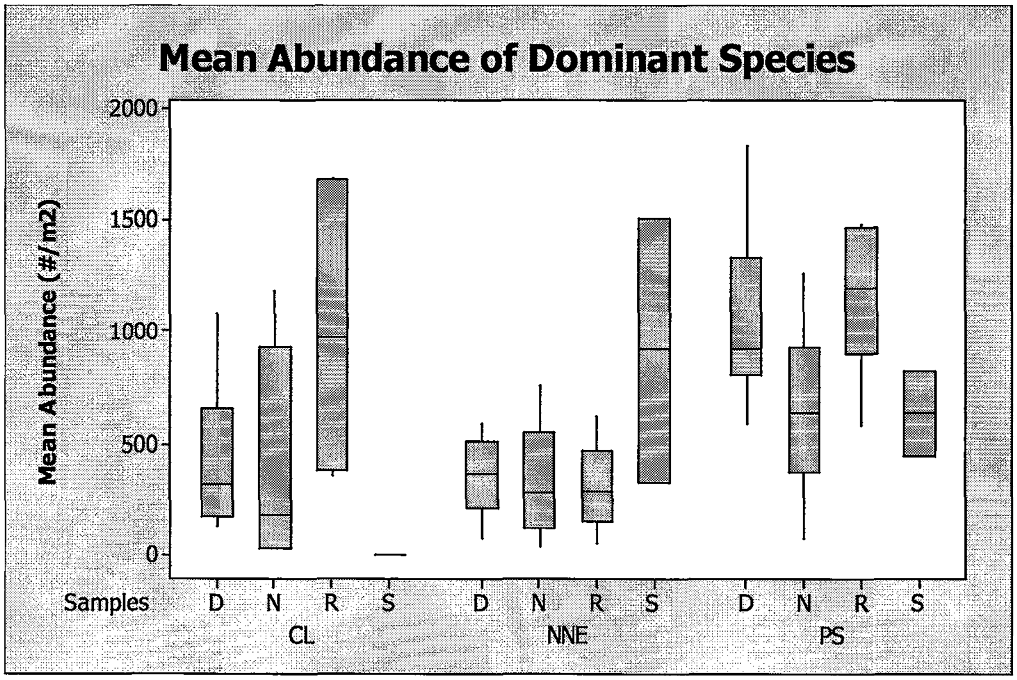


Figure 20. Mean abundance of dominant species (CL – *Cossura longocirrata*; NNE – *Ninoe nigripes*; PS – *Prionospio steenstrupi*) at the four studied areas (D – disposal site; N – nearfield site; R – reference site; S – outside area).

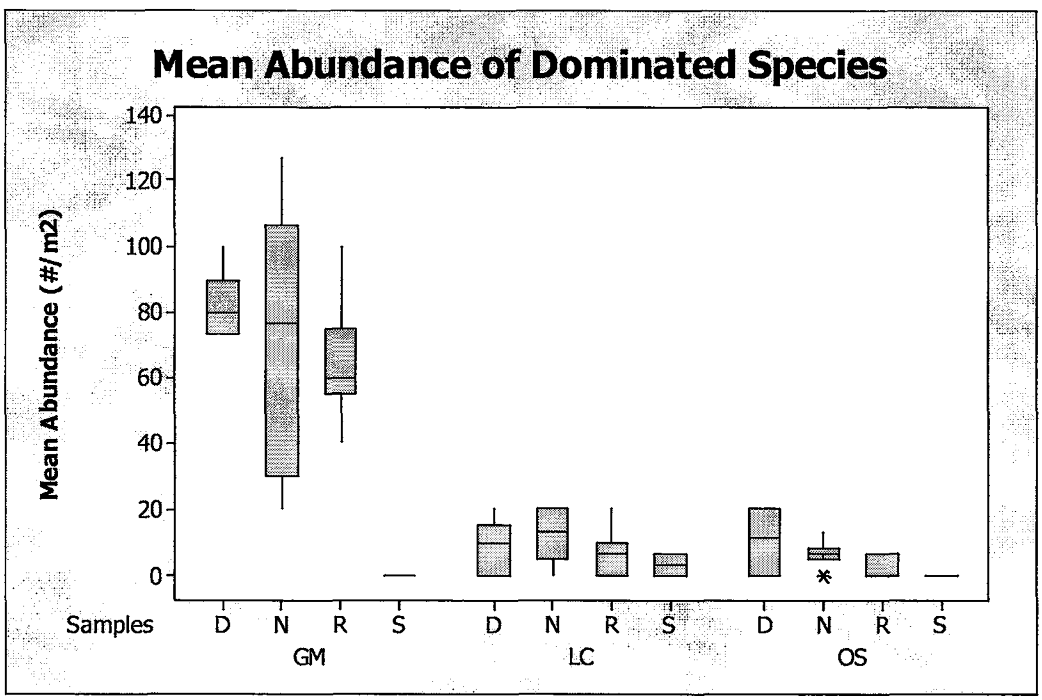


Figure 21. Mean abundance of dominant species (GM – *Goniada maculate*; LC – *Laonice cirrata*; OS – *Ophiura sarsi*) at the four studied areas (D – disposal site; N – nearfield site; R – reference site; S – outside area).

Table 6. Abundance (number per m^2) of dominant polycheate species found at the 2004 studied sites and in the 1987 disposal site study.

Polychaete Species	2004 All Sites	2004 Disposal and Nearfield Sites	2004 Disposal Site	2004 Reference Site	1987
Prionospio steenstrupi	54900	30360	20100	20740	1262
Spiophones kroyerl					437
Nereimyra punctata					397
Cossura longocirrata	33200	15000	8540	18200	286
Nephtys neotena	23640	12500	6480	5600	121
Goniada maculata	3920	2760	1560	1160	109
Ninoe nigripes	2840	260	40	120	195
Laonice cirrata	520	380	180	120	50
Eteone longa					153
Euchone incolor					106

disturbed seafloor where it is undergoing phase two of recovery succession (Pearson and Rosenburg, 1978). In a study to develop marine biotic index for the development of the ecological quality of soft-bottom benthos within European estuarine and coastal environments, *P. steenstrupi* was grouped in Ecological Group IV as the second-order opportunistic species that are commonly found in "slight to pronounced unbalanced environment" (Barja *et al.,* 2000). *C. longocierrata* was the third most numerous species but was absent in the two samples collected from the shoreline. It is also a mud loving worm which is primary found on muddy to silty substrates and always occur in large numbers. It is described by Wlodarska *et al.* (2007) as a high sedimentation resistant polychaete. The number of this polychaete is significantly correlated with the percent clay in the sediment ($r = 0.56$, $p = 0.01$, $n = 20$). *Nephtys neotena* was the next most dominant species in this study. The highest number of this worm was found in Station A3 which is the station along the shoreline where environmental degradation caused by industrialization of the Strait (Scarratt & Associates, 1994). The sediment sample collected from this station has the highest sand content (49.4%), PCBs (535 μg/kg) and third highest TOC (6.99%). The sample was described as crumbly black, highly organic sediment (fine wood fibre) with sulfide smell and mussel shells covered with a thin, brown 1 – 2 cm oxidized layer. This species of polychaete worm belonged to genus *Nephtys* which was found in phase 2 of recovery succession in areas under moderate organic inputs (Pearson *et al.*, 1983; Pearson and Rosenburg, 1978; Grant *et al.*, 1995).

The number of polychaetes *G. maculata, L. cirrata* and *N. nigripes* were less than those polychaetes described above, but their numbers were still quite high compared to other groups of organisms. Pearson and Rosenburg (1978) described *G. maculata* as another polychaete worm that are characteristic of Phase 2 of succession under moderate organic input. *L. cirrata* is not known to associate with pollution (Pocklington 1989). This polychaete was absent from some of the sampling stations and the number is not as high as the other polychaetes in this group. *N. nigripes*, which is able to inhabit and thrive in the inhospitable environment dominated by wood pulp, was mostly found in the two shoreline samples where layer of wood fiber with thickness of up to 4 metres thick was observed (Scarratt & Associates, 1994).

In addition to the groups of polychaete, the other most abundant species was a nemaertean of the genus *Cerebratulus*. *Cerebratulus* are bivalve predators (Roweel and Woo, 1990; Grant *et al.*, 1995) which are most commonly found in fall at mussel culture sites beneath the mussel lines. However, this nemertean was most abundant at the outside stations with only 6% found at the nearfield and disposal site stations. The brittlestar, *O. sarsi*, was observed in mean numbers as high as 20 individuals at two disposal site samples. They were mostly collected from the stations at the disposal site and nearfield site, a few from the reference site and none from the two shoreline stations.

Other groups of organisms were generally sparsely distributed and occurred individually. Included in these groups were the bivalves *Cerastoderma pinnulatum* and *Nucula tenuis*, the gastropod *Oenopota turricula*, the oedicerotid amphipods (*monoculodes, Metopella?* and *Westwoodilla caecula*), and the cumaceans (*Eudorellopsis integra?* and *Leucon nascicoides*).

Several other species that can be considered characteristic of the soft bottom environments of the Strait occurred occasionally in the samples. They included the burrowing sea cucumber *Caudina arenata* from the reference site, the shrimp *Aargis dentate* and the snake blenny (*Lumpenus lumpretaeformis*).

Community Measures

Univariate measurement

Table 7 showed the community measures at all the studied areas. Community measures, including Shannon Wiener Diversity, Pielou's Evenness and biomass were not significantly different between disposal, nearfield and reference sites with the exception of the species abundance measurement which was significantly higher at the reference site than those measured at the nearfield site ($F_{1,30} = 8.09$, $P = 0.008$). Measures were consistent between stations within each of the areas, showing no significant difference between stations except for biomass measurements at the reference and nearfield site stations, where significant differences between stations occurred ($F_{4,45} = 3.28$, $P = 0.019$) indicating that the pattern of variability differed between areas.

Abundance (number/m^2) of organisms of the community in this study (ranged from 413.3 mean number of individuals/m^2 at station 4 of the nearfield site to 3967 at station 3 of the reference site) was rated as moderate to high when compare with those in the rating table of biological community for ocean disposal site developed by Envirosphere Consultant Ltd. for Environment Canada (1999) (Table 8). The number of species (ranged from 5 species in station 4 of the nearfield site to 10 species in station 2 of the reference site) and the Pielou's Evenness (ranged from 0.541 in station 6 to 0.825 in station 4, both of the nearfield site) were moderate to low to moderate. Shannon-Wiener Diversity (ranged from 0.426 in station 3 of the reference site to 0.749 in station 5 of the disposal site) was low to moderate while biomass (ranged from 5.8 grams/m^2 wet weight in station 3 of the disposal site to 122.4 grams/m^2 wet weight in station 2 of the reference site) ranged from low to moderate to high. The rating showed that the communities living at the disposal and reference sites have the characteristics of low to moderate diversity, species number, and evenness with high abundance and a wide range of biomass measurements when comparing with unpolluted nearshore biological communities in Atlantic Region (Environment Canada 1999).

Table 7. Community measures at all studied areas of the Canso Disposal Site. Based on 3 replicates (0.05 m^2) per station.

Station	Abundance (number/m2)		Species per 0.05 m^2 sample		Shannon-Wiener Diversity (Log10)		Pielou's Evenness		Simpson's Index		Biomass (grams/m2 wet weight)	
	0	S.D.	0	S.D.	0	S.D	0	S.D.	0	S.D.	0	S.D.
Nearshore												
A3	3647	1544	13	2.9	0.69	0.033	0.621	0.071	0.269	0.039	51.7	38.1
C3	1053	891	6	1.5	0.588	0.096	0.804	0.156	0.318	0.056	8.3	6.7
Disposal Site												
Dump 1	2080	1422.8	7	1	0.501	0.059	0.6	0.116	0.425	0.081	46.3	33.5
Dump 2	2720	1119.5	7.7	0.6	0.5	0.027	0.566	0.028	0.4	0.038	43.9	21.2
Dump 3	1520	904.2	6.7	2.1	0.5	0.031	0.627	0.105	0.405	0.056	5.8	5
Dump 4	3933.3	2435.3	6	1	0.482	0.021	0.627	0.072	0.389	0.042	21.3	15.8
Dump 5	1000	1128	9	1	0.749	0.143	0.793	0.186	0.234	0.1	92.3	119.4
Dump 6	1446.7	1270.6	8	1	0.651	0.146	0.726	0.181	0.305	0.123	31.1	4.7
Nearfield												
Near 1	1686.7	1436.4	8.3	2.5	0.587	0.02	0.658	0.113	0.334	0.03	46.4	39.4
Near 2	1400	87.2	8.7	0.6	0.523	0.037	0.558	0.024	0.398	0.025	21	11.2
Near 3	1353.3	674.5	7.7	2.1	0.527	0.073	0.61	0.128	0.382	0.047	13.3	10.5
Near 4	413.3	291.4	5	1.7	0.565	0.171	0.825	0.054	0.335	0.149	19.9	28.7
Near 5	886.7	421.6	7.3	0.6	0.635	0.09	0.737	0.129	0.318	0.113	21.8	10.2
Near 6	2780	1594	7	1	0.454	0.007	0.541	0.034	0.438	0.01	21.5	14.7
Reference												
Ref1	1967	1052	7	1.5	0.566	0.039	0.662	0.054	0.346	0.022	46	35.7
Ref2	3967	352	10	1.2	0.539	0.038	0.549	0.036	0.348	0.027	122.4	131.5
Ref3	3040	2598	6	2.5	o.426	0.075	0.553	0.035	0.452	0.064	33.1	23.8
Ref4	1887	1398.5	7	2	0.503	0.103	0.628	0.225	0.407	0.12	87	91.5
Ref5	2093	925.7	7	1	0.465	0.033	0.552	0.013	0.474	0.023	56.6	33.9
Ref6	2807	685.4	8	2	0.496	0.057	0.565	0.04	0.395	0.057	39.2	13.6

Table 8. Ratings of biological community measures based on statistical distributions in 0.1 m^2 samples. (extracted from Environment Canada 1999)

Rating	Number of Species	Abundance (No./m^2)	Biomass (g/ m^2)	Shannon Wiener Diversity	Pielou's Evenness
LOW	0 - 4	0 - 130	0 – 1.3	0 – 0.41	0 – 0.33
LOW TO MODERATE	4 - 8	130 - 350	1.3 – 7.2	0.41 – 0.64	0.33 – 0.66
MODERATE	8 - 16	350 - 865	7.2 – 45.2	0.64 – 0.88	0.66 – 0.78
MODERATE TO HIGH	16 - 32	865 - 1730	45.2 - 90	0.88 – 1.24	0.78 – 0.88
HIGH	> 32	> 1730	> 90	> 1.24	0.88 – 1.00

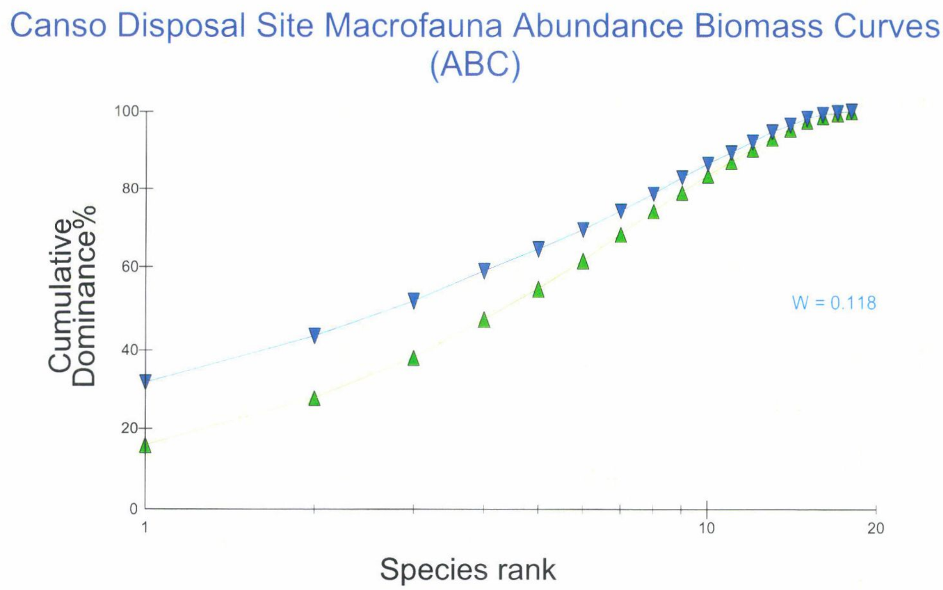
ABC curves for the disposal and reference sites were quite similar (Figure 22), showing a balance of distribution of biomass and abundance among species, which suggests that the communities are not stressed (Warwick, 1986; Warwick *et al.*, 1987). The pattern of colonization after the bottom community has been stressed usually involves development of small species which typically occur in large numbers, and give an ABC plot in which the abundance curve has a higher initial slope than the biomass curve.

Cluster analysis showed that the communities were very similar between disposal and reference sites. Four of the six nearfield site stations have very different communities comparing with those at the disposal and reference sites (Figure 23).

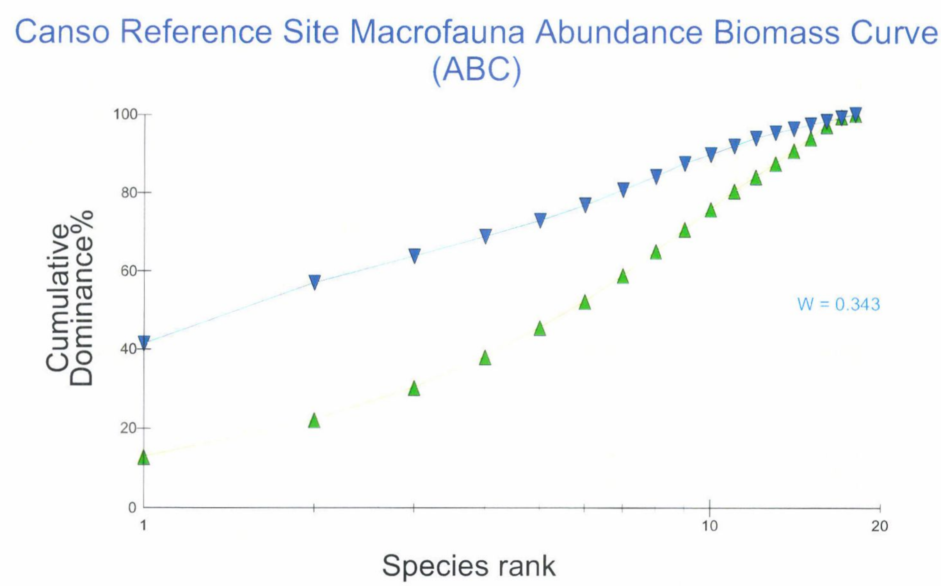
Overall, it can be concluded that the benthic communities living at the nearfield, disposal and reference sites, were similar in their abundance in species and numbers, species richness and evenness and their diversity.

Community measurements at the disposal site from 1987 to 2004

Twelve randomly selected stations of the seventeen sampled in 1987 (Figure 24) were compared with the twelve stations of the combined disposal site and nearfield areas from 2004. Number of species and Shannon-Wiener Diversity were significantly lower in 2004 than in 1987 (ANOVA,



(A)



(B)

Figure 22. Abundance-Biomass comparison (ABC) curves of the benthic communities at (A) disposal site and (B) reference site. (▲ – Abundance; ▼ – Biomass).

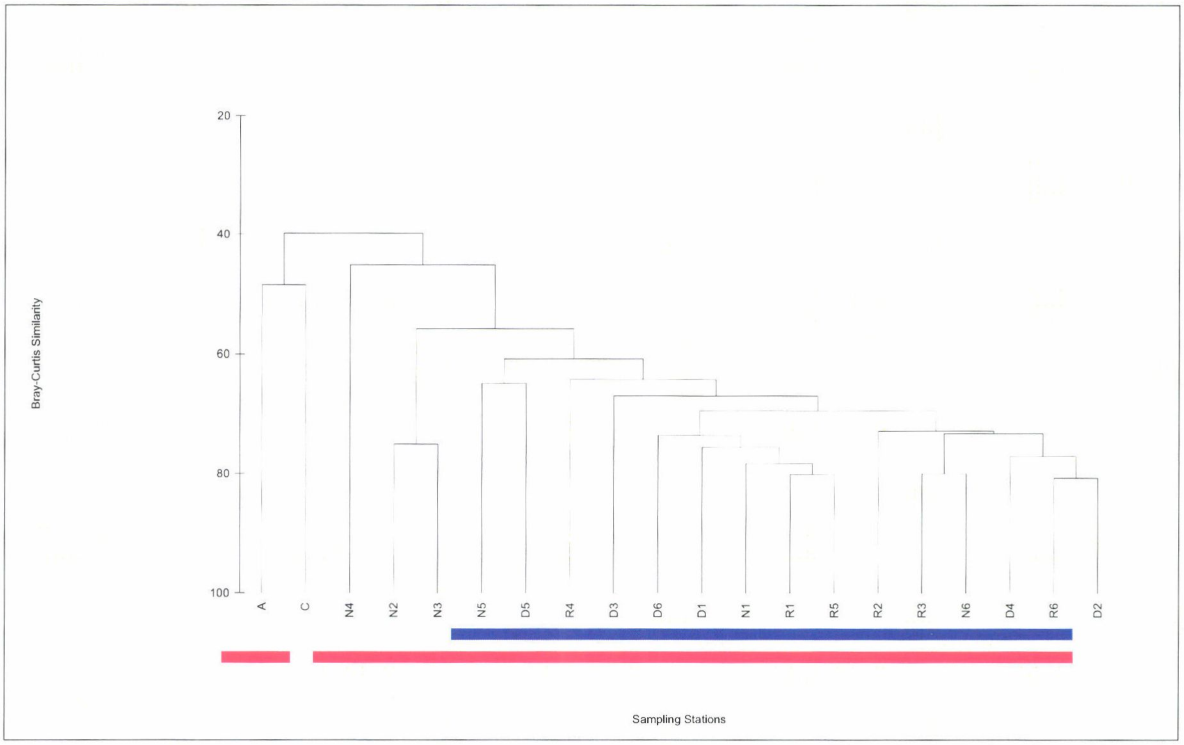


Figure 23. Bray-Curtis similarity plot for all benthic community data collected from this study. The blue bar shows the closer grouping of the benthic communities from the reference and disposal sites. The red bars show the different grouping of the benthic communities from the outside area.

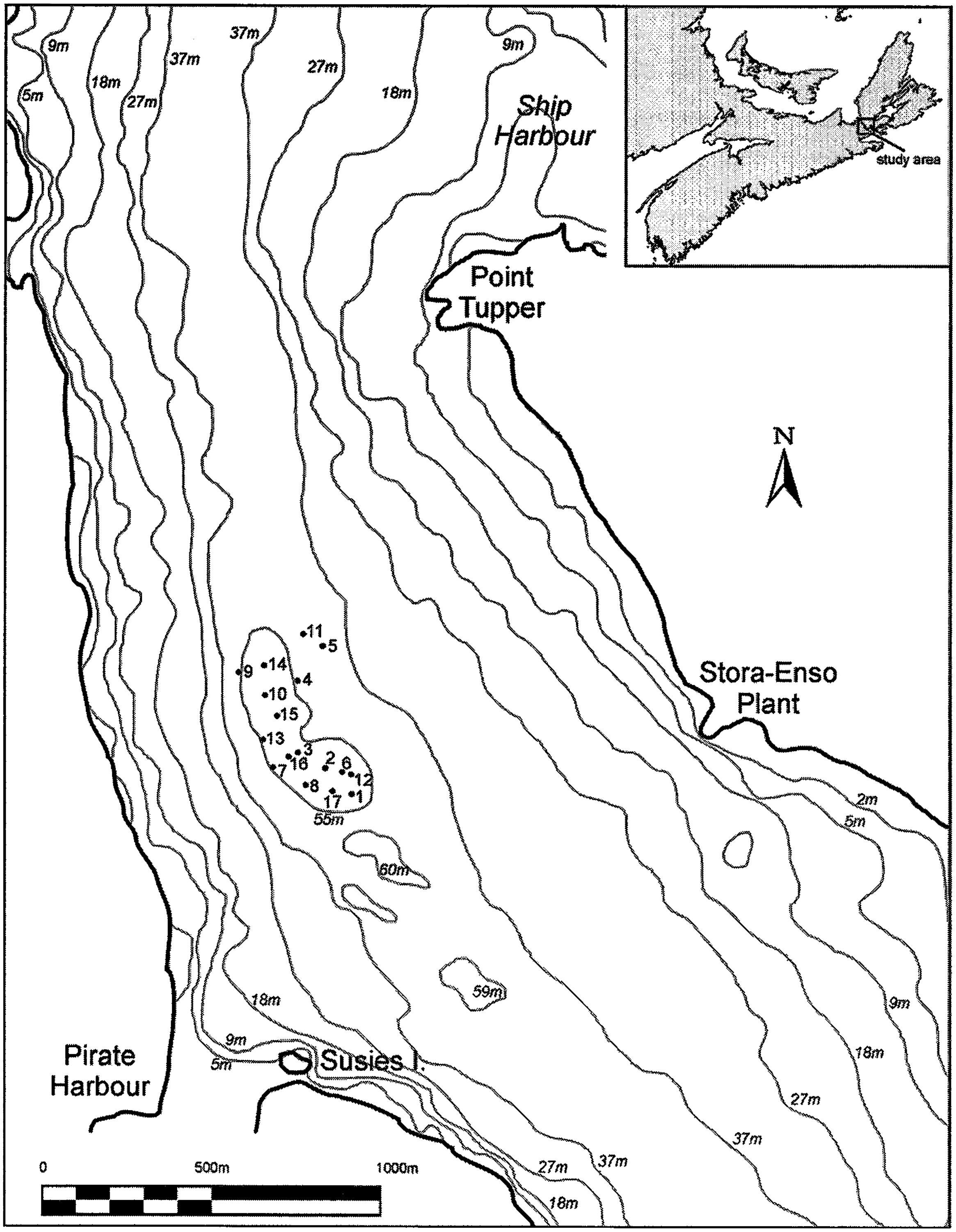


Figure 24. Canso disposal site area showing the seventeen 1987 sampling stations. Twelve randomly selected stations (1, 2, 3, 6, 8, 9, 10, 12, 14, 15, 16, and 17) were used for temporal benthic community study.

$p < 0.001$) and Simpson's Index was significantly higher ($p < 0.001$). Several species occurred in 1987 which were not present in 2004, including two dominants, the polychaetes *Nereimyra punctata* and *Spiophanes kroyeri* (Table 6), and the difference accounted for the approximately 7.4 species per sample versus 10.3 observed in 1987. Number of species is an element in the calculation of the Shannon Wiener Diversity Index and higher values lead to higher values of the Shannon Wiener Index, and so the difference in number of species between years was the important factor in the differences of the other measures. The lower Simpson's Index in 1987 also reflects the larger number of dominant species (Simpson's Index is a measure of evenness and is highest when few species dominate). Biomass was not significantly different ($p < 0.05$) between 1987 and 2004.

Of the dominant species, numbers typically increased from 1987 to 2004 except for the polychaete *N. nigripes*, which decreased significantly (ANOVA, $P < 0.001$). Increases in abundance of *C. longocirrata, N. neotena, P. steenstrupi* and *G. maculata* between 1987 and 2004 were significant ($p < 0.001$ for the first three species and $p < 0.05$ for *Goniada*). The 1987 data set showed greater patchiness in abundance of dominant species with significant differences occurring between stations for *Goniada, N. neotena and Prionospio* ($p < 0.05$), as well as for *Ninoe* ($p < 0.001$). Overall abundance and number of species per sample also showed significant differences between stations in 1987 ($p < 0.01$ and $p < 0.05$ respectively).

The present study focused on evaluation of the benthic community and assessment of differences between the disposal site and adjacent areas, as well as with a comparable study carried out in 1987 (OceanChem, 1987; Pocklington, 1987). Absence of differences at these spatial and temporal scales is an indication of the lack of impacts and/or recovery of the disposal site from disposal impacts. Benthic studies have been carried out at this disposal site, beginning in 1983 and summarized for the 1983-85 period by Rath (1986); however the early studies cannot be readily compared with those in the 1987 to present period because of the difference of sampling gears used between the studies. Also, the Martec (1984) study only identified the animals to the level of Phylum.

Community composition at the disposal site was comparable in 1987, in 2000 (in a survey that was undertaken for an ocean dumping permit for the Georgia Pacific Marine Terminal disposal) and at present, in terms of several dominant polychaete worm species, including *P. steenstrupi, G. maculata, N. neotena, C. longocirrata* and *L. cirrata.* The brittlestar *O. sarsi* has also been found characteristically in the area.

Differences in dominant species in an area over time are common in benthic environments and marine communities in general. Absence in 2004 of two of the dominant species of polychaetes, *Nereimyra punctata* and *Spiophanes kroyeri,* found in 1987 is therefore not surprising although it is possible that they represented an early response to disposal activities conducted before 1987. Neither species is known in particular to respond to disturbance, but spionids, the polychaete family of which *Spiophanes* is a member, exhibit lifestyles which favour colonization of disturbed areas and resistance to low oxygen conditions. Many of the same dominant species in 1987 were found, however, in the 1999 disposal site survey (Seatech 1999), and in the present study. Of the remaining groups of animals represented in the samples, there were few species which occurred in all three surveys: the brittle star *O. sarsi,* the cumacean *Leucon nascicoides (L. nascica* in 1987), and nemerteans (identified as *Cerebratulus* sp only in the 2004 study).

The soft bottom of the sites in this study may itself be a factor in the reduced diversity, including the possibility that tidal currents re-suspend some of it. The polychaete *C. longocirrata* which was a dominant in 1987 and 2004 can tolerate fine re-suspended sediments (Olsgard and Hasle, 1993; Envirosphere Consultants Ltd., 2003) and *P. steenstrupi* commonly occurs in organically enriched harbours where it deals with fine particle loading (e.g. Saint John Harbour, Halifax Harbour). Soft bottom environments in Saint John Harbour, albeit at shallower depths and higher current strengths, typically have reduced species richness, and moderate Shannon Wiener Diversity as occurs in the present study.

Disposal Site Benthic Community Recovery

The group of dominant polychaete species at the disposal site in this study is similar both in species composition and in number to those at the reference site. The most dominant polychaete species, *P. steenstrupi,* found both at the disposal and reference sites is a species that can occur in very high numbers in disturbed seafloor where it is undergoing phase two of recovery succession. The third and fourth most dominant polychaete, *N. neotena* and *G. maculata*, which are also phase two worms but not as numerous as *P. steenstrupi*, also occurred in very similar number at the disposal and reference sites.

In the 1987 survey, it was concluded that the increased number of species and individuals, especially the second stage recolonizers, indicated that the infaunal community at the disposal site had undergone a recolonization process after the initial disposal impact. In this study, with the exception of the increase of individual of several dominant species, the species of polychaetes found at the disposal site remained quite similar to those found in the 1987 survey, indicating that the community at the site may still remain in the second stage of the recolonization process.

To assess the degree of site recovery for this study, we may need to understand the general marine environment within the southern portion of the Strait where the disposal and reference sites are located. Past studies showed that the construction of the causeway in the Strait of Canso in 1954 has changed a natural marine environment in the southern portion of the Strait to a "bisected fjord" with abnormal water quality and geochemical and biological characteristics (Buckly *et al.*, 1974). The change of environment was mainly caused by the reduction of currents, lack of stratification of water, and the continued discharges of industrial wastes from the surrounding lands into the Strait. Wagner (1975) and Schafer *et al.* (1975) found a "molluscan barren zone" in the deeper waters of the Strait, south of the Causeway. Broken and dead shells of bivalve and gastropod, wood fibers and wood chips were frequently observed in grab sediment samples, especially around the disposal site area (OceanChem, 1987; Pocklington, 1987). Martech (1984) reported a less diverse benthic community in this portion of the Strait dominated by polychaetes and nemerteans with no mollusks while echinoderms were represented only by the brittle star, *O. sarsi*. A similar observation was reported by Seatech in 1986 (Seatech, 1986). Dense beds of ophiuroids are always low-diversity communities (Rex, 1983) and are only found where predation pressure is low (Aronson and Sues, 1987; Fujita and Ohta, 1989). In this study, the seafloor photographs showed that much of the seafloor is covered by thin veneer of fine sediments and is densely populated by the brittle star *Ophiura sarsi* (Kostylev *et al.*, 2005) (Figures 11 and 12).

The disposal and reference sites of this study were located in the deeper waters of the Strait where the "molluscan barren zone" and cold waters with lack of stratification and slow currents were reported. The communities of these two sites were very similar in species composition, diversity and other community measures. With this unusual environmental setting, it is quite possible that the impacted community at the disposal site may have already fully recovered from the last disposal impacts which occurred in 1999, except that the community remained in the second stage of the recolonization process because of the adverse local environment caused by the construction of the causeway.

The benthic communities in the southern portion of the Strait may still be developing as they colonize the new environment created when the Strait of Canso was closed by the Canso Causeway in 1954. Second stage colonizers, *N. neotena* and *P. steenstrupi*, were the two dominate polychaetes found in the benthic communities living at the two shoreline stations, A3 and C3, and all the disposal, reference and nearfield stations (Table 5).

The use of the recovery of benthic communities at a dredged materials disposal site to assess disposal impacts has been extensively studied by researchers (Levings *et al.*, 1985; Rees *et al.*, 1992; Harvey *et al.*, 1998; Roberts and Forrest, 1999; Smith and Rule, 2001; Zimmerman *et al.*, 2003; Bolam and Rees, 2003; Bolam *et al.*, 2006; Wiber *et al.*, 2007). Recovery time of an impacted community ranged from 18 months (Zimmerman *et al.*, 2003) to about two years (Harvey *et al.*, 1998). For this present study, we conclude that, four years after the 1999 disposal, the benthic community at the Canso disposal site has fully recovered to the stage that it is similar to the community found at the reference site.

Recovery time estimates for benthic communities depend not only on biological responses, but the analytical approach used to evaluate and report the data (Bolam and Rees, 2003; Newell *at al.*, 1998). Several studies of dredged material disposal impacts showed that while species composition of a benthic community had not returned to the pre-dredging composition or had failed to converge with reference sites within 12 months, the univariate indices had recovered within only a few months (Bolam and Whomersley, 2005; Johnson and Nelson, 1985; Wilber *et al.*, 2007). To set a standard for matching the pre-disturbance species composition, Newell at al. (1998) suggested a method that includes the establishment of a community that is capable of maintaining itself and in which at least 80% of the species diversity and biomass has been restored.

CONCLUSIONS

In this study, none of the disposal site sediments contained elevated concentrations of the sediment of concern (e.g. Cd, PCBs and PAHs) and none of the sediments was toxic to the test organisms in the amphipod and Microtox solid phase tests. With the exception of As, the concentrations of other heavy metals, PCBs and PAH were not significantly different between the disposal and the reference sites. Two sediment samples from the disposal site were toxic to polychaete survival and growth while all except one sample were toxic to sea urchin gametes. For all the toxicity endpoints used in this study, none was statistically correlated to sediment contaminants. While the polychaete test results may be confounded by the effects of particle size, ammonia and sulfide, it is not known what caused the toxicity in the sea urchin test. In comparisons between the study areas, no statistically significant difference in sediment toxicity was detected for any of the toxicity tests.

The 30% increase of silt/clay mixtures at the disposal site four years after the disposal of 39,529 cu.m. of dredged materials is unusually high considering that closed to 100% of the dredged materials disposed of at the site in 1999 were gravel/sand mixtures. It is possible that the particle size data of the pre-dredged sediment samples were misrepresentative of the sediment grain size at the dredged areas or that some of these fine grain materials may have migrated from other parts of the Strait to the deeper water areas where the disposal site is located.

In addition to the lack of high sediment contaminants and contaminant related toxicity detected at the disposal site in this study, benthic community surveys also showed that the benthic community at the disposal site has recovered to the stage that is similar to the community at the reference site.

Based on the 1983 Martec's disposal site selection study and the physical survey conducted in this study, the Canso disposal site still has the capacity to receive more than 300,000 cu.m. of dredged materials before significant water depth changes would occur. The site is located in a very stable environment within a low current regime with colder temperature and no water stratification comparing to the surrounding areas. Recovery of the community at the site may be slow, but it has demonstrated that it can sustain and recover from the impacts of multiple disposals of dredged materials. Use of this site for future dredging projects within the Canso Strait is recommended.

The findings of Cd and PCBs hot spots in the Strait sediments, especially along the eastern shoreline, suggested that special attention should be paid to sediment chemical characterization of any dredging projects within the Strait to ensure that potential risk of sediment contamination can be minimized when dredging and disposing dredged materials that may contain elevated levels of Cd and PCBs.

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